Guidelines for Preparing Economic Analyses

December 17, 2010

(updated May 2014)

National Center for Environmental Economics
Office of Policy
U.S. Environmental Protection Agency

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Acronyms and Abbreviations

ABC	Air Benefit and Cost (Group)
AC	annualized costs
ACN	AirControlNET
ADP	Action Development Process
BAT	best available technology
BCA	benefit-cost analysis
BLS	Bureau of Labor Statistics
BMP	Best Management Practice
BPT	best practicable technology
CA	conjoint analysis
CAA	Clean Air Act
CAFO	Combined Animal Feeding Operations
CAAA	Clean Air Act Amendments
CAIR	Clean Air Interstate Rule
CAMR	Clean Air Mercury Rule
CBO	Congressional Budget Office
CE	certainty equivalent
CEA	cost-effectiveness analysis
CEM	continual emissions monitoring
CEQ	Council on Environmental Quality
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFC	chlorofluorocarbons
CFOI	Census of Fatal Occupational Injuries
CFR	Code of Federal Regulations
CGE	computable general equilibrium
CO	carbon monoxide
CO_2	carbon dioxide
COI	cost of illness
CPI	Consumer Price Index
CR	contingent ranking
CS	compensating surplus
CV	contingent valuation
CV	compensating variation
DALY	disability-adjusted life year
DICE	Dynamic Integrated model of Climate and the Economy
DOE	Department of Energy
DOT	Department of Transportation
DWL	deadweight loss
EA	economic analysis
EBIT	earnings before interest and taxes
EEAC	Environmental Economics Advisory Committee
EIA	economic impact analysis
ELG	Effluent Limitation Guidelines

EO	Executive Order
EPA	Environmental Protection Agency
ES	equivalent surplus
EV	equivalent variation
EVRI	Environmental Valuation Reference Inventory
FINDS	Facility Index Data System
FR	Federal Register
FTE	full-time equivalent employment
GDP	gross domestic product
GHG	greenhouse gases
GIS	Geographic Information System
HCFC	hydrochlorofluorocarbon
Hg	mercury
IAM	integrated assessment model
ICR	Information Collection Request
IEc	Industrial Economics, Inc.
IMPLAN	Impact Analysis for Planning
IPCC	Intergovernmental Panel on Climate Change
IPM	Integrated Planning Model
LP	linear programming
MAC	marginal abatement cost curve
MD	marginal external damage curve
MR	marginal revenue
MPC	marginal private costs
MSC	marginal social costs
MSD	marginal social damages
NAAQS	National Ambient Air Quality Standards
NAICS	North American Industrial Classification System
NB	net benefits
NEI	National Emissions Inventory
NEPA	National Environmental Policy Act
NESHAP	National Emission Standard for Hazardous Air Pollutant
NFV	net future value
NH_3	ammonia
NIOSH	National Institute of Occupational Safety and Health
NOAA	National Oceanic and Atmospheric Administration
NO_x	nitrogen oxide
NPDES	National Pollutant Discharge Elimination System
NPV	net present value
NWPCAM	National Water Pollution Control Assessment Model
OAQPS	Office of Air Quality Planning and Standards
OCC	opportunity cost of capital
OECD	Organization for Economic Cooperation and Development
OGC	Office of General Counsel
OIRA	Office of Information & Regulatory Affairs
OLS	ordinary least squares

OMB	Office of Management and Budget
OSHA	Occupational Safety and Health Administration
OTEA	Office of Trade and Economic Analysis
PACE	Pollution Abatement Costs and Expenditures
PAOC	pollution abatement operating cost
$PM_{2.5}$	particulate matter, 2.5 microns in diameter or less
PM_{10}	particulate matter, 10 microns in diameter or less
POTW	publicly-owned (wastewater) treatment work
PRA	Paperwork Reduction Act
PVC	present value of costs
QA	quality assurance
QALY	quality-adjusted life year
R&D	research and development
RAPIDS	Rule and Policy Information Development System
RACT	Reasonably Available Control Technology
RCRA	Resource Conservation and Recovery Act
RDD	random digit dialing
REMI	Regional Economic Models, Inc.
RFA	Regulatory Flexibility Act
RIA	regulatory impact analysis
RUM	random utility maximization
S&P	Standard & Poor's
SAB	Science Advisory Board
SAM	social accounting matrix
SBA	Small Business Administration
SBREFA	Small Business Regulatory Enforcement Fairness Act
SCC	social cost of carbon
SIC	Standard Industrial Classification
SISNOSE	significant economic impact on a substantial number of small entities
SO_2	sulfur dioxide
SWC	Survey on Working Conditions
TAMM	Timber Assessment Market Model
TMDL	Total Maximum Daily Loadings
TRI	Toxics Release Inventory
TSLS	two-stage least squares
UMRA	Unfunded Mandates Reform Act
UPF	utility possibility frontier
USC	United States Code
VOC	volatile organic compounds
VSL	value of statistical life
VSLY	value of a statistical life-year
WTA	willingness to accept
WTP	willingness to pay

Glossary

Annualized value
An annualized value is a constant stream of benefits or costs. The annualized cost is the amount that a party 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. Similarly, the annualized benefit is the amount that a party would accrue at the end of each period *t* to add up to the same benefit in present value terms as the stream of benefits being annualized.

Baseline
A baseline describes an initial, status quo scenario that is used for comparison with one or more alternative scenarios. In typical economic analyses the baseline is defined as the best assessment of the world absent the proposed regulation or policy action.

Benefit-cost analysis (BCA)
A BCA evaluates the favorable effects of policy actions and the associated opportunity costs of those actions. It 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. The calculation of net benefits helps ascertain the economic efficiency of a regulation.

Benefits
Benefits are the favorable effects society gains due to a policy or action. Economists define benefits by focusing on changes in individual well-being, referred to as welfare or utility. Willingness to pay (WTP) is the preferred measure of these changes as it theoretically provides a full accounting of individual preferences across trade-offs between income and the favorable effects.

Benefit-cost ratio
A benefit-cost ratio is the ratio of the net present value (NPV) of benefits associated with a project or proposal, relative to the NPV of the costs of the project or proposal. The ratio indicates the benefits expected for each dollar of costs. Note that this ratio is not an indicator of the magnitude of net benefits. Two projects with the same benefit-cost ratio can have vastly different estimates of benefits and costs.

Cessation lag
Cessation lag is the time interval between the cessation of exposure and the reduction in risk. See *latency* for a definition of a related but distinct concept.

Command-and-control regulation
Command-and-control regulation requires polluters to meet specific emission-reduction targets defining acceptable levels of pollution. This type of regulation often requires the installation and use of specific types of equipment to reduce emissions. Command-and-control regulations usually impose the same requirements on all sources, although new and existing sources, taken as groups, are frequently subject to different standards.

Compliance cost
A compliance cost is the expenditure of time or money needed to conform to government requirements such as legislation or regulation. In the case of environmental regulation, these direct costs are associated with: (1) purchasing, installing, and operating new pollution control equipment; (2) changing a production process by using different inputs or different mixtures of inputs; and (3) capturing waste products and selling or reusing them.

Consumption rate of interest
Consumption rate of interest is the rate at which individuals are willing to exchange consumption over time. Simplifying assumptions, such as 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.

Cost-effectiveness analysis (CEA)
CEA examines the costs associated with obtaining an additional unit of an environmental outcome. It is designed to identify the least expensive way of achieving a given environmental quality target, or the way of achieving the greatest improvement in some environmental target for a given expenditure of resources.

Costs

Costs are the dollar values of resources needed to produce a good or service; once allocated, these resources are not available for use elsewhere. *Private costs* are the costs that the buyer of a good or service pays the seller. *Social costs*, also called *externalities*, are the costs that people other than the buyers are forced to pay, often through non-pecuniary means, as a result of a transaction. The bearers of social costs can be either particular individuals or society at large.

Distributional analysis

Distributional analysis assesses changes in social welfare by examining the effects of a regulation across different subpopulations and entities. Two types of distributional analyses are the economic impact analysis (EIA) and the equity assessment.

Economic efficiency

Economic efficiency refers to the optimal production and consumption of goods and services. This generally occurs when prices of products and services reflect their marginal costs, or when marginal benefits equal marginal costs.

Economic impact analysis (EIA)

An EIA examines the distribution of monetized effects of a policy, such as changes in industry profitability or in government revenues, as well as non-monetized effects, such as increases in unemployment rates or numbers of plant closures.

Elasticity of demand

Elasticity of demand measures the relationship between changes in quantity demanded of a good and changes in its price. It is calculated as the percentage change in quantity demanded that occurs in response to a percentage change in price. As the price of a good rises, consumers will usually demand a lower quantity of that good. The greater the extent to which quantity demanded falls as price rises, the greater is the price elasticity of demand. Some goods for which consumers cannot easily find substitutes, such as gasoline, are considered price inelastic. Note that elasticity can differ between the short term and the long term. For example, if the price of gasoline rises, consumers will eventually find ways to conserve their use of the resource. Some of these ways, like finding a more fuel-efficient car, take time. Hence gasoline would be price inelastic in the short term and more price elastic in the long term.

Elasticity of supply

Elasticity of supply measures the relationship between changes in quantity supplied of a good and changes in its price. It is measured as the percentage change in quantity supplied that occurs in response to a percentage change in price. For many goods the quantity supplied can be increased over time by locating alternative sources, investing in an expansion of production capacity, or developing competitive products that can substitute. One might therefore expect that the price elasticity of supply will be greater in the long term than the short term for such a good, that is, that supply can adjust to price changes to a greater degree over a longer period of time.

Emissions tax

An emissions tax is a charge levied on each unit of pollution emitted.

Environmental justice

Environmental justice is the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies. Fair treatment means that no group of people, including racial, ethnic, or socioeconomic groups should bear a disproportionate share of the negative environmental consequences resulting from industrial, municipal, and commercial operations or the execution of federal, state, local, and tribal programs and policies. Meaningful involvement means that: (1) people have an opportunity to participate in decisions about activities that can affect their environment and/or health; (2) the public's contribution can influence the regulatory agency's decision; (3) their concerns will be considered in the decision-making process; and (4) the decision makers seek out and facilitate the involvement of those potentially affected.[1]

1 Definition taken from http://www.epa.gov/compliance/environmentaljustice/index.html (accessed December 22, 2010).

Equity assessment

An equity assessment examines the distribution of benefits and costs associated with a regulation across specific sub-populations. Disadvantaged or vulnerable sub-populations, for example low-income households, may be of particular concern.

Expert elicitation

Expert elicitation is a formal, highly-structured and well-documented process for obtaining the judgments of multiple experts. Typically, an elicitation is conducted to evaluate uncertainty. This uncertainty could be associated with: the value of a parameter to be used in a model; the likelihood and frequency of various future events; or the relative merits of alternative models.

Externalities

An externality is a cost or benefit resulting from an action that is borne or received by parties not directly participating in the action.

Flow pollutant

A flow pollutant is a pollutant for which the environment has some absorptive capacity. It does not accumulate in the environment as long as its emission rate does not exceed the absorptive capacity of the environment. Animal and human wastes are examples of flow pollutants.

Hotspot

A hotspot is a geographic area with a high level of pollution/contamination within a larger geographic area of low or "normal" environmental quality.

Kaldor-Hicks criterion

The Kaldor-Hicks criterion is really a combination of two criteria: the Kaldor criterion and the Hicks criterion. The Kaldor criterion states that an activity will contribute to Pareto optimality if the maximum amount the gainers are prepared to pay is greater than the minimum amount that the losers are prepared to accept. Under the Hicks criterion, an activity will contribute to Pareto optimality if the maximum amount the losers are prepared to offer to the gainers in order to prevent the change is less than the minimum amount the gainers are prepared to accept as a bribe to forgo the change. In other words, the Hicks compensation test is conducted from the losers' point of view, while the Kaldor compensation test is conducted from the gainers' point of view. The Kaldor-Hicks criterion is widely applied in welfare economics and managerial economics. It forms an underlying rationale for BCA.

Latency

Latency is the time interval from the first exposure of a pollutant until the increase in health risk. See *cessation lag* for a definition of a related but distinct concept.

Leakages

A leakage is the displacement of pollution from one location to another as a result of the imposition of tighter pollution controls. Under tradable permit systems, leakages occur when pollution is displaced to an area not affected by a cap on allowed emissions.

Marginal benefit

The marginal benefit is the benefit received from an incremental increase in the consumption of a good or service. It is calculated as the increase in total benefit divided by the increase in consumption.

Marginal cost

The marginal cost is the change in total cost that results from a unit increase in output. It is calculated as the increase in total cost divided by the increase in output.

Marginal social benefit

The marginal social benefit is the marginal benefit received by the consumer of a good (marginal private benefit) plus the marginal benefit received by other members of society (external benefit).

Marginal social cost

The marginal social cost is the marginal cost incurred by the producer of a good (marginal private cost) plus the marginal cost imposed on other members of society (external cost).

Market failure

Market failure is a condition where the allocation of goods and services by a market is not efficient. Causes of market failure include: externalities, concentration of market power, information asymmetry, transactions costs, and the nature of the good (e.g., public goods). For environmental conditions, externalities are the most likely causes of the failure of private and public sector institutions to correct pollution damages.

Market permit systems

A market permit system is a system under which emissions sources are required to have emissions permits matching their actual emissions. Each permit specifies how much the source is allowed to emit and is transferable among firms.

Market-based incentives

Market-based incentives include a wide variety of methods for environmental protection. Instruments such as taxes, fees, charges, and subsidies generally "price" pollution and leave decisions about the level of emissions to each source. Another example is the market permit system, which sets the total quantity of emissions and then allows trading of permits among firms.

Meta-analysis

Meta-analysis is a statistical method of pooling data and/or results from a set of comparable studies of a problem. Pooling in this way provides a larger sample size for evaluation and allows for a stronger conclusion than can be provided by any single study. Meta-analysis yields a quantitative summary of the combined results.

Net benefits

Net benefits are calculated by subtracting total costs from total benefits.

Net future value

Net future value is similar to NPV, however, instead of discounting all future values back to the present, values are accumulated forward to some future time period — for example, to the end of the last year of a policy's effects.

Net present value (NPV)

The NPV is calculated as the present value of a stream of current and future benefits minus the present value of a stream of current and future costs.

Non-use value

Non-use value is the value that an individual may derive from a good or resource without consuming it, as opposed to the value obtained from use of the resource. Non-use values can include *bequest value*, where an individual places a value on the availability of a resource to future generations; *existence value,* where an individual values the mere knowledge of the existence of a good or resource; and *paternalistic altruism*, where an individual places a value on others' enjoyment of the resource.

Opportunity cost

Opportunity cost is the value of the next best alternative to a particular activity or resource. Opportunity cost need not be assessed in monetary terms. It can be assessed in terms of anything that is of value to the person or persons doing the assessing. For example a grove of trees used to produce paper may have a next-best-alternative use as habitat for spotted owls. Assessing opportunity costs is fundamental to assessing the true cost of any course of action. In the case where there is no explicit accounting or monetary cost (price) attached to a course of action, ignoring opportunity costs could produce the illusion that the action's benefits cost nothing at all. The unseen opportunity costs then become the implicit hidden costs of that course of action.

Quality-adjusted life year (QALY)

QALY is a composite measure used to convert different types of health effects into a common, integrated unit, incorporating both the quality and quantity of life lived in different health states. This metric is commonly used in medical arenas to make decisions about medical interventions.

Shadow price of capital

The shadow price of capital takes into account the social value of displacing private capital investments. For example, 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."

Social cost

From a regulatory standpoint, social cost represents the total burden a regulation will impose on the economy. It can be defined as the sum of all opportunity costs incurred as a result of the regulation. These opportunity costs consist of the value lost to society of all the goods and services that will not be produced and consumed if firms comply with the regulation and reallocate resources away from production activities and towards pollution abatement. To be complete, an estimate of social cost should include both the opportunity costs of current consumption that will be foregone as a result of the regulation, and also the losses that may result if the regulation reduces capital investment and thus future consumption.

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 equitable distribution of wealth, ranks better or worse than a less efficient alternative with more egalitarian distributional consequences.

Stock pollutants

A stock pollutant is a pollutant for which the environment has little or no absorptive capacity, such as non-biodegradable plastic, heavy metals such as mercury, and radioactive waste. A stock pollutant accumulates through time.

Subsidies

A subsidy is a kind of financial assistance, such as a grant, tax break, or trade barrier, that is implemented in order to encourage certain behavior. For example, the government may directly pay polluters to reduce their pollution emissions.

Tax-subsidy

A tax-subsidy is any form of subsidy where the recipients receive the benefit through the tax system, usually through the income tax, profit tax, or consumption tax systems. Examples include tax deductions for workers in certain industries, accelerated depreciation for certain industries or types of equipment, or exemption from consumption tax (sales tax or value added tax).

Total cost

Total cost is defined as the sum of all costs associated with a given activity.

Use value

Use value is an economic value based on the tangible human use of some environmental or natural resource.

Value of statistical life (VSL)

VSL is a summary measure for the dollar value of small changes in mortality risk experienced by a large number of people. VSL estimates are derived from aggregated estimates of individual values for small changes in mortality risks. For example, if 10,000 individuals are each willing to pay $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. Note that this does not mean that any single identifiable life is valued at this amount. Rather, the *aggregate value* of reducing a collection of small individual risks is, in this case, worth $5 million.

Value of statistical life year (VSLY)

The VSLY is the estimated dollar value for a year of statistical life. In practice this metric is often derived by dividing the VSL by remaining life expectancy. This approach is controversial in that it assumes that each year of life over the life cycle has the same value, and it assumes that the value of a statistical life equals the present discounted value of these annual amounts.

Willingness to accept (WTA)

WTA is the amount of compensation an individual is willing to take in exchange for giving up some good or service. In the case of an environmental policy, WTA is the least amount of money that an individual would accept to forego an environmental improvement (or endure an environmental decrement).

Willingness to pay (WTP)

WTP is the largest amount of money that an individual or group would pay to receive the benefits (or avoid the damages) resulting from a policy change, without being made worse off. In the case of an environmental policy, WTP is the maximum amount of money an individual would pay to obtain an improvement (or avoid a decrement) in an environmental effect of concern.

Chapter 1

Introduction

The *Guidelines for Preparing Economic Analyses* are part of a continuing effort by U.S. Environmental Protection Agency (EPA) to develop improved guidance on the preparation and use of sound science in support of the decision-making process. This document builds on previous work first issued in December of 1983 as the *Guidelines for Performing Regulatory Impact Analysis* (U.S. EPA 1983) and later revised in the late 1990s. In September of 2000, the EPA issued its *Guidelines for Preparing Economic Analyses* (*Guidelines*) (U.S. EPA 2000b), revised to reflect the evolution of environmental policy making and economic analysis that had accrued over the decade and a half since the original guidelines were released. At the time of release, EPA committed to periodically revise the *Guidelines* to account for further growth and development of economic tools and practices.

In an effort to fulfill that commitment, this document incorporates new literature published since the last revision of the *Guidelines*. It describes new Executive Orders (EOs) and recent guidance documents that impose new requirements on analysts, and fills information gaps by providing more expansive information on selected topics. Furthermore, a loose-leaf format has been adopted to facilitate the incorporation of new information in the future. This new, more flexible format, in addition to the electronic release of the document, will allow future updates and additions without requiring a wholesale revision of the document.

1.1 Background

While economic analysis can provide valuable insights into the setting of Agency priorities and plans for meeting them, the focus of this document is on the conduct of economic analysis to support policy decisions and meeting the requirements described by related statutes, EOs, and recommendations in guidance materials. With a few exceptions, the collection of EOs and statutes that govern the conduct of economic analysis and distributional analysis has remained largely unchanged since 2000. EO 12866, directing federal agencies to perform a benefit-cost analysis (BCA) for economically significant rules (those with an economic impact of $100 million or more), still provides the primary impetus for much of the formal BCA within the Agency.[1] However, new guidance documents and handbooks on how to comply with a number of EOs and statutes have been issued both within and outside the Agency in the last several years. The Office of Management and Budget (OMB), for instance, released its *Circular A-4* in 2003 to replace both its "Best Practices" document (OMB 1996) and its "OMB Guidelines" (OMB 2000). *Circular A-4* provides recommendations to federal agencies on the development of economic analyses supporting regulatory actions. As such, it greatly influences the conduct of economic analysis and the development of new analytic tools and approaches within the Agency. The OMB recommendations, as well as other

1 EO 13422, a 2007 amendment to EO 12866, contributed to the formal benefit-cost framework by requiring agencies to "identify in writing the specific market failure (such as externalities, market power, lack of information) or other specific problem that [the regulation] intends to address … as well as assess the significance of that problem." However, EO 13422 was revoked in January 2009 through EO 13497.

guidance documents, are referenced in the revised *Guidelines* where appropriate.

As a result of these modifications and updates the new, revised *Guidelines* will ensure that EPA's economic analyses are prepared to inform the policy-making processes and satisfy OMB's requirements for regulatory review. The new *Guidelines* also seeks 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.

Underlying these efforts is the recognition that a thorough and careful economic analysis is an important component in informing sound environmental policies. Preparing high-quality economic analysis can greatly enhance the effectiveness of environmental policy decisions by providing policy makers with the ability to systematically assess the consequences of various actions. An economic analysis can describe the implications of policy alternatives not just in terms of economic efficiency, but also in terms of the magnitude and distribution of an array of impacts. Economic analysis also serves as a mechanism for organizing information carefully. Thus, even when data are insufficient to support particular types of economic analysis, the conceptual scoping exercise can provide useful insights.

It is important to note that economic analysis is but one component in the decision-making process and under some statutes it cannot be used in setting standards. Other factors that may influence decision makers include enforceability, technical feasibility, affordability, political concerns, and ethics, to name but a few. Nevertheless, economic analysis provides a means to organize information and to comprehensively assess alternative actions and their consequences. Provided early in the regulatory design phase, economic analysis can help guide the selection of options. Ultimately, good economic analysis based on sound science should lead to better, more defensible rules.

1.2 The Scope of the *Guidelines*

The scope of the *Guidelines* is on economic analysis typically conducted for environmental policies using regulatory or non-regulatory management strategies. Separate 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.

As with the 2000 *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. To fully understand and apply the approaches and recommendations presented in the *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. Appendix A provides the reader with a brief review of economic foundations and the Glossary defines selected key terms.

These *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. The *Guidelines* should be viewed as a summary of analytical methodologies, empirical techniques, and data sources that can assist in performing economic analysis of environmental policies. When drawing upon these various resources, there is no substitute for reviewing the original source materials.

In all cases, the *Guidelines* recommends adhering to the following general principles as stated by OMB (1996):

> "'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."'

Economic analyses should always strive to be transparent by acknowledging and characterizing important uncertainties that arise. In addition, 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. Analysts must take care to avoid double counting of benefits and costs when there are overlapping regulatory initiatives. 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.

1.3 Economic Framework and Definition of Terms

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).

One of the first methodological questions an analyst must answer when conducting economic analysis is: who has "standing?" The most inclusive answer allows *all* persons who may be affected by the policy to have standing, regardless of where (or when) they live. For domestic policy making, however, the norm is to limit standing to the national level. This decision is based on the fact that authority to regulate only extends to a nation's own residents who have consented to adhere to the same set of rules and values for collective decision making, as well as the assumption that most domestic policies will have negligible effects on other countries (Kopp et al. 1997, Whittington et al. 1986).

OMB's *Circular A-4* gives the following guidance to agencies with regard to conducting economic analyses in support of rulemakings: "Analysis should focus on benefits and costs that accrue to citizens and residents of the United States. Where you choose to evaluate a regulation that is likely to have effects beyond the borders of the United States, these effects should be reported separately" (OMB 2003, p. 15). Potential regulatory alternatives are then modeled as economic changes that move the economy from a state of equilibrium absent the regulation (the baseline) to a new state of equilibrium with the regulation in effect. The differences between the old and new states are measured as changes in prices, quantities produced and consumed, income and other economic quantities. These measurements 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*. These issues often take the form of three distinct questions:

1. Is it theoretically possible for the "gainers" from the policy to fully compensate the "losers" and still remain better off?
2. Who are the gainers and losers from the policy and associated economic changes?
3. How did a particular group, especially a group considered to be disadvantaged, fare as a result of the policy change?

The first question is directed at the measurement of efficiency, and is based on the *Potential Pareto criterion*. This criterion is the foundation of BCA, requiring that a policy's net benefits to society be positive. Measuring net benefits by summing all of the welfare changes for all affected groups provides an answer to this question. Net benefits are derived by summing all of the benefits that accrue as a result of a policy change (including spillover effects) less costs imposed by the policy on society (including externalities). Since spillovers and externalities by definition are not captured in market transactions, counting private costs and private benefits accruing to market participants is not sufficient for estimating social benefits and costs. The policy that maximizes net benefits is considered the most efficient.[2]

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.

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.[3] 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. Analysts should weigh the need for additional precision against the cost of employing general equilibrium models over other methods. In doing so analysts should consider the size, impact, and complexity of the question at hand. In general, the more detailed methods are justified by questions with larger and more complex impacts. This question is considered in each of the chapters on specific models.

The *Guidelines* follows more traditional practices and adopts 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 impacts on industry, governments, and non-profit organizations using an *economic impacts analysis* (EIA); and
- the examination of effects on various sub-populations, particularly low-income, minority, and children, using *distributional analyses*.

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

2 Appendix A gives a conceptual overview of this discussion. See in particular Section A.3 on BCA.

3 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 be unable to analyze relatively small sectors of the economy. For more on general equilibrium analysis see Chapter 8, Section 4.6.

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 is comprised of a BCA, an EIA, *and* an equity assessment.

BCA evaluates the favorable effects of policy actions and the associated opportunity costs of those actions. The favorable effects are defined as benefits. Opportunities foregone define economic costs. While conceptually symmetric, benefits and costs are often evaluated separately for "traditional" environmental problems (e.g., emissions of pollutants from point sources into air and water) due to practical considerations. Analysts may organize the analysis of benefits differently from the analysis of costs, but they should be aware of the conceptual relationship between the two. Assessing the effects of environmental policy is inherently a complex process in which results from various disciplines are integrated to predict environmental outcomes and their economic consequences. As EPA addresses increasingly complex environmental problems (e.g., climate change), so in turn will be the models needed to track the various processes to describe and capture policy effects. Computable general equilibrium (CGE) models for these types of policies will become increasingly important.

Once the change in pollution levels resulting from a policy is predicted, this change is translated into health outcomes or other outcomes of interest using information provided by risk assessors. Benefits analyses then apply a variety of economic methodologies to estimate the value of these 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. More information on analyzing benefits can be found in Chapter 7 while details on estimating social costs can be found in Chapter 8.

The assumptions and modeling framework developed for the BCA can describe gains and losses to assess efficiency. However the BCA framework often limits detailed examination of the gainers and losers and the impacts on disadvantaged sub-populations. To estimate these two categories of impacts analysts rely upon EIA and equity assessments, which use a multiplicity of estimation techniques. Chapters 9 and 10 provide information on how these analyses relate to BCA and detail estimation techniques.

Note that none of these three types of analyses (BCA, EIA, and equity assessment) address the cost-effectiveness of a policy option. Cost-effectiveness analyses (CEA) report the estimated costs needed to achieve a specific goal or an additional unit of environmental improvement. Costs-per-life-saved and costs-per-ton-of-pollution-reduction are examples of cost-effectiveness measures. When comparisons are made across policies, CEA can be used to help identify the least costly approach to achieving a specific goal.[4]

1.4 Organization of the *Guidelines*

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

- Chapter 2: **Statutory and Executive Order Requirements for Conducting Economic Analyses** reviews the major statutes and other directives mandating certain economic assessments of the consequences of policy actions;

4 Note that CEA is not covered extensively in this document. Additional sources for details on CEA include IOM (2006) and Boardman et al. (2006).

- 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 the problem;
- 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: **Baselines** provides a definition of baseline and discusses how analysts should approach conducting a baseline analysis;
- 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 Costs** presents the basic theoretical approach for assessing the costs of environmental policies and describes how this can be applied in practice;
- Chapter 9: **Economic Impact Analyses and Equity Assessment** provides guidance for performing a variety of different assessments of the economic impacts of environmental policies;
- Chapter 10: **Environmental Justice, Children's Environmental Health and Other Distributional Considerations** discusses key analytical issues and considerations to keep in mind when performing distributional analyses; and
- Chapter 11: **Presentation of Analysis and Results** concludes the main body of the *Guidelines* with suggestions 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

Agencies are subject to a number of statutes and executive orders (EOs) that direct the conduct of specific types of economic analyses.[1] Many of these directives are potentially relevant for all of EPA's programs while others target individual programs. This chapter highlights directives that may apply to all of EPA's programs.[2]

The scope of requirements for economic analysis can vary substantially. In some cases, a statute or EO may contain language that limits its applicability to only those regulatory actions, or rules, that fall above a specified threshold in significance or impact. Economic analysis may be necessary to determine if a regulatory action exceeds a significance or impact threshold, and thus falls in the class of regulatory actions targeted by the statute or EO. If a regulatory action must comply with the requirements of a given statute or EO, additional economic analysis (e.g., analysis of benefits and costs as required by EO 12866), procedural steps (e.g., consultation with affected state and local governments as required by EO 13132), or a combination of economic analysis and procedural steps may be required. This chapter describes the general requirements for economic analysis contained in selected statutes and EOs, identifies thresholds beyond which a regulatory action must follow additional economic analysis requirements, and provides further direction for analysts seeking guidance on compliance with the statute or EO.[3] For each EO or statute highlighted in this chapter, references to applicable OMB and EPA guidelines are provided. Another resource for determining the type and scope of economic analysis required for a rule is a program's Office of General Counsel (OGC) attorney.[4] Requirements of the statutes and EOs that do *not* necessitate economic analysis are not covered in this chapter.

1 For the text statutes and EOs appearing in this chapter, and guidance specific to them, or for more information on their implications for EPA rule development, visit the Action Development Process (ADP) Library on EPA's intranet http://intranet.epa.gov/adplibrary (accessed April 28, 2004, internal EPA document). Many of the citations for other applicable guidelines included in this section can be found at that site. Alternatively, information on statutes and EOs can easily be found using http://usasearch.gov/.

2 Statutory provisions that require economic analysis but 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 Note that for some statutes and EOs, requirements for *proposed* regulatory actions may vary slightly from the requirements for *final* regulatory actions.

4 See U.S. EPA (2005b) for more information.

2.1 Executive Orders

2.1.1 Executive Order 12866, "Regulatory Planning and Review"

Threshold: Significant regulatory actions. A "significant regulatory action" is defined by Section 3(f)(1)-(4) as one that is likely to result in a rule that may:

- *Have an annual effect on the economy of $100 million or more or adversely affect in a material way the economy, a sector of the economy, productivity, competition, jobs, the environment, public health or safety, or State, local, or tribal governments or communities;*
- *Create a serious inconsistency or otherwise interfere with an action taken or planned by another agency;*
- *Materially alter the budgetary impact of entitlements, grants, user fees, or loan programs or the rights and obligations of recipients thereof; or*
- *Raise novel legal or policy issues arising out of legal mandates, the President's priorities, or the principles set forth in this Executive order.*

Any one of the four criteria listed above can trigger a regulatory action to be defined as "significant;" a regulatory action that meets the first criteria is generally defined as "economically significant." While the determination of economic significance is multi-faceted, it is most often triggered by the $100 million threshold. This threshold is interpreted as being *based on the annual costs or benefits of the proposed or finalized option*. If one rule option poses costs or benefits in excess of $100 million, but the rule option to be proposed or finalized has costs and benefits that fall below the $100 million range, the rule is not considered economically significant. The same definition applies whether the rule is regulatory or deregulatory in nature. In the case of a deregulatory rule with cost savings, transfers should not be netted out. For example, if there are additional costs in one market and cost savings in another, they should not be combined to get "net" cost savings. If one company loses $100 million in business to another company, that is sufficient for an economic significance determination, even if the net effect is zero. The EO is silent on whether the threshold should be adjusted for inflation. As such, nominal values have been used in practice, implying that as inflation increases the threshold becomes more stringent.

Requirements contingent on threshold: A statement of the need for the proposed action and an assessment of social benefits and costs (Section 6(a)(3)(B) are required. The requirements for BCA increase in complexity and detail for *economically* significant rules (i.e., those that fall under the definition in the first bullet above). For these rules, the EO requires that agencies conduct an assessment of benefits and costs of the action, that benefits and costs be quantified to the extent feasible, and that the benefits and costs of alternative approaches also be assessed (Section 6(a)(3)(C)).[5]

Guidance: Chapters 3 through 8 of this document provide guidance for meeting these requirements. OMB's *Circular A-4* (2003) provides guidance to federal agencies on the development of regulatory analysis of *economically* significant rules as required by EO 12866. More specifically, *Circular A-4* is intended to define good regulatory analysis and standardize the way benefits and costs of federal regulatory actions are measured and reported. Chapter 9 of this document describes methods for analyzing and assessing distributional effects of a rule through EIA. Chapter 10 addresses how to assess environmental justice implications.[6]

5 EO 13422 and amended EO 12866 formerly required analysts to "identify in writing the specific market failure (such as externalities, market power, lack of information) or other specific problem" and extended the BCA requirement to "significant" guidance documents. Although EO 13497, issued in January 2009, revoked EO 13422 together with any "orders, rules, regulations, guidelines, or policies" enforcing it, a subsequent memo issued by then Director of OMB Peter R. Orszag offering guidance on the implementation of the new EO indicated that "significant policy and guidance documents…remain subject to OIRA's review."

6 In its Statement of Regulatory Philosophy, EO 12866 states that agencies should consider the distributional and equity effects of a rule (Section 1(a)).

2.1.2 Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations"

Threshold: No specific threshold; Agencies are required to "...identify and address disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low-income populations..."

Requirements contingent on threshold: No specific analytical requirements.

Guidance: EPA issued interim guidance for considering environmental justice in the Action Development Process (U.S. EPA 2010); EPA and the Council on Environmental Quality (CEQ) have prepared guidance for addressing environmental justice concerns in the context of National Environmental Policy Act (NEPA) requirements [U.S. EPA 1998a and CEQ (1997)]. These materials provide guidance on key terms in the EO. Chapter 10 of this document addresses environmental justice analysis.

2.1.3 Executive Order 13045, "Protection of Children from Environmental Health Risks and Safety Risks"

Threshold: Economically significant regulatory actions as described by EO 12866 that involve environmental health risk or safety risk that an agency has reason to believe may disproportionately affect children.

Requirements contingent on threshold: An evaluation of the health or safety effects of the planned regulation on children, as well as an explanation of why the planned regulation is preferable to other potentially effective and reasonably feasible alternatives the agency is considering.

Guidance: EPA has prepared guidance for rule writers on compliance with EO 13045 (U.S. EPA 1998b). EPA's *Children's Health Valuation Handbook* (U.S. EPA 2003b) discusses special issues related to estimation of the value of health risk reductions to children. Guidance in Chapter 10 of this document addresses equity analyses focused on children.

2.1.4 Executive Order 13132, "Federalism"

Threshold: Rules that have "federalism implications" due to either substantial compliance costs or preemption of state or local law. Rules with federalism implications are defined as those rules "that have substantial direct effects on the States [including 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." Rules may be considered to impose substantial compliance costs on state or local governments unless the costs are expressly required by statute or there are federal funds available to cover them.

Requirements contingent on threshold: Submission to OMB of a Federalism Summary Impact Statement and consultation with elected officials of affected state and local governments.

Guidance: Specific guidance on EO 13132 can be found in the internal EPA document *Guidance on Executive Order 13132: Federalism* (U.S. EPA 2008c).[7]

2.1.5 Executive Order 13175, "Consultation and Coordination with Indian Tribal Governments"

Threshold: Rules and policy statements that have tribal implications; that is, those that have "substantial direct effects on one or more Indian tribes, on the relationship between the Federal Government and Indian tribes, or on the distribution of power and responsibilities between the Federal Government and Indian tribes."

7 This document is located at http://intranet.epa.gov/adplibrary/documents/federalismguide11-00-08.pdf (accessed March 4, 2010, internal EPA document).

Requirements contingent on threshold: To the extent practicable and permitted by law, if a regulatory action with tribal implications is proposed and imposes substantial direct compliance costs on Indian tribal governments, and is not required by statute, then the agency must either provide the funds necessary to pay the tribal governments' direct compliance costs, or consult with tribal officials early in the process of regulatory development and provide to OMB a Tribal Summary Impact Statement.

Guidance: A tribal guidance document is currently under development by EPA's Regulatory Management Division.[8] Guidance in Chapter 9 of this document addresses equity analyses focusing on minority populations.

2.1.6 Executive Order 13211, "Actions Concerning Regulations that Significantly Affect Energy Supply, Distribution, or Use"

Threshold: Rules that are a significant regulatory action under EO 12866 and that are likely to have significant adverse effects on the supply, distribution, or use of energy.

Requirements contingent on threshold: Submission of a Statement of Energy Effects to OMB. The Statement of Energy Effects addresses the magnitude of expected adverse effects, describes reasonable alternatives to the action, and describes the expected effects of such alternatives on energy supply, distribution, and use.

Guidance: EPA has prepared guidance on what effects might be considered significant in *Memorandum on Energy Executive Order 13211 — Preliminary Guidance (2008d)*. OMB has guidance for implementing EO 13211 as well.[9]

8 Please check the ADP Library on EPA's intranet, http://intranet.epa.gov/adplibrary (accessed April 8, 2010, internal EPA document) for the status of this guidance.

9 U.S. EPA 2008d, *Memorandum on Energy Executive Order 13211 — Preliminary Guidance*, located at http://intranet.epa.gov/adplibrary/statutes.htm#energy under the heading "Preamble Template" (accessed July 8, 2008, internal EPA document). OMB's guidance for implementing EO 13211 is located at http://www.whitehouse.gov/omb/memoranda/m01_27.html (accessed July 8, 2008).

2.2 Statutes

2.2.1 The Regulatory Flexibility Act of 1980 (RFA), as Amended by The Small Business Regulatory Enforcement Fairness Act of 1996 (SBREFA) (5 U.S.C. 601-612)

Threshold: Regulations that have a significant economic impact on a substantial number of small entities, including small businesses, governments and non-profit organizations.

Requirements contingent on threshold: Preparation of a regulatory flexibility analysis, and compliance with a number of procedural requirements to solicit and consider flexible regulatory options that minimize adverse economic impacts on small entities.

Guidance: EPA has issued specific guidance for complying with RFA/SBREFA requirements in the internal document *EPA Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act* (2006c).[10]

2.2.2 The Unfunded Mandates Reform Act of 1995 (UMRA) (P.L. 104-4)

Threshold one (Sections 202 and 205 of UMRA): Regulatory actions that include federal mandates "that may result in the expenditure by State, local, and tribal governments, in the aggregate, or by the private sector, of $100 million or more (adjusted annually for inflation) in any one year."[11]

Requirements contingent on threshold one: Section 202 of UMRA requires preparation of a written statement that includes the legal authority for the action; a BCA; a distributional analysis; estimates of macroeconomic impacts; and a description of an agency's consultation with elected representatives of the affected state, local, or tribal governments. Section 205 of UMRA

10 U.S. EPA 2006c, available at http://intranet.epa.gov/adplibrary (accessed May 1, 2008, internal EPA document).

11 Note that the threshold in this case is "adjusted annually for inflation" as opposed to the threshold under EO 12866.

requires an agency to consider a reasonable number of regulatory alternatives and select the least costly, most cost-effective, or least burdensome alternative, or to publish with the final rule an explanation of why such alternative was not chosen.

Threshold two (Section 203 of UMRA): Regulatory requirements that might "significantly" or "uniquely" affect small governments.

Requirements contingent on threshold two: Agencies must solicit involvement from, and conduct outreach to, potentially affected small governments during development and implementation.

Guidance: EPA has issued *Interim Guidance on the Unfunded Mandates Reform Act of 1995,* (1995b), and OMB provides general guidance on complying with requirements contingent on each of the two thresholds under UMRA.[12]

2.2.3 The Paperwork Reduction Act of 1995 (PRA) (44 U.S.C. 3501)

Threshold: Actions (both regulatory and non-regulatory) that include record-keeping, reporting, or disclosure requirements or other information collection activities calling for answers to identical questions imposed upon or posed to ten or more persons, other than federal agency employees.

Requirements contingent on threshold: The agency must submit an information collection request (ICR) to OMB for review and approval and meet other procedural requirements including public notice. Note that 1320.3(c)(4)(ii) states that "any collection of information addressed to all or a substantial majority of an industry is presumed to involve ten or more persons." However, OMB guidance on this issue indicates that if agencies have evidence showing that this presumption is incorrect in a specific situation (i.e., fewer than 10 persons would be surveyed), the agency may proceed with the collection without seeking OMB approval. Agencies must be prepared to provide this evidence to OMB on request and abide by OMB's determination as to whether the collection of information ultimately requires OMB approval.

Guidance: Both guidance and templates for completing an ICR and associated Federal Register (FR) notices can be found on EPA's intranet site, "ICR Center."[13]

12 See U.S. EPA 1995b available at http://intranet.epa.gov/adplibrary/statutes/umra.htm (accessed December 21, 2010).

13 See http://intranet.epa.gov/icrintra/ (accessed April 14, 2004, internal EPA document).

Chapter 3

Statement of Need for Policy Action

A clear *statement of need for policy action* is an essential component in economic analyses of environmental policy prepared for economically significant rules.[1] This chapter discusses the key elements that comprise this statement:

- Problem Definition: Section 3.1 provides components to include in a definition of the environmental problem to be addressed;
- Reasons for Market or Institutional Failure: Section 3.2 identifies factors relevant to an analysis of the reasons existing legal and other institutions have failed to correct the problem; and
- Need for Federal Action: Section 3.3 describes items to consider in preparing a justification of the need for federal intervention instead of other alternatives.

The statement of need for policy action should also describe any statutory or judicial requirements that mandate the promulgation of particular policies or the evaluation of specific effects pertaining to the action. In some instances, statutes prohibit the use of certain types of analysis in policy making. In these cases, the guidance presented in *Guidelines* should be applied in a manner consistent with such mandates.

3.1 Problem Definition

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

- The primary pollutants causing the problem and their concentration;
- The 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 resulting outcome;
- Expected evolution of the environmental problem over the time horizon of the analysis;
- Current control and mitigation techniques;
- The amount or proportion (or both) of the environmental problem likely to be corrected by federal action.

3.2 Reasons for Market or Institutional Failure

After defining the problem, the statement of need should examine the reasons why the market and other public and private sector institutions have failed to correct it. This identification is an important component of policy development because the underlying failure itself often suggests the most appropriate remedy for the problem.

1 EO 12866 states that "Federal agencies should promulgate only such regulations as are required by law, *are necessary to interpret the law, or are made necessary by compelling need, such as material failures of private markets to protect or improve the health and safety of the public, the environment, or the well-being of the American people…"* (emphasis added). EO 13422 extended the requirements in EO 12866 to guidance documents, but has since been revoked.

OMB's *Circular A-4* discusses three categories of market failure, including externalities, market power, and inadequate or asymmetric information.[2] *Circular A-4* also points out that there may be other social purposes for regulation beyond correcting market failures, such as improving government function, removing distributional unfairness, or promoting privacy and personal freedom. Externalities are the most likely cause of the failure of private and public sector institutions to completely correct environmental damages. However, information asymmetries and pre-existing government-induced distortions can also be responsible for these problems.

Externalities occur when the market does not compensate for the effect of one party's activities on another party's well-being. Externalities can occur for many reasons, for example, high transaction costs can make it difficult for injured parties to ensure that polluters internalize the cost of damage through bargaining, legal, or other means. Externalities can also result when activities that pose environmental risks are difficult to link to the resulting damages, such as those that occur over long periods of time or those that are transferred from one location to another.

Consistent with EO 12866, the statement of need should assess the significance of the problem. Economic analyses should explore, for example, why transaction costs are high or what information asymmetries exist. Similar analyses are appropriate for situations where other factors are responsible for the failure of the market or public and private sector institutions to adequately address an environmental problem.

3.3 Need for Federal Action

The final component of the statement of need for policy action is an analysis of why a federal remedy is preferable to actions by private and other public sector entities, such as the judicial system or state and local governments.[3] Federal involvement is often required for environmental problems that cross jurisdictional boundaries (for instance, international environmental problems). In some cases, federal involvement is mandated by statute or EO as described in Chapter 2. This analysis should justify the basis for federal involvement by comparing it to the performance of a variety of realistic alternatives that rely on other institutional arrangements. This component of the statement of need for policy action should verify that the proposed action is within the jurisdiction of the relevant statutory authorities, and that the results of the policy will be preferable to no action. Finally, the statement of need should identify any aspects of the regulations being proposed that are necessitated by statutory requirements rather than being discretionary, as this may have an influence on the development of the economic analysis and presentation of the results.

2 For further discussion of market failure, see Perman et al. (2003), Hanley et al. (2001), and Nicholson (1995).

3 See EO 13132 on "Federalism" for introductory statements regarding principles of federalism, and a section describing the special requirements for preemption.

Chapter 4

Regulatory and Non-Regulatory Approaches to Pollution Control

This chapter briefly describes several regulatory and non-regulatory approaches used in environmental policy making. The goals of this chapter are to introduce several important analytic terms, concepts, and approaches; to describe the conceptual foundations of each approach; and to provide additional references for those interested in a more in-depth discussion.[1] Specifically, this chapter discusses the following four general approaches to environmental policy making: (1) command-and-control regulation; (2) market-based incentives; (3) hybrid approaches; and (4) voluntary initiatives. While command-and-control regulations have been a commonly used method of environmental regulation in the United States, EPA also employs the three other approaches. Market-based incentives and hybrid approaches offer the regulated community an opportunity to meet standards with increased flexibility and lower costs compared to many command-and-control regulations, while voluntary initiatives may allow environmental improvements in areas not traditionally regulated by EPA. The chapter also includes a discussion of criteria used to evaluate the effectiveness of regulatory and non-regulatory approaches to pollution control.

4.1 Evaluating Environmental Policy

Once federal action is deemed necessary to address an environmental problem, policy makers have a number of options at their disposal to influence pollution levels. In deciding which approach to implement, policy makers must be cognizant of constraints and limitations of each approach in addressing specific environmental problems. It is important to account for how political and information constraints, imperfect competition, or pre-existing market distortions interact with various policy options. Even when a particular approach is appealing from a social welfare perspective, it may not be consistent with statutory requirements, or may generate additional concerns when considered along with other existing regulations. While any policy option under consideration must balance cost considerations with other important policy goals (including benefits), economic efficiency and cost-effectiveness are two economic concepts useful for framing the discussion and comparison of the regulatory options presented in the remaining sections of this chapter.

4.1.1 Economic Efficiency

Economic efficiency can be defined as the maximization of social welfare. An efficient market is one that allows society to maximize the net present value (NPV) of benefits: the difference between a stream of social benefits and social costs over time. The efficient level of production is referred to as *Pareto optimal* because there is no way to rearrange production or reallocate goods in such a way that someone is better off without making someone else worse off in the process. The efficient

1 Baumol and Oates (1988), particularly Chapters 10-14; Kahn (1998); Kolstad (2000); Sterner (2003); and Field and Field (2005) are useful references on the economic foundations of many of the approaches presented here.

level of production occurs without government intervention in a market characterized by no market failures or externalities (see Appendix A for a more detailed discussion of efficiency and for a graphical representation of the efficient point of production). Government intervention may be justified, however, when a market failure or externality exists (see Appendix A), in which case the government may attempt to determine the socially optimal point of production once such externalities have been internalized. Said differently, government analysts may evaluate which of the various policy approaches under consideration maximizes the benefits of reducing environmental damages, net the resulting abatement costs.

Conceptually, the socially optimal level is determined by reducing emissions until the benefit of abating one more unit of pollution (i.e., the marginal abatement benefit) — measured as a reduction in damages — is equal to the cost of abating one additional unit (i.e., the marginal abatement cost).[2] In the simplest case, when each polluter chooses the level at which to emit according to this decision rule (i.e., produce at a level at which the marginal abatement benefit is equal to the marginal abatement cost), an efficient aggregate level of emissions is achieved when the cost of abating one more unit of pollution is equal across all polluters. Any other level of emissions would result in a reduction in net benefits.

This definition of efficiency describes the simplest possible world where a pollutant is a uniformly mixed flow pollutant — the pollutant does not accumulate or vary over time — and the marginal damages that result are independent of location. When pollution levels and damages vary by location, the efficient level of pollution is achieved when marginal abatement costs adjusted by individual transfer coefficients are equal across all polluters. Temporal variability also implies an adjustment to this equilibrium condition. In the case of a stock pollutant, marginal abatement costs are equal across the discounted sum of damages from today's emissions in all future time periods. In the case of a flow pollutant, this condition should be adjusted to reflect seasonal or daily variations. Under uncertainty, it is useful to think of the efficient level of pollution as a distribution instead of as a single point estimate.

The reality of environmental decision making is that Agency analysts are rarely in the position to select the economically efficient point of production when designing policy. This is partly because the level of abatement required to reduce a particular environmental problem is often determined legislatively, while the implementation of the policy to achieve such a goal is left to the Agency. In cases where the Agency has some say in the stringency of a policy, its degree of flexibility in determining the approach taken varies by statute. This may limit its ability to consider particular approaches or to use particular policy instruments. It is also important to keep in mind analytic constraints. In cases where it is particularly difficult to quantify benefits, cost-effectiveness may be the most defensible analytic framework.

4.1.2 Cost-Effectiveness

The efficiency of a policy option differs from its cost-effectiveness. A policy is cost-effective if it meets a given goal at least cost, but cost-effectiveness does not encompass an evaluation of whether that goal has been set appropriately to maximize social welfare. All efficient policies are cost-effective, but it is not necessarily true that all cost-effective policies are efficient. A policy is considered cost-effective when marginal abatement costs are equal across all polluters. In other words, for any level of total abatement, each polluter has the same cost for their last unit abated.

4.2 Traditional Command-and-Control or Prescriptive Regulation

Many environmental regulations in the United States are prescriptive in nature (and are often

2 The idea that a given level of abatement is efficient — as opposed to abating until pollution is equal to zero — is based on the economic concept of diminishing returns. For each additional unit of abatement, marginal social benefits decrease while marginal social costs of that abatement increase. Thus, it only makes sense to continue to increase abatement until the point where marginal benefits and marginal costs are just equal. Any abatement beyond that point will incur more additional costs than benefits.

referred to as command-and-control regulations).[3] A prescriptive regulation can be defined as a policy that prescribes how much pollution an individual source or plant is allowed to emit and/or what types of control equipment it must use to meet such requirements. Such a standard is often defined in terms of a source-level emissions rate. Despite the introduction of potentially more cost-effective methods for regulating emissions, this type of regulation is still commonly used and is sometimes statutorily required. It is almost always available as a "backstop" if other approaches do not achieve desired pollution limits.

Because a prescriptive standard is commonly defined in terms of an emissions *rate*, it does not directly control the aggregate emission *level*. In such cases, aggregate emissions will depend on the number of polluters and the output of each polluter. As either production or market size increase, so will aggregate emissions. Even when the standard is defined in terms of an emission level per polluting source, aggregate emissions will still be a function of the total number of polluters.

When abatement costs are similar across regulated sources, a source-level standard may be reasonably cost-effective. However when abatement costs vary substantially across polluters, reallocating abatement activities so that some polluters have stricter standards than others could lead to substantial cost savings. If reallocation were possible (e.g., through a non-prescriptive approach), a polluter facing relatively high abatement costs would continue to emit at its current level but would pay for the damages incurred (e.g., by paying a tax or purchasing permits), while a polluter with relatively low abatement costs would reduce its emissions.

Note that regulators can at least partially account for some variability in costs by allowing prescriptive standards to vary according to size of the polluting entity, production processes, geographic location, or other factors. Beyond this, however, a prescriptive standard usually does not allow for reallocation of abatement activities to take place — each entity is still expected to achieve a specified emissions standard. Thus, while pollution may be reduced to the desired level, it is often accomplished at a higher cost under a prescriptive approach.[4]

It is common to "grandfather," or exempt, older polluters from new prescriptive regulations, thereby subjecting them to a less stringent standard than newer polluters. Grandfathering creates a bias against constructing new facilities and investing in new pollution control technology or production processes.[5] As a result, grandfathered older facilities with higher emission rates tend to remain active longer than they would if the same emissions standard applied to all polluters.

The most stringent form of prescriptive regulation is one in which the standard specifies zero allowable source-level emissions. For instance, EPA has completely banned or phased out the use or production of chlorofluorocarbons (CFCs) and certain pesticides. This approach to regulation is potentially useful in cases where the level of pollution that maximizes social welfare is at or near zero.[6]

Two types of prescriptive regulations exist: technology or design standards; and performance-based standards.

4.2.1 Technology or Design Standards

A *technology* or *design standard*, mandates the specific control technologies or production

3 Goulder and Parry (2008) refer to these as "direct regulatory instruments" because they feel that "command-and-control" has a "somewhat negative connotation." Ellerman (2003) refers to them as prescriptive regulations. We follow that convention here. Notable exceptions to this type of regulation in the U.S. experience include the phase-down in lead content in gasoline, which allowed trading of credits among refineries and offset programs applied in non-attainment areas. For more information on early applications of market incentives, see U.S. EPA (2001b).

4 See Tietenberg (2004) for a discussion of empirical studies that examine the cost-effectiveness of prescriptive air pollution regulations. Of the ten studies included, eight found that prescriptive regulations cost at least 78 percent more than the most cost-effective strategy.

5 For a discussion of grandfathering, see Helfand (1991).

6 For cases where the optimal level of pollution is at or near zero, the literature also indicates that market-based incentives can sometimes be useful as a transition instrument for the phasing-out of a particular chemical or pollutant. See Sterner (2003) and Kahn (1998).

Text Box 4.1 - Coase Solution

Government intervention for the control of environmental externalities is only necessary when parties cannot work out an agreement between themselves. Coase (1960) outlined conditions under which a private agreement between affected parties might result in the attainment of a social welfare maximizing level of pollution without government intervention. First, property rights must be clearly defined. In situations where the resource in question is not "owned" by anyone, there are no incentives to negotiate, and the offending party can "free ride," or continue to pollute, without facing the costs of its behavior.

When property rights have been allocated, a social welfare maximizing solution can be reached regardless of which party is assigned the property rights, although the equity of the assignment may vary. Take for example a farm whose pesticide application to its crops contributes pollution to the well water of nearby homeowners. If property rights of the watershed are assigned to the homeowners, then the farm may negotiate with the homeowners to allow it to continue to use the pesticide. The payment need not be in the form of cash but could be payments in kind. If property rights of the watershed are given to the farm, then the homeowners would have to pay the farm to stop applying the pesticide.

In each case, the effectiveness of the agreement is contingent on meeting additional conditions: bargaining must be possible, and transaction costs must be low. These conditions are more likely to be met when there are only a small number of individuals involved. If either party is unwilling to negotiate or faces high transaction costs, then no private agreement will be reached. Asymmetric information can also hinder the ability of one or more party to come to an agreement. Going back to the example, consider a case where there are many farms in the watershed using the pesticide on their crops. Clearly homeowners would have more difficulty in negotiating an agreement with every farm than they would when negotiating with one farm.

processes that an individual pollution source must use to meet the emissions standard. This type of standard constrains plant behavior by mandating how a source must meet the standard, regardless of whether such an action is cost-effective. Technology standards may be particularly useful in cases where the costs of emissions monitoring are high but determining whether a particular technology or production process has been put in place to meet a standard is relatively easy. However, since these types of standards specify the abatement technology required to reduce emissions, sources do not have an incentive to invest in more cost-effective methods of abatement or to explore new and innovative abatement strategies or production processes that are not permitted by regulation.

4.2.2 Performance-based Standards

A *performance-based standard* also requires that polluters meet a source-level emissions standard, but allows a polluter to choose among available methods to comply with the standard. At times, the available methods are constrained by additional criteria specified in a regulation. Performance-based standards that are technology-based do not specify a particular technology, but rather consider what is possible for available and affordable technology to achieve when establishing a limit on emissions.[7]

In the case of a performance-based standard, the level of flexibility a source has in meeting the standard depends on whether the standard specifies an emission *level* or an emission *rate* (i.e., emissions per unit of output or input). A standard that specifies an emission level allows a source to

7 As an example, Reasonably Available Control Technology (RACT) specifies that the technology used to meet the standard should achieve "the lowest emission limit that a particular source or source category is capable of meeting by application of control technology that is reasonably available considering technological and economic feasibility." RACT defines the standard on a case-by-case basis, taking into account a variety of facility-specific costs and impacts on air quality. EPA has been restrictive in its definition of technologies meeting this requirement and eliminates those that are not commercially available (see Swift 2000).

choose to implement an appropriate technology, change its input mix, or reduce output to meet the standard. An emission rate, on the other hand, may be more restrictive depending on how it is defined. If the emissions rate is defined per unit of output, then it does not allow a source to meet the standard through a reduction in output. If the standard is defined as an average emissions rate over a number of days, then the source may still reduce output to meet the standard.

The flexibility of performance-based standards encourages firms to innovate to the extent that they allow firms to explore cheaper ways to meet the standard; however, they generally do not provide incentives for firms to reduce pollution beyond what is required to reach compliance.[8] For emissions that fall below the amount allowed under the standard, the firm faces a zero marginal abatement cost since the firm is already in compliance. Also, because permitting authority is often delegated to the States, approval of a technology in one state does not ensure its use is allowed in another. Firm investment in research to develop new, less expensive, and potentially superior technologies is therefore discouraged.[9]

4.3 Market-Oriented Approaches

Market-oriented approaches (or market-based approaches) create an incentive for the private sector to incorporate pollution abatement into production or consumption decisions and to innovate in such a way as to continually search for the least costly method of abatement.[10] Market-oriented approaches can differ from more traditional regulatory methods in terms of economic efficiency (or cost-effectiveness) and the distribution of benefits and costs. In particular, many market-based approaches minimize polluters' abatement costs, an objective that often is not achieved under command-and-control based approaches. Because market-based approaches do not mandate that each polluter meet a given emissions standard, they typically allow firms more flexibility than more traditional regulations and capitalize on the heterogeneity of abatement costs across polluters to reduce aggregate pollution efficiently. Environmental economists generally favor market-based policies because they tend to be least costly, they place lower information burden on the regulator, and they provide incentives for technological advances. Four classic market-based approaches are discussed in this section:

- Marketable permit systems;
- Emission taxes;
- Environmental subsidies; and
- Tax-subsidy combinations. [11]

While operationally different (e.g., taxes and subsidies are price-based while marketable permits are quantity-based), these market-based instruments are more or less functionally equivalent in terms of the incentives they put in place. This is particularly true of emission taxes and cap-and-trade systems, which can be designed to achieve the same goal at equivalent cost. The sections that follow discuss each of these market-based approaches in turn.

4.3.1 Marketable Permit Systems

Several forms of emissions trading exist, including cap-and-trade systems, project-based trading

8 For a theoretical analysis of incentives for technological change, see Jung et al. (1996) and Montero (2002). Empirical analyses can be found in Jaffe and Stavins (1995), and Kerr and Newell (2003).

9 See Swift (2000) and U.S. EPA (1991) for a detailed discussion of how emission rate-based standards hinder technological innovation.

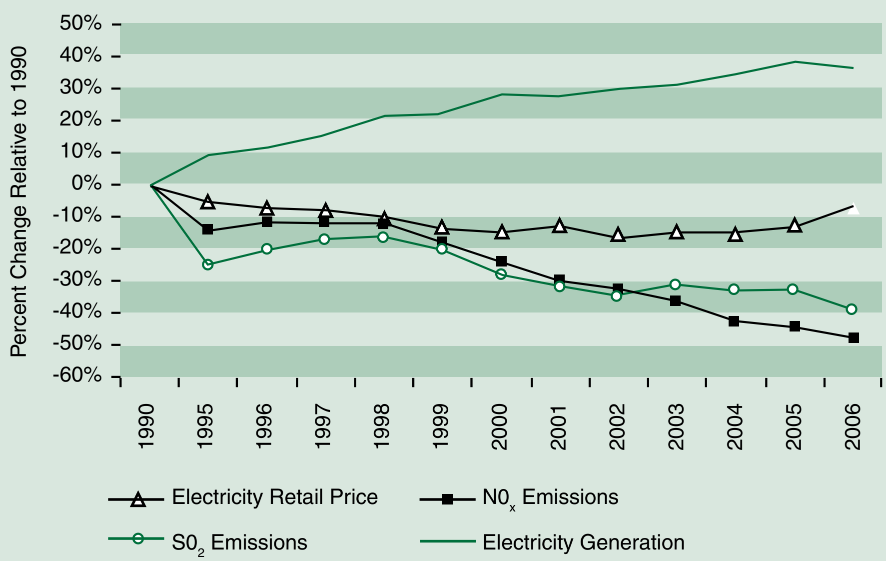
10 The incentive to innovate means that the marginal abatement cost curve shifts downward over time as cheaper abatement options are introduced.

11 The literature on applied market-based approaches for environmental protection should be consulted, along with the references they contain, for information concerning the design, operation, and performance of these approaches. Anderson and Lohof (1997) and Stavins (1998a, 2000b) compile information on both the theory and empirical use of economic incentives. Newell and Stavins (2003) generate rules-of-thumb designed to make it easy for policy makers to determine when market-based incentives may result in cost savings over command-and-control regulations. Harrington et al. (2004) compare the costs and outcomes of command-and-control and incentives-based regulatory approaches to the same environmental problem in the United States and Europe. Additional sources include Sterner (2003), Stavins (2003), Tietenberg (1999, 2002), U.S. EPA (2004a, 2001a), OECD (1994a, 1994b), and proceedings published under the "Project 88" forum, Stavins (1988, 1991).

Text Box 4.2 - Acid Rain Trading Program for Sulfur Dioxide (SO_2)

In 1995, Title IV of the 1990 Clean Air Act Amendments established a cap-and-trade system for SO_2 emissions to address the problem of acid rain. Two hundred and sixty three of the highest emitting SO_2 units of 110 electricity-generating plants were selected to participate in the first phase of the trading program. Emissions of SO_2 in 1995 were initially limited to 8.7 million tons for those facilities. Of the plants that participated, most were coal-fired units located east of the Mississippi River. Under this system, allowances were allocated to units on a historical basis, after which they could use the allowances, sell them to other units, or "bank" the allowances for use in subsequent years. Continual emission monitoring (CEM) systems have allowed the government to easily monitor and enforce emission restrictions in accordance with the allowances. The second phase of the program, initiated in 2000, imposed a national SO_2 emissions cap of 10 million tons and brought almost all SO_2 generating units into the system.

Initial evaluations of the first phase of implementation suggest that the SO_2 trading system has significantly reduced emissions at a relatively low cost. In fact, allowance prices have been considerably lower than predicted, reflecting lower than expected marginal costs. A significant level of trading has occurred and has resulted in savings of over $1 billion per year as compared to command-and-control alternatives. Emissions in 1995 were almost 40 percent below the 10 million ton limit. The evaluations demonstrated that one reason for such large reductions in SO_2 emissions below the allowable limit is the ability to bank allowances for future use. The success of the program has continued into the second phase, with recent estimates of the full U.S. Acid Rain Program's benefits [including SO_2 trading and direct nitrogen oxide (NO_x) controls] reaching upwards of $120 billion annually in 2010 with annual costs around $3 billion (in 2000$); a benefit to cost ratio of about 40 to 1. Trends over the life of the program show that while electricity generation has grown steadily and SO_2 and NO_x emissions have fallen substantially, electricity retail prices, until very recently, have declined in real terms.



Source: U.S. EPA 2007a

For more information, see Burtraw and Bohi (1997), Schmalensee et al. (1998), Stavins (1998b, 2003), Carlson et al. (2000), Chestnut and Mills (2005), and U.S. EPA (2007a).

systems and emissions rate trading systems. The common element across these programs is that sources are able to trade credits or allowances so that those with opportunities to reduce emissions at lower costs have an incentive to do so. Each of these systems is discussed in turn below.[12]

12 For a more detailed discussion of the various systems and how to design them, see U.S. EPA (2003c).

4.3.1.1 Cap-and-Trade Systems

In a cap-and-trade system the government sets the level of aggregate emissions, emission allowances are distributed to polluters, and a market is established in which allowances may be bought or sold. The price of emission allowances is allowed to vary. Because different polluters incur different private abatement costs to control emissions, they are willing to pay different amounts for allowances. Therefore, a cap-and-trade system allows polluters who face high marginal abatement costs to purchase allowances from polluters with low marginal abatement costs, instead of installing expensive pollution control equipment or using more costly inputs. Cap-and-trade systems also differ from command-and-control regulations in that they aim to limit the aggregate emission level over a compliance period rather than establish an emissions rate.

If the cap is set appropriately, then the equilibrium price of allowances, in theory, adjusts so that it equals the marginal external damages from a unit of pollution. This equivalency implies that any externality associated with emissions is completely internalized by the firm. For polluters with marginal abatement costs greater than the allowance price, the cheapest option is to purchase additional units and continue to emit. For polluters with marginal abatement costs less than the allowance price, the cheapest option is to reduce emissions and sell their permits. As long as the price of allowances differs from individual firms' marginal abatement costs, firms will continue to buy or sell them. Trading will occur until marginal abatement costs equalize across all firms.[13]

Generally, allowances initially sold at auction represent income transfers from the purchasers to the government in the amount of the price paid for the allowances. The collection of revenue through this method of allowance allocation gives the government the opportunity to reduce pre-existing market inefficiencies, to reduce distributional consequences of the policy, or to invest in other social priorities. Allowances may also be allocated to polluters according to a specified rule. This represents a transfer from the government to polluting firms, some of which may find that the value of allowances received exceeds the firm's aggregate abatement costs.

The distribution of rents under cap-and-trade systems should be considered when comparing these systems with more traditional regulatory approaches. If the allowances are auctioned or otherwise sold to polluters, the distributional consequences will be similar to those experienced when regulating using taxes. If allowances are distributed for free to polluters, however, distributional consequences will depend on the allocation mechanism (e.g., historical output or inputs), on who receives the allowances, and on the ability of the recipients to pass their opportunity costs on to their customers. If new entrants must obtain allowances from existing polluters, then the policy maker should also consider potential barrier-to-entry effects. Differing treatment applied to new versus existing polluters can affect the eventual distribution of revenues, expenses, and rents within the economy.

Additional considerations in designing an effective cap-and-trade system include "thin" markets, transaction costs, banking, effective monitoring, and predictable consequences for noncompliance. The United States' experience suggests that a market characterized by low transaction costs and being "thick" with buyers and sellers is critical if pollution is to be reduced at the lowest cost. This is because small numbers of potential traders in a market make competitive behavior unlikely, and fewer trading opportunities result in lower cost savings. Likewise, the number of trades that occur could be significantly hindered by burdensome requirements that increase the transaction costs associated with each trade.[14]

13 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 economic analyses of this program see Joskow et al. (1998), Stavins (1998b), Ellerman et al. (2000), and Chestnut and Mills (2005). For more information on the program itself see Text box 4.2 and EPA's (2008a) Acid Rain Website at http://www.epa.gov/acidrain (accessed April 5, 2004).

14 This is also often the case for bubbles and offsets. See O'Neil (1983) for an evaluation of an early example of a permit-trading program in the United States and the main reasons for its failure.

Cap-and-trade systems should also be sensitive to concerns about potential temporal or spatial spikes (i.e., hotspots — areas in which the pollution level has the potential to increase as a result of allowance trading). This may happen, for example, in an area in which two facilities emit the same amount of pollution, but due to differences in exact location and site characteristics, one facility's impact on environmental quality differs substantially from that of the other polluter. While one potential solution to this problem is to adjust trading ratios to equalize the impact of particular polluters on overall environmental quality, determining the appropriate adjustments to these ratios can be costly and difficult. Other possible solutions include zone-based trading and establishing pollution "floors."

Two recent reviews of the literature (Burtraw et al. 2005 and Harrington et al. 2004) find little evidence of spatial or temporal spikes in pollution resulting from the use of market-based approaches. In fact, market-based approaches have led to smoothing of emissions across space in some cases. These results come primarily from studies of the SO_2 and NO_x trading programs and if the market-based policy is not carefully designed, the results may not transfer to other pollutants that have more localized effects.

Banking introduces increased flexibility into a trading system by allowing polluters to bank unused permits for future use. A firm may reduce emissions below the allowance level now, and bank (or save) remaining allowances to cover excess emissions or sell to another polluter at a later time. In this way, polluters that face greater uncertainty regarding future emissions, or that expect increased regulatory stringency, can bank allowances to offset potentially higher future marginal abatement costs.

For a cap-and-trade system to be effective, reliable measurement and monitoring of emissions must occur with predictable consequences for noncompliance. At the end of the compliance period, emissions at each source are compared to the allowances held by that source. If a source is found to have fewer allowances than the monitored emission levels, it is in noncompliance and the source must provide allowances to cover its environmental obligation. In addition, the source must pay a penalty automatically levied per each ton of excess emissions.[15]

4.3.1.2 Project-Based Trading Systems

Offsets and bubbles (sometimes known as "project-based" trading systems) allow restricted forms of emissions trading across or within sources to allow sources greater flexibility in complying with command-and-control regulations such as emission limits or facility-level permits. An offset allows a new polluter to negotiate with an existing source to secure a reduction in the latter's emissions. A bubble allows a facility to consider all sources of emissions of a particular pollutant within the facility to achieve an overall target level of emissions or environmental improvement. While offsets and bubbles are mostly used to control air pollution in non-attainment areas, they have been historically hindered by high administrative and transaction costs because they require case-by-case negotiation to convert a technology or emission rate limit into tradable emissions per unit of time, to establish a baseline, and to determine the amount of credits generated or required (U.S. EPA 2001a).

4.3.1.3 Rate-Based Trading Systems

Rather than establish an emissions cap, the regulatory authority under a rate-based trading program, establishes a performance standard or emissions rate. Sources with emission rates below the performance standard can earn credits and sell them to sources with emission rates above the standard. As with the other trading systems, sources able to improve their emissions rate at low cost have an incentive to do so since they can sell the resulting credits to those sources facing higher costs of abatement. However, emissions may increase under these programs if sources increase their utilization or if new sources enter the market. Therefore, the regulating authority

15 Notably, the U.S. Acid Rain Trading Program has nearly 100 percent compliance and requires only about 50 EPA staff to administer.

may need to periodically impose new rate standards to achieve and maintain the desired emission target, which in turn may lead to uncertainty in the long term for the regulated sources. Rate-based trading programs have been used in the United States to phase out lead in gasoline (1985) and to control mobile source emissions (U.S. EPA 2003c).

4.3.2 Emissions Tax

Emissions taxes are exacted per unit of pollution emitted and induce a polluter to take into account the external cost of its emissions. Under an emissions tax, the polluter will abate emissions up to the point where the additional cost of abating one more unit of pollution is equal to the tax, and the tax will result in an efficient outcome if it is set equal to the additional external damage caused by the last unit of pollution emitted.

As an example of how an emissions tax works, suppose that emissions of a toxic substance are subject to an environmental charge based on the damages the emissions cause. To avoid the emissions tax, polluters find the cheapest way to reduce pollution. This may involve a reduction in output, a change in inputs to production, the installation of pollution control equipment, or a process change that prevents the creation of pollution. Polluters decide individually how much to control their emissions, based on the costs of control and the magnitude of the tax. The polluting firm reduces emissions to the point where the cost of reducing one more unit of emissions is just equal to the tax per unit of emissions. For any remaining emissions, the polluter prefers to pay the tax rather than to abate further. In addition, the government earns revenue that it may use to reduce other pollution or reduce other taxes, or may redistribute to finance other public services.[16] While difficult to implement in cases where there is temporal and/or spatial variation in emissions, policy makers can more closely approximate the ambient impact of emissions by incorporating adjustment factors for seasonal or daily fluctuations or individual transfer coefficients in the tax.

Despite the apparent usefulness of such a tax, true emissions taxes — those set equal or close to marginal external damages — are relatively rare in the United States.[17] This is because taxing emissions directly may not be feasible when emissions are difficult to measure or accurately estimate, when it is difficult to define and monetarily value marginal damages from a unit of emissions (which is needed to properly set the tax), or when taxes are applied to emissions that are difficult to monitor and/or enforce. In addition, attempts to measure and tax emissions may lead to illegal dumping.[18] Other considerations when contemplating the use of emission taxes include the potential imposition of substantially different cost burdens on polluters as compared with other regulatory approaches, political incentives to set the tax too low, and the collection of revenues and distribution of economic rents that result from such programs.

User or product charges are a variation on emission taxes that are occasionally utilized in the United States. These charges may be imposed directly upon users of publicly operated facilities or upon intermediate or final products whose use or disposal harms the environment. User or product charges may be effective approximations of an emissions tax for those cases in which the product taxed is closely related to emissions. User 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. Product charges have been imposed on products that release CFCs into the atmosphere, that utilize more gasoline (such as cars), or require more fertilizer. In practice, both user and product charges are usually set at a level only sufficient to recover the *private costs* of operating the public system, rather than being set at a level selected to create proper incentives for reducing pollution to the socially optimal level.

16 For more information on how the government can use revenues from taxes to offset distortions created by other taxes, see Goulder (1995) and Goulder et al. (1997).

17 These taxes are called "Pigovian" after the economist, Arthur Pigou, who first formalized them. See Pigou (1932).

18 See Fullerton (1996) for a discussion of the advantages and disadvantages of emission taxes.

Taxes and charges facilitate environmental improvements similar to those that result from marketable permit systems. Rather than specifying the total quantity of emissions, however, taxes, fees, and charges specify the effective "price" of emitting pollutants.

4.3.3 Environmental Subsidies

Subsidies paid by the government to firms or consumers for per unit reductions in pollution create the same abatement incentives as emission taxes or charges. If the government subsidizes the use of a cleaner fuel or the purchase of a particular control technology, firms will switch from the dirtier fuel or install the control technology to reduce emissions up to the point where the private costs of control are equal to the subsidy. It is important to keep in mind that an environmental subsidy is designed to correct for an externality not already taken into account by firms when making production decisions. This type of subsidy is fundamentally different from the many subsidies already in existence in industries such as oil and gas, forestry, and agriculture, which exist for other reasons apart from environmental quality, and therefore can exacerbate existing environmental externalities.

Unlike an emissions tax, a subsidy lowers a firm's total and average costs of production, encouraging both the continued operation of existing polluters that would otherwise exit the market, and the entry into the market by new firms that would otherwise face a barrier to entry. Given the potential entrance of new firms under a subsidy, the net result may be a decrease in pollution emissions from individual polluters but an increase in the overall amount.[19] For this reason, subsidies and taxes may not have the same aggregate social costs, or result in the same degree of pollution control. A subsidy also differs from a tax because it requires government expenditure. Analysts should always consider the opportunity costs associated with using public funds.

It is possible to minimize the entry and exit of firms resulting from subsidies by redefining the subsidy as a partial repayment of verified abatement costs, instead of defining it as a per unit payment for emissions reductions relative to a baseline. Under this definition, the subsidy now only relates to abatement costs incurred and does not shift the total or average cost curves, thereby leaving the entry and exit decisions of firms unaffected. Defining the subsidy in this way also minimizes strategic behavior because no baseline must be specified. [20]

Instead of pursuing a per unit emissions subsidy, the government may choose to lower the private costs of particular actions to the firm or consumer through cost sharing. For example, if the government wishes to encourage investment in particular pollution control technologies, the subsidy may take the form of reduced interest rates, accelerated depreciation, direct capital grants, and loan assistance or guarantees for investments. Cost-sharing policies alone may not induce broader changes in private behavior. In particular, such subsidies may encourage investment in pollution control equipment, rather than encouraging other changes in operating practices such as recycling and reuse, which may not require such costly capital investments. However, in conjunction with direct controls, pollution taxes, or other regulatory mechanisms, cost sharing may influence the nature of private responses and the distribution of the cost burden. As is the case with emissions taxes, subsidy rates also can be adjusted to account for both spatial and temporal variability.

A government "buy-back" constitutes another type of subsidy. Under this system, the government either directly pays a fee for the return of a product or subsidizes firms that purchase recycled materials. For instance, consumers may be offered

19 See Sterner (2003) for a more in-depth discussion of how subsidies work and for numerous examples of subsidy programs in the United States and other countries.

20 Strategic behavior is a problem common to any instrument or regulation that measures emissions relative to a baseline. In cases where a firm or consumer may potentially receive funds from the government, they may attempt to make the current state look worse than it actually is, in order to receive credit for large improvements. If firms or consumers are responsible for paying for certain emissions above a given level, they may try to influence the establishment of that level upward in order to pay less in fines or taxes.

a cash rebate on the purchase of a new electric or push mower when they scrap their old one. The rebate is earned when the old gasoline mower is turned in and a sales receipt for the new device is provided.[21] Buy-back programs also exist to promote the scrapping of old, high-emission vehicles.

Environmental subsidies in the United States have been used to encourage proper waste management and recycling by local governments and businesses; the use of alternative fuel vehicles by public bus companies, consumers, and businesses; and land conservation by property owners using cost-sharing measures. While most of these subsidies are not defined per unit of emissions abated, they can be effective when the behavioral changes they encourage are closely related to the use of products with reduced emissions.

4.3.4 Tax-Subsidy Combinations

Emission taxes and environmental subsidies can also be combined to achieve the same level of abatement as achieved when the tax and subsidy instruments are used separately. One example of this type of instrument is referred to as a **deposit-refund system** in which the deposit operates as a tax and the refund serves as a partially offsetting subsidy. As with the other market instruments already discussed, a deposit-refund system creates economic incentives to return a product for reuse or proper disposal, or to use a particular input in production, provided that the deposit exceeds the private cost of returning the product or switching inputs.

Under the deposit-refund system, the deposit is applied to either output or consumption, under the presumption that all production processes of the firm pollute or that all consumption goods become waste. A refund is then provided to the extent that the firm or consumer provides proof of the use of a cleaner form of production or of proper disposal. In the case where a deposit-refund is used to encourage firms to use a cleaner input, the deposit on output induces the firm to reduce its use of *all* inputs, both clean and dirty. The refund, however, provides the firm with an incentive to switch a specific input or set of inputs that result in a refund, such as a cleaner fuel or a particular pollution control technology.

A tax and offsetting subsidy combination functions best when it is possible to discern a direct relationship between an input, or output, and emissions. For instance, a tax on the production or use of hydrochlorofluorocarbons (HCFCs) combined with a refund for HCFC recycled or collected in a closed system is a good proxy for a direct emissions tax on ozone depletion.[22]

The most common type of tax-subsidy combination is the deposit-refund system, which is generally designed to encourage consumers to reduce litter and increase the recycling of certain components of municipal solid waste.[23] The most prominent examples are deposit-refunds for items such as plastic and glass bottles, lead acid batteries, toner cartridges and motor oil. Other countries have implemented deposit-refund systems on a wider range of products and behaviors that contribute to pollution, including the sulfur content of fuels (Sweden), product packaging (Germany), and deforestation (Indonesia). Tax-subsidy combinations have also been discussed in the literature as a means of controlling nonpoint source water pollution, cadmium, mercury, and the removal of carbon from the atmosphere.[24]

The main advantage of a combined tax and subsidy is that both parts apply to a market transaction. Because the taxed and subsidized items are easily observable in the market, this type of economic instrument may be particularly appealing when it is difficult to measure emissions or to control illegal dumping. In addition, polluters have an incentive to reveal accurate information on abatement activity to qualify for the subsidy.

21 For more information on the Office of Air's Small Engine Buy-back Program see U.S. EPA (2006c).

22 See Sterner (2003) for a more detailed description of this and other examples of tax-subsidy combinations.

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

24 See U.S. EPA (2004a), Fisher et al. (1995), and O'Connor (1994).

Because firms have access to better information than the government does, they can measure and report emissions with greater precision and at a potentially lower cost.

Disadvantages of the combined tax-subsidy system may include potentially high implementation and administrative costs, and the political incentive to set the tax too low to induce proper behavior (a danger with any tax). Policy makers may adjust an emissions tax to account for temporal variation in marginal environmental damages, but a tax on output sold in the market cannot be matched temporally or spatially to emissions during production. In addition, to the extent that emissions (e.g., SO_2 from power plants) are easily and accurately monitored, other market incentives may be more appropriate. If a production process has many different inputs with different contributions to environmental damages, then it is necessary to tax the inputs at different rates to achieve efficiency. Likewise, if firms are heterogeneous and select a different set of clean inputs or abatement options based on firm-specific cost considerations, then the subsidy should be adjusted for differences in these production functions.[25] A uniform subsidy combined with an output tax may be a good proxy, however, when there is limited heterogeneity across inputs' contribution to emissions and across firms.

Conceptually similar to the tax-subsidy combination is the requirement that firms post performance bonds that are forfeited in the event of damages, or that firms contribute up-front funds to a pool. Such funds may be used to compensate victims in the event that proper environmental management of a site for natural resource extraction does not occur. To the extent that the company demonstrates it has fulfilled certain environmental management or reclamation obligations, the deposited funds are usually refunded. Financial assurance requirements have been used to manage closure and post-closure care for hazardous waste treatment, storage, and disposal facilities. Performance bonds have also been required in extraction industries such as mining, timber, coal, and oil.[26]

4.4 Other Market-Oriented or Hybrid Approaches

In addition to the four classic market-based instruments discussed above, several other market-oriented approaches are often discussed in the literature and are increasingly used in practice. Often, these approaches combine aspects of command-and-control and market-based incentive policies. As such, they do not always present the most economically efficient approach. Either the level of abatement or the cost of the policy is likely to be greater than what would be achieved through the use of a pure market-based incentive approach. Nevertheless, such approaches are appealing to policy makers because they often combine the certainty associated with a given emissions standard with the flexibility of allowing firms to pursue the least costly abatement method. This section discusses the following market-oriented approaches:

- Combining standards and pricing approaches;
- Liability rules; and
- Information as regulation.

4.4.1 Combining Standards and Pricing Approaches

Pollution standards set specific emissions limits, thereby reducing the probability of excessively high damages to health or the environment. Such standards may impose large costs on polluters. Emissions taxes restrict costs by allowing polluters to pay a tax on the amount they emit rather than undertake excessively expensive abatement. Taxes, however, do not set a limit on emissions, and leave open the possibility that pollution may be excessively high. Some researchers suggest a policy that limits both costs and pollution, referred to as a "safety-valve" approach to regulation, which combines standards with pricing mechanisms.[27] In the case of a standard and tax combination, the same emissions standard is imposed on all

25 The main advantages and disadvantages of deposit-refund systems are discussed in U.S. GAO (1990); Palmer, Sigman, and Walls (1997); and Fullerton and Wolverton (2001, 2005).

26 For more information on the use of financial assurance or performance bonds, see Boyd (2002).

27 See Roberts and Spence (1976) and Spence and Weitzman (1978).

polluters and all polluters are subject to a unit tax for emissions in excess of the standard.

While a standard and pricing approach does not necessarily ensure the maximization of social welfare, it can lead to the most cost-effective method of pollution abatement. This policy combination has other attractive features. First, if the standard is set properly, the desired protection of health and the environment will be assured. This feature of the policy maintains the great advantage of a standards approach: protection against excessively damaging pollution levels. Combining approaches allows for more certainty in the expected environmental and health effects of the policy than would occur with a market-based approach alone. Second, high abatement cost polluters can defray costs by paying the emissions fee instead of cleaning up. This feature preserves the flexibility of emissions taxes: overall abatement costs are lower because polluters with low abatement costs reduce pollution while polluters with high abatement costs pay taxes.

4.4.2 Information Disclosure

Requiring disclosure of environmental information has been increasingly used as a method of environmental regulation. Disclosure strategies are most likely to work when there is a link between the polluting firm and affected parties such as consumers and workers.[28] Disclosure requirements attempt to minimize inefficiencies in regulation associated with asymmetric information, such as when a firm has more and better information on what and how much it pollutes than is available to the government or the public. By collecting and making such information publicly available, firms, government agencies, and consumers can become better informed about the environmental and human health consequences of their production and consumption decisions. In some cases, the availability of this information may also encourage more environmentally benign activities and discourage environmentally detrimental ones. For example, warning labels on hazardous substances that describe safe-handling procedures or the risks posed by the product may encourage hazardous substance handlers to take greater precautions, and/or may encourage consumers to switch to less damaging substitutes for some or all uses of the substance. Similarly, a community with information on a nearby firm's pollution activity may exert pressure on the firm to reduce emissions, even if formal regulations or monitoring and enforcement are weak or nonexistent.[29]

Requirements for information disclosure need not be tied explicitly to an emissions standard; however, such requirements are consistent with a standard-based approach because the information provided allows a community to easily understand the level of emissions and the polluters' level of compliance with existing standards or expectations. As is the case with market-based instruments, polluters still have the flexibility to respond to community pressure by reducing emissions in the cheapest way possible.

The use of information disclosure or labeling rules has other advantages. When expensive emissions monitoring is required to collect such information, reporting requirements that switch the burden of proof for monitoring and reporting from the government to the firm might result in lower costs, because firms are often in a better position to monitor their own emissions. If accompanied by spot checks to ensure that monitoring equipment functions properly and that firms report results accurately, information disclosure can be an effective form of regulation. Without the appropriate monitoring, however, information disclosure might not result in an efficient outcome.

While information disclosure has its advantages, it is important to keep three caveats in mind when considering this method for environmental regulation. First, the use of information as regulation is not costless: U.S. firms report spending approximately $346 million per year

28 See OMB (2010b) for guidance issued to regulatory agencies on the use of information disclosure and simplification in the regulatory process.

29 For more information on how information disclosure may help to resolve market failures, see Pargal and Wheeler (1996), Tietenberg (1998), Tietenberg and Wheeler (2001), and Brouhle and Khanna (2007).

to monitor and report releases.[30] Any required investments in pollution control are in addition to this amount. Second, the amount of pressure a community exerts on an emitting plant may be related to socioeconomic status. Poorer, less-educated populations tend to exert far less pressure than communities with richer, well-educated populations.[31] Third, information disclosure may not result in a socially efficient level of pollution when consumers either consider only the effect of emissions on them as individuals, ignoring possible ecological or aggregate societal effects, or when they do not understand how to properly interpret the released information in terms of the health risks associated with exposure to particular pollutants.

EPA-led information disclosure efforts include the Toxics Release Inventory (TRI) and the mandatory reporting of greenhouse gases (GHG). Both the TRI and the GHG reporting rule require firms to provide the government and public with information on pollution at each plant, on an annual basis, if emissions exceed a threshold. There are also consumer-based information programs targeting the risks of particular toxic substances, the level of contamination in drinking water, the dangers of pesticides, and air quality index forecasts for more than 300 cities. There is some evidence in the literature regarding the impact of TRI reporting on firm value: the most polluting firms experience small declines in stock prices on the day TRI emission reports are released to the public. Hamilton (1995) finds a stock price return of -0.03 percent due to TRI report release. Firms that experienced the largest drop in their stock prices also reduced their reported emissions by the greatest quantity in subsequent years.[32]

4.4.3 Liability Rules

Liability rules are legal tools of environmental policy that can be used by victims (or the government) to force polluters to pay for environmental damages after they occur. These instruments serve two main purposes: (1) to create an economic incentive for firms to incorporate careful environmental management and the potential cost of environmental damages into their decision-making processes; and (2) to compensate victims when careful planning does not occur. These rules are used to guide courts in compensation decisions when the court rules in favor of the victim. Liability rules can serve as an incentive to polluters. To the extent that polluters are aware that they will be held liable before the polluting event occurs, they may minimize or prevent involvement in activities that inflict damages on others. In designing a liability rule it is important to evaluate whether damages depend only on the amount of care taken on the part of the polluter or also on the level of output; and whether damages are only determined by polluter actions or are also dependent on the behavior of victims. For instance, if victims do not demonstrate some standard of care in an attempt to avoid damages, the polluter may not be held liable for the full amount. If damages depend on these other factors in addition to polluter actions, then the liability rule should be designed to provide adequate incentives to address these other factors.

While a liability rule can be constructed to mimic an efficient market solution in certain cases, there are reasons to expect that this efficiency may not be achieved. First, uncertainty exists as to the magnitude of payment. The amount that polluters are required to pay after damages have occurred is dependent on the legal system and may be limited by an inability to prove the full extent of damages or by the ability of the firm to pay. Second, liability rules can generate relatively large costs, both in terms of assessing the environmental damage caused, and the damages paid.[33] Thus, liability rules are most useful in cases where damages requiring compensation are expected to be stochastic (e.g., accidental releases), and where monitoring firm compliance with regulatory procedures is

30 See O'Connor (1996) for information on the costs of monitoring and reporting environmental information. See World Bank (2000) for a discussion of the main advantages and disadvantages of information disclosure as a policy tool.

31 See Hamilton (1993), and Arora and Cason (1999).

32 Hamilton (1995); Konar and Cohen (1997); and Khanna, Quimio, and Bojilova (1998) are empirical studies that have investigated how the TRI has affected firm behavior and stock market valuation.

33 See Segerson (1995), and Alberini and Austin (2001) for discussions of the types of liability rules, the efficiency properties of each type of rule, and an extensive bibliography.

difficult. Depending on the likely effectiveness of liability rules to provide incentives to firms to avoid damages, they can be thought of as either an alternative to or as a complement to other regulatory approaches.

Strict liability and *negligence* are two types of liability rules relevant to polluters. Under strict liability, polluters are held responsible for all health and environmental damage caused by their pollution, regardless of actions taken to prevent the damages. Under negligence, polluters are liable only if they do not exhibit "due standard of care." Regulations that impose strict liability on polluters may reduce the transactions costs of legal actions brought by affected parties. This may induce polluters to alter their behavior and expend 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.

Liability rules have been used in the remediation of contaminated sites under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), also known as *Superfund*, and under the Corrective Action provisions of the Resource Conservation and Recovery Act (RCRA). These rules have also been used in the redevelopment of potentially contaminated industrial sites, known as *brownfields*.

4.5 Selecting the Appropriate Market-Based Incentive or Hybrid Approach

Selection of the most appropriate market-based incentive or hybrid regulatory approach depends on a wide variety of factors, including: [34]

- The type of market failure being addressed;
- The specific nature of the environmental problem;
- The type of pollutant information that is available and observable;
- The degree of uncertainty surrounding costs and benefits;
- Concerns regarding market competitiveness;
- Monitoring and enforcement issues;
- Potential for exacerbating economy-wide distortions; and
- The ultimate goals of policy makers.

4.5.1 The Type of Market Failure

There are two main types of market failure that are commonly addressed through the use of market-based or hybrid instruments. The first, externality, occurs when firms or consumers fail to integrate into their decision making the impact of their own production or consumption decisions on entities external to themselves. The second type of market failure, asymmetric information, occurs when firms or consumers are unable to make optimal decisions due to lack of information on available abatement technologies, emission levels, or associated risks. Market-based or hybrid instruments that incorporate the marginal external damages of a unit of pollution into a firm or consumer's cost function address the first type of market failure. Information disclosure or labeling are often suggested when the second type of market failures occurs. As discussed in Section 4.4.2, policy makers believe that private- and public-sector decision makers will act to address an environmental problem once information has been disseminated.

4.5.2 The Nature of the Environmental Problem

The use of a particular market-oriented approach is often directly associated with the nature of the environmental problem. Do emissions derive from a point source or a nonpoint source? Do emissions stem from a stock or flow pollutant? Are emissions uniformly mixed or do they vary by location? Does pollution originate from stationary or mobile

34 Helpful references that discuss aspects to consider when comparing among different approaches include Hahn and Stavins (1992), OECD (1994a, 1994b), Portney and Stavins (2000), and Sterner (2003).

sources?[35] Point sources, which emit at identifiable and specific locations, are much easier to control than diffuse and often numerous nonpoint sources, and therefore are often responsive to a wide variety of market instruments. Although nonpoint sources are not regulated under EPA, the pollution emitted from a nonpoint source is. Clearly, this makes the monitoring and control of nonpoint source emissions challenging. In instances where both point and nonpoint sources contribute to a pollution problem, a good case can be made for a tax-subsidy combination or a marketable permit system. Under these alternatives, emissions from point sources might be taxed while nonpoint source controls are subsidized.

Flow pollutants tend to dissipate quickly, and it is possible to rely on a wide variety of market and hybrid instruments for emissions control. But stock pollutants persist in the environment and tend to accumulate over time. Controlling stock pollutants may require strict limits to prevent bioaccumulation or detrimental health effects at small doses, making direct regulation a potentially more appealing approach. If these limits are not close to zero, then potentially practical instrument options include a standard-and-pricing approach or a marketable permit approach that defines particular trading ratios to ensure that emission standards are not violated at any given source are. These same instruments are appealing when pollutants are not uniformly mixed across space. In the case of non-uniformly mixed emissions, it is important to account for differences in baseline pollution levels, and differences in emissions across more and less polluted areas.

Stationary sources of pollution are easier to identify and control through a variety of market instruments than are mobile sources. Highly mobile sources are usually numerous, each emitting a small amount of pollution. Emissions therefore vary by location and damages can vary by time of day or season. For example, health impacts associated with vehicle traffic are primarily a problem at rush hour when roads are congested and cars spend time idling or in stop-and-go traffic. Differential pricing of resources used by these mobile sources (such as higher tolls on roads or greater subsidies to public transportation during rush hour) is a potentially useful tool.

35 For a detailed discussion of how the nature of the environmental problem affects instrument choice, see Kahn (1998), Goulder et al. (1999), Parry and Williams (1999), Harris (2002), Tietenberg (2002), and Sterner (2003).

4.5.3 The Type of Pollutant Information that is Available and Observable

The selection of market-oriented approach may depend on the available data. Is the level of pollutant actually observable or measurable? Or will the level need to be imputed based on inputs and technology used? Are the sources heterogeneous? Does the pollutant vary across time and space? Are information technologies available to the analyst to improve data collection? When the pollutant concentration can be directly and easily measured then it is possible to directly regulate the level of the pollutant. But if monitoring costs are high, it may be easier to target a particular input or require a specific technology known to reduce pollutants by a certain amount. The pollutant levels can be imputed based on regulation placed on the input or the technology used.

The link between pollution and heterogeneous sources is often difficult and costly to determine, and costs may increase if the pollutant levels vary over time. Uniform policies are often used for the sake of simplicity. However, information technologies such as continuous emissions monitoring equipment (CEMs) or geographical information systems (GIS) can be used to link sources to pollutant levels. In these cases, policies that make use of this new information may be used and often can reduce costs. As technology improves or more data become available, analysts should consider reassessing the regulation design.[36]

4.5.4 Uncertainty in Abatement Costs or Damages

The choice between price-based instruments (e.g., taxes or charges) and quantity-based

36 For more information see Xabadia, Goetz, and Zilberman (2008).

instruments (e.g., marketable permits) has been shown theoretically to rest on the uncertainty surrounding estimated benefits and costs of pollution control, as well as on how marginal benefits and costs change with the stringency of the pollution control target. If uncertainty associated with the cost of abatement exists but damages do not change much with additional pollution, then policy makers can effectively limit costs by using a price instrument without having much impact on the benefits of the policy. If, on the other hand, there is more uncertainty associated with the benefits of controlling pollution and policy makers wish to guard against high environmental damages, a quantity instrument is likely preferable.[37] In this way, the policy maker can avoid potentially costly or damaging mistakes. The policy maker should also be aware of any discontinuities or threshold values above which sudden large changes in damages or costs could occur in response to a small increase in the required abatement level.

4.5.5 Market Competitiveness

Market power is a type of market failure in and of itself, as it may result in output that is too low and prices that are too high compared to what would occur in a competitive market. Instruments that cause firms to further restrict output may create additional inefficiencies in sectors where firms have some degree of market power. A combination of market-based instruments may work more effectively than a single instrument in this instance. To the extent that cost burdens are differentiated, the use of certain market-based instruments may cause a change in market structure that favors existing firms by creating barriers of entry and allowing existing firms a certain amount of control over price. Permit systems that set aside a certain number of permits for new firms, for instance, may guard against such barriers.

4.5.6 Monitoring and Enforcement Issues

Market-oriented instruments differ in the degree of effort required to monitor and enforce the desired emissions level. For example, subsidies, deposit-refund systems, and information disclosure shift the burden of proof to demonstrate compliance from government to the regulated entities. Because firms are generally in a better position than government to monitor and report their own emissions, they likely can do so at a potentially lower cost. This feature makes such approaches attractive when monitoring is difficult or emissions must be estimated (e.g., when there are nonpoint sources or large numbers of small polluters). In these cases, attempts to prohibit or tax the actions of polluters are likely to fail due to the risk of widespread noncompliance (e.g., illegal dumping to avoid the tax) and costly enforcement.

4.5.7 Potential for Economy-Wide Distortions

Analysts should consider the potential distortionary effects of any policy option considered. Even if a policy is deemed relatively efficient on its own, it may interact with pre-existing environmental, economic, or agricultural policies (e.g., product standards, non-environmental subsidies, trade barriers) in non-intuitive ways that can exacerbate distortions in the economy and result in unintended environmental consequences. Instruments that include a revenue-raising component, such as auctioned permits or taxes, may allow for opportunities to direct collected resources to reduce other taxes and fees and the associated inefficiencies.[38] See Chapter 8 and Appendix A for a more detailed discussion of economy-wide distortions.

37 See Weitzman (1974) for the classic paper on the ways in which uncertainty (also referred to as lack of information) affects instrument choice. See Chapter 10 of these *Guidelines* for more information on the treatment of uncertainty in analyses.

38 For useful references on the 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 (1995), Bovenberg and Goulder (1996), Goulder et al. (1997), and Jorgenson (1998a, 1998b).

4.5.8 The Goals of the Policy Maker

Finally, the goals of policy makers may influence the instrument selected to regulate pollution. Each considered instrument may have different distributional and equity implications for both costs and benefits; these implications should be accounted for when deciding among instruments. For example, policy makers may wish to ensure clean-up of future pollution by firms. Policy makers may consider using insurance and financial assurance mechanisms 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 method. In addition, the degree to which policy makers want to allow the market to determine exact outcomes may influence the choice of instrument. The quantity of marketable permits issued, for example, sets the total level of pollution control, but the market determines which polluters reduce emissions. On the other hand, taxes let the market determine both the extent of control by individual polluters and the total level of control.

4.6 Non-Regulatory Approaches

EPA has pursued a number of non-regulatory approaches that rely on **voluntary initiatives** to achieve emissions reductions and improve management of environmental hazards. These programs are usually not intended as substitutes for formal regulation, but instead act as important complements to existing regulation. Many of EPA's voluntary programs encourage polluting entities to go beyond what is mandated by existing regulation. Other voluntary programs have been developed to improve environmental quality in areas that policy makers expect may be regulated in the future but are currently not regulated, such as GHG emissions and nonpoint source water pollution.[39]

Much of the technical foundation for these voluntary initiatives rests on the concepts underlying a "pollution prevention" approach to environmental management choices. In the Pollution Prevention Act of 1990, Congress established 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 environmental should be employed as a last resort and should be conducted in an environmentally safe manner.

EPA typically designs its voluntary programs through regular consultation (but little direct negotiation) with affected industries or consumers.[40] In many cases, voluntary programs facilitate problem solving between EPA and industry because information on procedures or practices that reduce or eliminate the generation of pollutants and waste at the source are shared through the consultative process.

In slightly more than a decade, voluntary programs at EPA have increased from two programs to approximately 40 programs involving more than 13,000 organizations. Partner organizations include small and large businesses, citizen groups, state and local governments, universities, and trade associations.[41] Voluntary programs in which these groups participate tend to have either broad environmental objectives targeting a variety of firms from different industries, or focus on more specific environmental problems relevant to a single industrial sector. In the United States, nearly

39 While this chapter only discusses government-led voluntary initiatives at the federal level at EPA, other government agencies, industry, non-profits, and international organizations have also initiated and organized voluntary initiatives designed to address particular environmental issues. These initiatives are beyond the scope of this chapter, which limits itself to a brief description of policy options available to EPA.

40 Because these programs are voluntary there is no need for formal public comment. However, industry often is consulted during the design phase.

41 For information on EPA's voluntary programs, see the Partners for the Environment List of Programs at http://www.epa.gov/partners/programs/index.htm (accessed November 03, 2010) (U.S. EPA 2008e).

one third of all multi-sector federal voluntary programs focus on energy efficiency and climate change issues. General pollution prevention efforts represent the next most popular type of voluntary program. Single-sector federal voluntary programs tend to target environmental problems associated with transportation-related issues and energy producing sectors such as coal mining and power generation. These programs strive to provide participating firms with targeted and effective technological expertise and assistance.[42]

4.6.1 How Voluntary Approaches Work

Voluntary programs can use the following four general methods to achieve environmental improvements: (1) require firms or facilities to set specific environmental goals; (2) promote firm environmental awareness and encourage process change; (3) publicly recognize firm participation; and (4) use labeling to identify environmentally responsible products. These methods are not mutually exclusive, and most U.S. voluntary programs use a combination of methods.

Goal setting is a very common method used in the design of voluntary programs. Implementation-based goals are typically EPA-specified, program-wide targets designed to provide a consistent objective across firms. Target-based goals are usually qualitative and process-oriented so that firms may individually set a unique target. EPA's WasteWise and Climate Challenge programs are examples of programs with target-based goals. EPA's 33/50 program, which set a goal of a 33 percent reduction of toxic emissions by firms in the chemical industry by 1992, and a 50 percent reduction by 1995 (relative to a 1988 baseline), is an example of a voluntary program with an implementation-based goal.

Programs designed to promote environmental awareness and to encourage process change within firms often involve implementing a system to evaluate firms' ongoing operations and to provide information on newly available technologies. Examples of this type of approach include the SmartWay program, which encourages firms to adopt energy efficient changes that also yield fuel savings for freight trucking companies, and the Green Suppliers Network program, which provides partner firms with technical reviews and suggestions on how to eliminate waste from production processes.

Voluntary programs that publicly recognize firm participation are designed to provide green consumers and investors with new information that may alter their consumption and investment patterns in favor of cleaner firms. Firms may also use their environmental achievements to differentiate their products from competitors' products.[43] These information and firm differentiation effects are the intent of the Green Power Partnership and the WasteWise program.

Finally, product labeling can be applied to either intermediate inputs in a production process or to a final good. Labels on intermediate goods encourage firms to purchase environmentally responsible inputs. Labels on final goods allow consumers to identify goods produced using a relatively clean production process. For example, products deemed by EPA to be energy efficient may be eligible for the Energy Star or Design for the Environment labels.

4.6.2 Economic Evaluation of Voluntary Approaches

A formal economic analysis is not required for the selection and implementation of a non-regulatory or voluntary approach to pollution reduction.

Several factors contribute to the difficulty of evaluating voluntary approaches. Many programs target general environmental objectives and thus

42 See Khanna (2001); OECD (1999, 2003); U.S. EPA (2002a); and Brouhle, Griffiths, and Wolverton (2005) for discussions of how voluntary programs work and how they are used in U.S. environmental policy making.

43 See Arora and Cason (1995); Arora and Gangopadhyay (1995); Konar and Cohen (1997, 2001); Videras and Alberini (2000); Brouhle, Griffiths, and Wolverton (2005); and Morgenstern and Pizer (2007) for more information on the main arguments for why firms participate in voluntary programs.

Text Box 4.3 - Water Quality Trading of Nonpoint Sources

In 2003, EPA issued a "Water Quality Trading Policy" (U.S. EPA 2003d) that encourages states and tribes to develop and implement voluntary water-quality trading to control nutrients and sediments in areas where it is possible to achieve these reductions at lower costs. Under the Clean Water Act, EPA is required to establish Total Maximum Daily Loadings (TMDL) of pollutants for impaired water bodies. The TMDL does not establish an aggregate cap on discharges to the watershed, but it does provide a method for allocating pollutant discharges among point and nonpoint sources. Point sources are regulated by EPA and, as such, are required to hold National Pollutant Discharge Elimination System (NPDES) permits that limit discharges. However, many water bodies are still threatened by pollution from unregulated, nonpoint sources. Nutrients and sediment from urban and agricultural runoff have led to water quality problems that limit recreational uses of rivers, lakes, and streams; that create hypoxia in the Gulf of Mexico; and that decrease fish populations in the Chesapeake Bay. The impetus for allowing effluent trading between point and nonpoint sources is to lower nutrient and sediment loadings and to improve or preserve water quality.

To ensure that the reduction resulting from the trade has the same effect on the water quality as the reduction that would be required without the trade, trading ratios are often applied. These ratios attempt to control for the differential effects resulting from a variety of factors, which may include:

- location of the sources in the watershed relative to the downstream area of concern;
- distance between the permit buyer and seller;
- uncertainty about nonpoint source reductions;
- equivalency of different forms of the same pollutant discharged by the trading partners; and
- additional water quality improvements above and beyond those required by regulation.

The idea behind trading is to allow point sources to meet the discharge limit at a lower cost. This allows continued growth and expansion of production, while giving nonpoint sources an incentive to reduce pollution through participation in the market. To the extent that it is cheaper for a nonpoint source to reduce pollution than to forgo revenues earned from the sale of any unused credits to point sources, the nonpoint source is predicted to choose to emit less pollution.

As of March 2007, 98 NPDES permits, covering 363 dischargers, included provisions for trading. However, only about a third of the dischargers had carried out one or more trades under these permits (U.S. EPA 2007f). Trading has been limited for several reasons. First, there is no aggregate "cap" on discharges that applies to both point and nonpoint sources within a watershed. Reductions by nonpoint sources are essentially voluntary. Point-source dischargers often explore trading as a way to expand production while meeting the requirements of their individual permits, but there is no general signal in the market to do so. Second, these are often thin markets. The way in which the market is designed or trading ratios are established can make it difficult or expensive for an entity to identify and complete a trade. Third, while Best Management Practices (BMPs) are typically used to define a pollution reduction credit from a nonpoint source, uncertain or changing climatic conditions, river flow, and stream conditions make it difficult to measure the effect of a BMP on water quality. Such uncertainty also makes measuring and enforcing a pollution reduction from a nonpoint source difficult. Fourth, encouraging nonpoint source involvement in trading, given the agriculture industry's distrust of regulators, is challenging. Finally, it is difficult to define appropriate trading ratios between point and nonpoint sources.

lack a measurable environmental outcome. Even if a measurable output exists, there may be a lack of data on a firm's or industry's environmental outputs. In order to perform an evaluation, a reasonable baseline from which to make a comparison must be established. This requires an extensive analysis comparing the actions of participants to non-participants in the program;

such data is likely difficult and costly to obtain.[44] Any economic evaluation of voluntary programs should net out pollution abatement activities that would have occurred even if the voluntary program were not in place. Some of these evaluation obstacles can be overcome if voluntary approaches use more defined and detailed goal setting and require more complete data collection and reporting from the outset.[45]

The economic literature evaluating the efficacy of voluntary programs is decidedly mixed. The vast majority of existing empirical studies focus on a few large, multi-sector voluntary programs such as 33/50, Green Lights, and Energy Star. For these programs, there is some evidence of success in reducing participant emissions. However, studies generally fail to account for non-program factors such as the ability to count reductions that occurred prior to the start of the program; to compare reductions relative to a baseline counterfactual may overstate these reductions. Researchers have been less successful in demonstrating that voluntary programs have led to greater emission reductions than would have occurred without the program in place. One thread of literature points to the positive impact of a regulatory threat on voluntary program effectiveness. When the threat of regulation is weak, abatement levels are likely to be lower. However, when the threat of regulation is strong, levels achieved are closer to those under optimal regulatory action.

4.7 Measuring the Effectiveness of Regulatory or Non-Regulatory Approaches

Several policy criteria should be considered when evaluating the success of regulatory or non-regulatory approaches. These include environmental effectiveness; economic efficiency; savings in administrative, monitoring and enforcement costs; inducement of innovation; and increased environmental awareness. In many cases, analysis of these factors will make evident the particular advantages of one or more market-based incentive approaches over command-and-control regulation. While a formal analysis may not be required when considering the implementation of a non-regulatory approach, these factors are still important to consider. According to recent reviews (Harrington et al. 2004, and Goulder and Parry 2008) it is unlikely that any one policy will dominate on all of these factors. However, in many areas an incentive policy, if available, can be more cost-effective than a competing command-and-control policy.

In determining the effectiveness of a policy approach, policy makers should consider the following factors and questions:

- **Environmental Effectiveness:** Does the policy instrument accomplish a measurable environmental goal? Does the policy instrument result in general environmental improvements or emission reductions? Does the approach induce firms to reduce emissions by greater amounts than they would have in the absence of the policy?
- **Economic Efficiency:** How closely does the approach approximate the most efficient outcome? Does the policy instrument reach the environmental goal at the lowest possible cost to firms and consumers?
- **Reductions in Administrative, Monitoring, and Enforcement Costs:** Does the government benefit from reductions in costs? How large are these cost savings compared to those afforded by other forms of regulation?
- **Environmental Awareness and Attitudinal Changes:** In the course of meeting particular goals, are firms educating themselves on the nature of the environmental problem and ways in which it can be mitigated? Does the promotion of firm participation or compliance affect consumers' environmental awareness or priorities and result in a demand for greater emissions reductions?
- **Inducement of Innovation:** Does the policy instrument lead to innovation in

44 See Chapter 5 for a discussion of baselines and specifically Section 5.7 for a discussion of behavioral responses.

45 See Segerson and Miceli (1998); Khanna and Damon (1999); National Research Council (2002); Segerson and Wu (2006); Morgenstern and Pizer (2007); and Brouhle, Griffiths, and Wolverton (2009).

abatement techniques that decrease the cost of compliance with environmental regulations over time?

To address a number of these key evaluation criteria, *Guidelines* Chapters 8 and 9 offer instruction on how to measure social costs and how to address equity issues, respectively.

Chapter 5
Baseline

The baseline of an economic analysis is a reference point that reflects the world without the proposed regulation. It is the starting point for conducting an economic analysis of the potential benefits and costs of a proposed regulation. Because an economic analysis considers the impact of a policy or regulation in relation to this baseline, its specification can have a profound influence on the outcome of the economic analysis. A careful and correct baseline specification assures the accuracy of benefit and cost estimates. The baseline specification can vary in terms of sources analyzed (e.g., facilities, industries, sectors of the economy), geographic resolution (e.g., census blocks, GIS grid cells, counties, state, regions), environmental objectives (e.g., effluents and emissions versus pollutant concentrations), and years covered. Because the level of detail presented in the baseline specification is an important determinant of the kinds of analysis that can be conducted on proposed regulatory options, careful thought in specifying the baseline is crucial.

The drive for a thorough, rigorous baseline analysis should be balanced against other competing objectives such as judicial and statutory deadlines, and legal requirements. The analyst is responsible for raising questions about baseline definitions early in the regulatory development process to ensure that the analysis is as comprehensive as possible. Doing so will facilitate analysis of regulatory changes to the baseline regulation.

5.1 Baseline Definition

A baseline is defined as the best assessment of the world absent the proposed regulation or policy action.[1] This "no action" baseline is modeled assuming no change in the regulatory program under consideration. This does not necessarily mean that no change in current conditions will take place, since the economy will change even in the absence of regulation. A proper baseline should incorporate assumptions about exogenous changes in the economy that may affect relevant benefits and costs (e.g., changes in demographics, economic activity, consumer preferences, and technology), industry compliance rates, other regulations promulgated by EPA or other government entities, and behavioral responses to the proposed rule by firms and the public.

On occasion a regulatory program may be set to expire or dramatically change, even in the absence of the proposed action. In this case, the baseline specification might consider a state of the world different from current conditions. This situation, however, is less common.

The baseline serves as a primary point of comparison for an analysis of a proposed policy action. An economic analysis of a policy or regulation compares the current state of the world, the *baseline scenario*, to the expected state of the world with the proposed policy or regulation in effect, the *policy scenario*. Economic and other impacts of policies or regulations are measured as the differences between these two scenarios.

1 A policy action includes both regulations and the issuance of Best Management Practices (BMPs) or guidance documents, which do not carry the same force as a regulation, but do affect the decisions of firms and consumers.

In most cases, a single, well-defined description of the world in the absence of the regulation is generally all that is needed as a baseline. A single baseline produces a clear point of comparison with the policy scenario and allows for an unequivocal measure of the benefits, costs, and other consequences of the rule. There are a few cases in which more than one baseline may be necessary.

Multiple baseline scenarios are needed, for example, when it is impossible to make a reasonable unique description of the world in the absence of the proposed regulation. For instance, if the current level of compliance with existing regulations is not known, then it may be necessary to compare the policy scenario to both a full compliance baseline and a partial compliance baseline. Further, if the impact of other rules currently under consideration fundamentally affects the economic analysis of the rule being analyzed, then multiple scenarios, with and without these rules in the baseline, may be necessary.

The decision to include multiple baselines should not be taken lightly as a complex set of modeling choices and analytic findings may result. These must be interpreted and communicated to decision makers, increasing the possibility of erroneous comparisons of costs and benefits across different baselines. When more than one baseline is required, analysts should endeavor to construct scenarios that can provide benchmarks for policy analysis. The number of baselines should be limited to as few as possible that cover the key dimensions of the economic analysis and any phenomena in the baseline about which there is uncertainty.

In some cases, probabilistic analysis can be used to avoid the need for multiple baselines and still provide an appropriate benchmark for policy analysis. A probabilistic analysis is a form of uncertainty analysis in which a single modeling framework is generally specified, but statistical distributions are assigned to the uncertain input parameters. The policy scenario is then compared to a continuum of baselines, with a probability for any given outcome, rather than being compared to a single baseline. The benefit-cost analysis (BCA) would then report the probability that a policy intervention produces net benefits rather than reporting the net benefits compared to one (or more) deterministic baseline(s).

Analysts are advised to seek clear direction from management about baseline definitions early on in the development of a rule. Each baseline-to-policy comparison should be internally consistent in its definition and use of baseline assumptions.

5.2 Guiding Principles of Baseline Specification

In specifying the baseline, analysts should employ the following guiding principles each of which is discussed more fully below:

1. Clearly specify the current and future state of relevant economic variables, the environmental problem that the regulation addresses and the regulatory approach being considered;
2. Identify all required parameters for the analysis;
3. Determine the appropriate level of effort for baseline specification;
4. Clearly identify all assumptions made in specifying the baseline conditions;
5. Specify the "starting point" of the baseline and policy scenario;
6. Specify the "ending point" of the baseline and policy scenario;
7. Detail all aspects of the baseline specification that are uncertain; and
8. Use the baseline assumptions consistently for all analyses for this regulation.

Though these principles exhibit a general common-sense approach to baseline specification, the analyst is advised to provide her own explicit statements on each point. Failure to do so may result in a confusing presentation, inefficient use of time and resources, and misinterpretation of the economic results.

Clearly specify the current and future state of relevant economic variables, the environmental problem that the regulation addresses and the regulatory approach being considered. A clear written statement about the current state of the relevant economic variables (see Chapter 8 in particular to determine what variables are relevant) and environment will help decision makers and the general public to understand both the positive and negative consequences of a regulation. The statement should include a description of: (1) the pollution problem being addressed; (2) the current regulatory environment; (3) the method by which the problem will be addressed; and (4) the affected parties. There should also be a discussion of why a particular approach [e.g., best available technology (BAT), performance measures, market incentives, or non-regulatory approaches] was chosen.

In general, the most appropriate baseline will be the "no change" or "reality in the absence of the regulation" scenario; but in some cases, a baseline of some other regulatory approach may be considered. For example, if an industry is certain to be regulated (e.g., by court order or congressional mandate) but that regulation has not yet been implemented, then a baseline including this regulation should be used. To ensure that provisions contained in statutes or policies preceding the regulatory action in question are appropriately addressed and measured, it is common practice to assume full compliance with regulatory requirements, although sensitivity analyses assuming less-than-full compliance may be considered. However, analysts should consult with their management and the Office of General Counsel (OGC) before doing so.

Identify all required parameters for the analysis. To ensure that the baseline scenario can be compared to the policy scenario, there should be a clear understanding of the path from environmental damage to adverse impact on humans. The models and parameters required for the baseline analysis should be chosen so that the baseline assumptions can feed into all subsequent analyses. Measured differences between the baseline and policy scenario can include changes in usage or production of toxic substances, changes in pollutant emissions and ambient concentrations, and incidence rates for adverse health effects associated with exposure to pollutants. This does not mean that the analyst must identify all parameters that could possibly change, but the analyst should recognize all relevant parameters needed to compare the baseline scenario to the policy scenario. As a general rule of thumb, at a minimum, the analyst should identify the parameters that are expected to vary by option, the parameters that are expected to have the largest impact on cost and benefit differences, and the parameters that are anticipated to come under close public scrutiny.

Determine the appropriate level of effort for baseline specification. The analyst should concentrate analytic efforts on those components (e.g., assumptions, data, models) of the baseline that are most important to the analysis, taking into consideration factors such as the time given to complete the analysis, the person-hours available, the cost of the analysis, and the available models and data. If several components of the baseline are uncertain, the analyst should concentrate limited resources on refining the estimates of those components that have the greatest effect on the interpretation of the results. Analysts should pay special attention to the components that will be used to calculate costs and benefits and those that are important determinants of the policy option selected.

Clearly identify all assumptions made in specifying the baseline conditions. Whether variables are modeled or set by fixed assumptions, the analyst should explain the assumptions and uncertainties about the parameters in detail. Assumptions should include changes in behavior and business trends, and how these trends may be affected by regulatory management options. Analysts 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 may change. Demand for some commodities may grow at rates faster than the rate of change in income, while demand for other goods may decrease. 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 the assumptions used in the analysis, with the goal of laying out the assumptions clearly enough so that other analysts (with access to the appropriate models) would be able to replicate the baseline specification.

Specify the "starting point" of the baseline and policy scenario. 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 the point in time at which the two scenarios diverge. For example, one approach is to organize the analysis assuming 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 as the promulgation of the final rule. These dates may be appropriate to use because they are clearly defined under administrative procedures or because they represent specific deadlines.

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 may not be the most appropriate starting point to define differences between the baseline and policy scenarios. Earlier starting points, such as the date when authorizing legislation was signed into law, the date the rule was first published in a Notice of Proposed Rulemaking, or other regulatory development process milestones, may be justified when divergence from the baseline occurs due to the anticipation of promulgation.

Specify the "ending point" of the baseline and policy scenario. The ending point of an analysis is the point in time at which the comparison between the baseline and policy scenarios ends. Generally, the duration of important effects of a policy determines the period chosen for the analysis and baseline. However, other analytical considerations, such as the relative uncertainty in projecting out-year conditions, may also need to be weighed. To compare the benefits and costs of a proposed policy, the analyst should estimate the present discounted values of the total costs and benefits attributable to the policy over the period of the study. How one defines the ending point of the baseline is particularly important in situations where the accrual of costs and/or benefits do not coincide due to lagged effects, or where they occur over an extended period of time. For example, the human health benefits of a policy that reduces leachate from landfills may not manifest themselves for many years if groundwater contamination occurs decades after closure of a landfill. In theory, the longer the time frame, the more likely the analysis will capture all of the major benefits and costs of the policy. Naturally, the forecasts of economic, demographic, and technological trends that are necessary for baseline specification should also span the entire period of the analysis. However, because forecasts of the distant future are less reliable than forecasts of the near future, the analyst should balance the advantages of structuring the analysis to include a longer time span against the disadvantages of the decreasing reliability of the forecasts for the future.

In some cases, the benefits of a policy are expected to increase over time. When this occurs, analysts should extend the analysis far enough into the future to ensure that benefits are not substantially underestimated. For example, suppose a proposed policy would greatly reduce greenhouse gas (GHG) emissions. In the baseline scenario, the level of GHG in the atmosphere would steadily increase over time, with a corresponding increase in expected impacts on human health and welfare and ecological outcomes. A BCA limited to the first decade after policy initiation would likely 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.

In some circumstances, it may make sense to model the annual flow of benefits and costs rather than model them over time. For example, if the benefits and costs remain constant (in real terms) over time, then an estimate for a single year is all that is necessary. The duration of the policy will not affect whether there are net benefits nor will it affect the choice of the most economically efficient option, although it will obviously still affect the magnitude of net benefits. In this case, an "ending point" may not be needed and a present discounted value of the net benefits may be unnecessary as well. However, the absence of these values should be explicit in the analysis. An alternative to providing no present discounted value is to conduct a single year estimate of costs and benefits, but calculate a present discounted value of net benefits assuming an infinite time period.

Detail all aspects of the baseline specification that are uncertain. Because the analyst does not have perfect foresight, the appropriate baseline conditions cannot be characterized with certainty. Future values always have some level of uncertainty associated with them, and current values often do as well. To the extent possible, estimates of current values should be based on actual data, and estimates of future values should be based on clearly specified models and assumptions. Where reliable projections of future economic activity and demographics are available, this information should be adequately referenced. In general, uncertainties underlying the baseline conditions should be treated in the same way as other types of uncertainties in the analysis. All assumptions should be clearly stated and, where possible, all models should be independently reproducible.

It is important to detail information that was not included in the analysis due to scientific uncertainty. For example, a health or ecological effect may be related to the regulated pollutant, but the science behind this connection may be too uncertain to include the effect in the quantitative analysis. In this case, the effect should not be included in the baseline, but a discussion of why the effect was excluded should be added — especially if the magnitude is such that it could significantly affect the net benefit calculation. A similar recommendation can be made for model choice or even the choice of parameter values; known aspects of the analysis, which are not included in the baseline due to scientific uncertainty, should be included in the uncertainty section.

Large uncertainty in significant variables may require the construction of alternative baselines or policy scenarios. This leads to numerous complications in policy analysis, especially in cost-effectiveness analysis (CEA) and the calculation of net benefits. While sensitivity analysis is usually a better choice, multiple scenarios may be beneficial in selecting policy options, especially if there is a significant probability of irreversible consequences or catastrophic events.

Use the baseline assumptions consistently for all analyses for this regulation. The models, assumptions, and estimated parameters used in the baseline should be carried through for all components of the analysis. For example, the calculation of both costs and benefits should draw upon estimates derived using the same underlying assumptions of current and future economic conditions. If the benefits and costs are derived from two different models, then the initial baseline conditions of costs and benefits should be compared to ensure that they are making identical assumptions. Likewise, when comparing and ranking alternative regulatory options, comparison to the same baseline should be used for all options under consideration.[2]

In some cases, an analysis may not have been anticipated during the baseline specification. For example, a sector might be singled out for more detailed analysis, or a follow-on analysis might be needed to assess impacts on a particular low-income or minority group. In this case, a complete baseline specification that would make this secondary analysis fully consistent with the primary analyses may not be available. Even in

2 In the less common case in which more than one baseline scenario is modeled, 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, the presentation of economic information should clearly describe and refer to the specific baseline scenario being used.

this case, however, some type of baseline will have to be produced in order to conduct the analysis. While it may not be identical to the baseline used to analyze the benefits and costs, the analyst should endeavor to make it as similar as possible. The analyst should explicitly state the differences between the two baselines or any uncertainty associated with the secondary baseline.

5.3 Changes in Basic Variables

Certain variables are very important for modeling both the baseline scenario and the policy scenario. Some of these variables, such as population and economic activity, are commonly modeled by other government agencies and are available for use in economic analyses. The values of these variables will change over the period of study and, as a result of the policy, may differ significantly between the two scenarios. Even when they are the same across scenarios, these values can have a substantial impact on the overall benefits and costs and should be explicitly reported over time. Other variables, such as consumer spending patterns and technological growth in an industry, are also important for modeling, but are more difficult to estimate. In these cases, the analyst should specify the variable levels and report whether these variables changed during the period of the study. When they are assumed to change, both over time and between scenarios, the analyst should explicitly state the assumptions of how and why they change.

5.3.1 Demographic Change

Changes in the size and distribution of the population can affect the impact of EPA programs and, as a consequence, can be important in economic analyses. For example, risk assessments of air toxics standards require assumptions about the number of individuals exposed. Therefore, assumptions about future population distributions are important for measuring potential future incidence reductions and for estimating the maximum individual risk or exposures. Another example is when population growth affects the level of vehicle emissions due to an increased number of cars and greater highway congestion. For most analyses, U.S. Census Bureau projections of future population growth and distribution can be used. In some cases, however, behavioral models may be required if the population growth or distribution changes as a consequence of the regulation. For example, demographic trends in an area may change as a result of cleaning up hazardous waste sites. EPA analyses should reflect the consequences of population growth and migration, especially if these factors influence the regulatory costs and benefits.

5.3.2 Future Economic Activity

Future economic activity can have a significant effect on regulatory costs and benefits because it is correlated with emissions and, in some cases, can influence the feasibility or cost-effectiveness of particular control strategies. Even small changes in the rate of economic growth may, over time, result in considerable differences in emissions and control costs. Assuming no change in the economic activity of the regulated sector, or in the nation as a whole, will likely lead to incorrect results. For example, if the regulated industry is in significant decline, or is rapidly moving overseas, this information should be accounted for in the baseline. In such a case, incremental costs to the regulated community (and corresponding benefits from the regulation) are likely to be less than if the targeted industry were growing.

Official government estimates of future economic growth are the most appropriate values to use. In many cases, however, the future economic activity of the particular sectors under regulation will have to be modeled. In both cases, the models and assumptions used should be made as explicit as possible. When economic growth is a significant determinant of the relative merits of regulatory alternatives or when there are significant differences between official and private growth estimates, then sensitivity analyses using alternative growth estimates should be included.

5.3.3 Changes in Consumer Behavior

The bundle of economic goods purchased by consumers can affect the benefits and costs of a rule. An increase in the price and decrease in the quantity of goods from the regulated sector should be included as part of the cost of the regulation. Likewise, a reduction in the number of goods (e.g., bottled water) that were previously purchased to reduce health effects caused by the regulated pollutant will result in economic benefits to the public. Thus, changes in consumer behavior are important in the overall economic analysis. Changes in consumer purchasing behavior should be supported by estimates of demand, cross-price, and income elasticities allowing changes in consumer behavior to be estimated over time and for the baseline and policy scenarios.[3]

One controversial extension involves the income elasticity for environmental protection. There is some evidence that the demand for environmental quality rises with income (Baumol and Oates 1988). However, this does not necessarily justify adjusting the benefit of environmental improvements upward as income rises. This is because the willingness to pay (WTP) for a marginal improvement in the environmental amenity, the appropriate measure of the benefits of environmental protection, may not necessarily have a positive income elasticity (Flores and Carson 1997). It is appropriate to account for income growth over time where there are empirical estimates of income elasticity for a particular commodity associated with environmental improvements (e.g., for reduced mortality risk). In the absence of specific estimates, it would be appropriate to acknowledge and explain the potential increase in demand for environmental amenities, as incomes rise.

5.3.4 Technological Change

Future changes in production techniques or pollution control may influence both the baseline and the costs and benefits of regulatory alternatives. Estimating the future technological change is quite difficult and often controversial. Technological change can be thought of as having at least two components: true technological innovation, such as a new pollution control method; and learning effects, in which experience leads to cost savings through improvements in operations, experience, or similar factors. It is not advisable to assume a constant, generic rate of technological progress, even if the rate is small, simply because the continuous compounding of this rate over time can lead to implausible rates of technological innovation. However, in some cases learning effects may be included in analyses.

Undiscovered technological innovation is often considered to be one reason why regulatory costs are overstated (Harrington et al. 1999). Because of the difficulty and controversy associated with estimating technological change in an economic analysis, analysts should be careful to avoid the perception of bias when introducing it. If technological change is introduced in the cost analysis, then it should be introduced in the benefits analysis as well. While technological innovation in the regulated sector can reduce the cost of compliance, technological innovation in other sectors can reduce the benefits of the regulation. For example, the cost of controlling CFCs has declined over time due to technological improvements. However, innovation in mitigating factors, such as improvements in skin cancer treatments and efficacy of sunscreen lotions — both of which decrease the benefits of the regulation — have also occurred. Further, the analysis should include the costs associated with research and development (R&D) for the innovations to correctly value cost-reducing technological innovation, but only if the costs are policy-induced and do not arise from planned R&D budgets. This distinction is sometimes difficult to make.

If technological innovation is included in the policy scenario, then it should be included in the baseline as well (see Text Box 5.1). While accepting that innovation will occur in the baseline and policy scenarios, rates across scenarios may differ because regulation may cause firms to innovate more to reduce the cost of compliance. In cases where small changes in technology could

3 Demand elasticities show how the quantity of a product purchased changes as its prices changes, all else equal. Cross-price elasticities show how a change in the price of one good can result in a change in the price of another good (either a substitute or a complement), thereby altering the quantity purchased. Income elasticity allows a modeler to forecast how much more of a good consumers will buy when their income increases. See Appendix A for more information on elasticity.

Text Box 5.1 - Technological Change, Induced Innovation, and the Porter Hypothesis

There are many proposed mechanisms by which environmental regulation could cause technological change. One mechanism is by induced innovation: the induced innovation hypothesis states that as the relative prices of factors of production change, the relative rate of innovation for the more expensive factor will also increase. This idea is well accepted; for example, Newell et al. (1999) found that a considerable amount of the increase in energy efficiency over the last few decades has been caused by the increase in the relative price of energy over that time.

A similar idea has also been described (somewhat less formally) as the "Porter Hypothesis" (Porter and van der Linde 1995, and Heyes and Liston-Heyes 1999). Jaffe and Palmer (1997) delineate three versions of the hypothesis: weak, narrow, and strong.

The weak version of the hypothesis assumes that an environmental regulation will stimulate innovation but it does not predict the magnitude of these innovations or the resulting cost savings. This version of the hypothesis is very similar to the induced innovation hypothesis. The narrow version of the hypothesis predicts that flexible regulation (e.g., incentive-based) will induce more innovation than inflexible regulation and vice versa. There is empirical evidence that this is the case (Kerr and Newell 2003, and Popp 2003). Analysts may be able to estimate the rate of change of innovation under the weak or narrow version of the hypothesis, or under induced innovation. However, this innovation may crowd out other forms of innovation.

The strong version predicts cost savings from environmental regulation under the assumption that firms do not maximize cost saving without pressure to do so. While anecdotal evidence of this phenomenon may exist, the available economic literature has found no statistical evidence supporting it as a general claim (Jaffe et al. 1995; Palmer, Oates, and Portney 1995; Jaffe and Palmer 1997; and Brännlund and Lundgren 2009). The strong version of the Porter Hypothesis may be true in some cases, but it requires special assumptions and an environmental regulation combined with other market imperfections (such as bounded rationality) that are difficult to generalize. Analysts should not assume cost savings from a regulation based on the strong version of the Porter Hypothesis.

dramatically affect the costs and benefits, or where technological change is reasonably anticipated, the analyst should consider exploring these effects in a sensitivity analysis. This might include probabilities associated with specific technological changes or adoption rates of a new technology, or it may be an analysis of the rate required to alter the policy decision. Such an analysis should show the policy significance of emerging technologies that have already been accepted, or are, at a minimum, in development or reasonably anticipated.

In some cases it may be possible to make the case that learning effects will lead to lower costs over time.[4] Estimated rates of learning effects often indicate that costs decline by approximately 5 percent to 10 percent for every doubling of cumulative production. If learning effects are to be included in an analysis, the analyst should carefully examine the existing data for relevance to the problem at hand. Estimated learning effects can vary according to many factors, including across industries and by the length of the time period considered. Also, because estimates of learning effects are based on doubling of cumulative production, inclusion of learning effects will have a greater influence on rules with longer time periods and may have little effect on rules with short time periods.

5.4 Compliance Rates

One aspect of baseline specification that is particularly complex, and for which assumptions are typically necessary, is the setting of compliance rates. The treatment of compliance in the baseline scenario can significantly affect the results of the

4 See U.S. EPA (1997b, 2007b).

analysis. It is important to separate the changes associated with a new regulation from actions taken to meet existing requirements. If a proposed regulation is expected to increase compliance with a previous rule, the correct measure of the costs and benefits generally excludes impacts associated with the increased compliance.[5] This is because the costs and benefits of the previous rule were presumably estimated in the economic analysis for that rule, and should not be counted again for the proposed rule. This is of particular importance if compliance and enforcement actions taken to meet existing requirements are coincident with, but not caused by, changes introduced by the new regulation.

Assumptions about compliance behavior for current and new requirements should be clearly presented in the description of the analytic approach used for the analysis. When comparing regulatory options on the basis of their social costs and benefits, the effect of compliance assumptions on the estimated economic impacts should be described, along with the sensitivity of the results to these assumptions.

In most cases, a full compliance scenario should be analyzed. If a baseline is used that assumes a scenario other than full compliance, the analyst should take care to explain the compliance assumption for the current regulation under consideration. The Agency is unlikely to propose a rule that it believes will not be followed, but if there is widespread non-compliance with previous rules then this suggests a persistent problem.

5.4.1 Full Compliance

As a general rule, when preparing analyses of regulations **analysts should develop baseline and policy scenarios that assume full compliance with existing and newly enacted (but not yet implemented) regulations**. Assuming full compliance with existing regulations enables the analysis to focus on the incremental economic effects of the new rule or policy without double counting benefits and costs captured by analyses performed for other rules.

Assuming full compliance with all previous regulations when current observed or reported economic behavior indicate otherwise may pose some challenges to the analyst. 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 apparent in the convictions of violators and negotiated settlements conducted by EPA.

As a practical matter, before rejecting full compliance assumptions for existing policies, the emissions from noncompliant firms should be known, estimable, and occurring at a rate that can affect the evaluation of policy options. In some cases, two baselines may have to be assumed: one assuming full compliance with existing regulation and a separate "current practice" baseline. In the case of a deregulatory rule, which is 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 make sense to perform the analysis using both baselines. 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.

5.4.2 Under-Compliance

When compliance issues are important and there is sufficient monitoring data to support the analysis, a "current practice" baseline can be used. A "current practice" baseline is established using the actual degree of compliance rather than assumed full compliance. Current practice baselines are useful for actions intended to address or "fix-up" compliance problems associated with existing policies. In these cases, assuming a full-compliance baseline that disregards under-compliant behavior

5 An exception would be if the proposed regulation were designed to correct the under-compliance from the previous rule. This is discussed in Section 5.4.2.

could obscure the value of investigating additional or alternative regulatory actions. This was the case in a review of the banning of lead from gasoline, which was precipitated, in part, by the noncompliance of consumers who put leaded gasoline in vehicles that required non-leaded fuel to protect their catalytic converters, resulting in increased vehicle emissions (U.S. EPA 1985).

If under-compliance is assumed in the baseline, then the nature of that non-compliance becomes important. For example, in a case where under-compliance occurs uniformly (or at random) across an industry, then changing the compliance rate assumption will not affect the benefit-cost ratio nor the sign of net benefits, assuming the effect on ambient concentrations is also uniform (or random), although it will affect the magnitude of net benefits. In other words, a proposed regulation that can be justified from a net benefit perspective under full compliance can also be justified under any baseline compliance rate. However, if non-compliance with previous regulation occurs selectively when compliance costs are high, then the benefit-cost ratio will decline as higher rates of compliance are assumed, and net benefits could potentially switch from positive to negative for a proposed regulation. This occurs because the cost per unit of benefit will continue to increase as full compliance is reached. Analysts may elect to incorporate predicted differences in compliance rates within policy options in cases where compliance behavior is known to vary systematically.

While a baseline assuming under-compliance can be useful in some cases, it should be executed carefully or the issue should be examined with a sensitivity analysis. A partial compliance baseline has the potential for double counting both benefits and costs. A sequence of emissions tightening rules could be justified by repeatedly factoring under-compliance into the baseline, while assuming that entities will fully comply with the new rule under consideration. Summing the benefits from the total sequence of rules would overstate benefits because each rule claims part of the same benefits each time. Additionally, while the benefits flowing from previous regulations may not have been realized due to lack of compliance, the full costs of their implementation may not have been realized either. The additional costs associated with coming into compliance should also be included to avoid producing inflated net benefits. In the case where an under-compliance baseline (or sensitivity analysis) is justified, care should be taken to explain these potential biases.

5.4.3 Over-Compliance

Over-compliance may occur due to risk aversion, technological lumpiness, uncertainty in pollution levels, or other behavioral responses. Here the benefits (and potentially the costs) of the previous regulation have been understated rather than overstated. In this case, as with under-compliance, true societal net benefits of a regulation will not be calculated correctly under an assumption of full compliance.

In cases of over-compliance with existing policies, current practices can be used to define baseline conditions unless these practices are expected to change. For example, over-compliance may be the result of choices made in anticipation of more stringent regulations. If these stringent regulations are not implemented, the analyst will need to establish whether over-compliance will be reduced to meet the relatively less stringent requirements. If the regulated entities are expected to continue to over-comply despite the absence of the more stringent regulation, then the costs and benefits attributable to this behavior are not related to the policy under consideration. In this case, it would be appropriate to account for the over-compliance in the baseline scenario that describes the "world without the regulation." However, if the regulated entities are expected to relax their pollution control practices to meet relatively less stringent requirements, then the costs and benefits of the over-compliance behavior should be attributed to the new policy scenario, and over-compliance should not be included in the baseline. In these situations, it may be useful to consider performing a sensitivity analysis to demonstrate the potential economic consequences of different assumptions associated with the expected changes in behavior.

5.5 Multiple Rules

Although regulations that have been finalized clearly belong in the baseline of a proposed rule, the baseline specification may be complicated if other regulations in addition to the one being implemented are under consideration or nearing completion. In this case it becomes difficult to determine which regulations are responsible for the environmental improvements and can "take credit" for reductions in risks. It is also necessary to determine how these other regulations affect market conditions that directly influence the costs or the benefits associated with the policy of interest. This is true not only for multiple rules promulgated by EPA, but also for rules passed by other federal, state, and local agencies. In addition to agencies that regulate environmental behavior, other agencies that regulate consumer and industrial behavior [e.g., Occupational Safety and Health Administration (OSHA), Department of Transportation (DOT), and Department of Energy (DOE)] develop rules that may overlap with upcoming EPA regulations. Even the *potential* implementation of another such rule may affect the benefits and costs of an EPA regulation being analyzed, due to the strategic behavior of regulated entities. Therefore, it is important to consider the impact of other rules when establishing a baseline. If another federal, state, or local agency is legally required to impose a regulation but is still in the process of finalizing that regulation, then a baseline which includes this impending regulation should be considered. The intent of the baseline is always to characterize the world in the absence of regulation being analyzed.

5.5.1 Linked Rules

In some cases it is possible to consider multiple rules together as a set. For example, some regulatory actions have linked together rules that affect the same industrial category. This was true of the pulp and paper effluent guidelines and National Emissions Standards for Hazardous Air Pollutants (NESHAP) rules (U.S. EPA 1997c). In other cases, multiple rules may not necessarily be a set of similar policies associated with the same industry, but rather are a set of different policies that are all necessary to achieve a policy objective. For example, EPA may issue effluent limitation guidelines (ELG) to provide technical requirements for a type of pollution discharge, and may then issue a complementary National Pollution Discharge Elimination System (NPDES) rule, providing details of the permitting system. Since ELG and NPDES work together to achieve one objective it would not make sense to analyze them separately.

The optimal solution in both of the cases described above is to include all of the rules in the same economic analysis. In this case, the multiple rules are analyzed as if they were one rule and the baseline specification simplifies to one with none of the rules included. While statutory requirements and judicial deadlines can inhibit promulgating multiple rules as one, coordination between rulemaking groups is still possible. The sharing of data, models, and joint decisions on analytic approaches may make a unified baseline possible so that the total costs and benefits resulting from the package of policies can be assessed.

5.5.2 Unlinked Rules

In some cases, it is simply not feasible to analyze a collection of overlapping rules together in a single economic analysis with a single baseline. This may be true for rules originating from different program offices or different regulatory agencies, or when the timing of the various rules is not clear. In this case, each rule should be analyzed separately with its own baseline, but the order in which the rules are analyzed may have a substantial effect on the outcome of a BCA. For example, in 2005, EPA promulgated both the Clean Air Interstate Rule (CAIR) and the Clean Air Mercury Rule (CAMR) to reduce pollution from coal fired power plants. While the primary purpose of CAIR was to reduce sulfur dioxide (SO_2) and nitrogen oxides (NO_x), the control technologies necessary to achieve this also reduced mercury emissions. Because the CAMR analysis assumed that CAIR had been implemented and was, therefore, in the baseline, the estimated incremental reduction in mercury from CAMR was much smaller than if CAIR had not been included in the baseline. In a similar fashion, if some of the costs of fully complying with the second rule are incurred in the process of complying with the first rule, then these costs are part of the baseline and are not considered as costs of the second rule. In general, only the incremental benefits and costs of the second rule should be included if the first rule is in the baseline.

The practical assumption commonly made when rules cannot be linked together is 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. However, this may not always be possible. For example, a rule may be phased in over time, complicating the analysis of a new rule going into effect during that same period. In that case, the baseline for the new rule should include the timing of each stage of the phased rule and its resulting environmental, health, and economic changes.

In the absence of some orderly sequence of events that allows the attribution of changes in behavior to a unique regulatory source, there is no non-arbitrary way to allocate the costs and benefits of a package of overlapping policies to each individual policy. That is, there is no theoretically correct order for conducting a sequential analysis of multiple overlapping policies that are promulgated simultaneously. The only solution in this case is to make a reasonable assumption and clearly explain it, detailing which rules are included in the baseline (see Text Box 5.2). If the costs and benefits from these rules are small, then this may be all that is necessary. It may not be worth additional time and resources to reconcile the overlapping rules. On the other hand, for major rules or if the number of overlapping rules is small, then a sensitivity analyses can be included to test for the implications of including or omitting other regulations. Under this sensitivity analysis, it may also be possible to use the overlapping nature of the regulations to allow for some regulatory flexibility in compliance dates and regulatory requirements.

5.5.3 Indirectly Related Policies and Programs

In some instances, less directly related environmental policies or programs can influence the baseline. For example, potential changes in farm subsidy programs may significantly influence future patterns of pesticide use. In an ideal analysis, all of the potential direct and indirect influences on baseline conditions (and on the costs and benefits of regulatory alternatives) would be examined and estimated. In other words, this situation can be handled in the same way as unlinked overlapping rules described above. Practically speaking, however, it is up to the analyst to determine if these indirect influences are important enough to incorporate into the regulatory analysis. If indirect influences are known but are not considered to be significant enough to be included in the quantitative analysis, they can be discussed qualitatively.

5.6 Partial Benefits to a Threshold

Some benefits only occur after a threshold has been reached. For example, the benefits associated with improving a stream to allow for recreational swimming are realized only when all of the pollutants have been reduced enough to allow for primary contact and an enjoyable swimming experience. Likewise, valued species populations may only recover when multiple limiting factors are addressed. However, a particular benefits threshold may not be met with a single rule. In such cases, associating the benefits only with the rule that actually passes the threshold could make it impossible to justify the incremental progress (via previous rules). It is generally reasonable to account for the benefits of making progress toward a goal, even if the threshold is not met in the rule under consideration.

For example, EPA's Office of Water has calculated the benefits associated with improving river miles for various designated uses (e.g., swimming, fishing, and boating) in a number of rules. In each case, some river miles were improved for the designated use, while other miles were improved, but not enough to change their designated use. Earlier rules claimed benefits only if a river mile actually changed its designation, implicitly giving a value of zero to partially improved river miles. More recent regulation claims partial benefit for incremental improvements toward the threshold. Neither approach is necessarily correct, but accounting for the benefits of partial gains provides better information to decision makers and the public and allows the Agency to justify incremental

Text Box 5.2 - Sequencing Unlinked Rules

It is impossible to identify all of the possible scenarios one might need to consider when determining which rules to include in a baseline, but a few illustrative cases are provided below.

Including final rules that have not yet taken effect: This is the most straightforward case. All final rules promulgated prior to the rule under consideration should be included in the baseline. The costs and benefits of the regulation under consideration must be evaluated against a baseline that assumes firms will comply with these promulgated rules. For example, on March 15, 2005, EPA issued the Clean Air Mercury Rule (CAMR) to reduce mercury emissions from coal-fired power plants. Five days earlier, on March 10, 2005, EPA finalized the Clean Air Interstate Rule (CAIR) to reduce sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions from coal-fired power plants. Because the control technology assumed under CAIR included some mercury reductions, the baseline used for CAMR included the actions that firms would need to take to comply with CAIR.

Including rules anticipated to occur after a regulation is promulgated but before it takes effect: This is a more difficult case and only applies to regulations that have a long lag between the date on which they are issued and the date when they take effect. The longer the difference between these two dates, the more important it is to include rules that can be expected in the interim. For example, National Ambient Air Quality Standards (NAAQS) can have a number of years between the date on which a standard is announced and the date on which designations of attainment or nonattainment are made. In this case, if another rule is imminent and will take effect prior to the effective date of the new NAAQS, then it should be included in the baseline for the NAAQS. It is important, however, that the analyst not simply speculate that another rule will be implemented. Any other rule included in the baseline, other than those already promulgated, should be imminent or reasonably anticipated with a high degree of certainty. In addition, the analyst should be clear as to what assumptions have been made.

Including state rules that are legally required but not yet implemented: This is probably the most difficult case. Actions by state (and even local) governments can affect the costs and benefits of federal rules, particularly if they are regulating the same sector or pollutant. As with the case above, any state regulation that has been finalized should be included in the baseline. The more difficult case occurs when the state has a legal obligation to implement a regulation but either has not done so or is in the process of doing so. In this case, the analyst must use professional judgment to determine what would happen in the absence of EPA action. If the state would implement the regulation in the absence of EPA action, then a reasonable case can be made that this state regulation should be included in the baseline.

Two of the most important things to remember when sequencing multiple unlinked rules are transparency and objective reasoning. Transparency requires that the analyst clearly state all assumptions. Objective reasoning requires that the analyst not engage in speculation. If there is uncertainty about the anticipated rules, then two baselines, one with anticipated rules and one without, should be considered. If resources are constrained and only one baseline can be considered, then it should be constructed using only final rules and those that are reasonably expected with a high degree of certainty in the absence of EPA action.

progress to a threshold.[6] Note that once partial gains to a threshold have been claimed, there is a danger of double counting when evaluating the potential benefits of future rules. If partial gains have been valued in one rule, then subsequent rules cannot claim full credit for crossing the threshold. In effect, some of the benefits have already been used to justify the previous incremental rules and therefore claiming full credit in future rules would double count those benefits.

6 Sometimes calculating partial benefits to a threshold may not be a satisfactory solution, either because the progress to a threshold is uncertain due to multiple limiting factors (e.g., in some ecological improvements) or because it does not comport with the economic values (e.g., the value of avoiding the extinction of a species). In this case, a rulemaking incremental progress to the threshold might have to be justified on something other than a benefit-cost test. This, however, does not affect the choice of a baseline.

While the actual valuation of incremental progress is a benefits issue, the specification of that portion of the benefits that have been claimed in previous rules is a baseline issue. If previous rules have claimed partial benefits, the benefits available for the current rule should be clearly identified in the baseline specification. In the simplest case, this means calculating benefits in the same way as previous rules. However, this approach is not always possible, or even reasonable. New valuation studies or new models of ambient pollution may make the previous benefits estimates obsolete. In this more complicated case, the baseline specification should be developed so that the current benefits estimates can be compared with the previous estimates while avoiding double counting.

5.7 Behavioral Responses

To measure a policy's costs and benefits, it is important to clearly characterize the behavior of firms and individuals in both the baseline and the policy scenarios. Behavior is contrasted with the baseline and is often anticipated to change in response to the policy options. Some policies are prescriptive in specifying what actions are required — for example, mandating the use of a specific type of pollution control equipment. Responses to less-direct performance standards, such as bans on the production or use of certain products or processes or market-based incentive programs are somewhat more difficult to predict and commonly require some underlying model of economic behavior. Estimating responses is often difficult for pollution prevention policies because these options are more site- and process-specific when compared to end-of-pipe control technologies. Predicting the costs and environmental effects of these rules may require detailed information on industrial processes.

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 the costs and benefits of changed behavior in the analysis of the policy action, and not subsume them into the baseline scenario. Nevertheless, the dynamic aspects of market and consumer behavior, and the many motivations leading to change, can make it difficult to attribute economic costs and benefits to specific regulatory actions. Where behavioral changes are uncertain, an uncertainty analysis using various behavioral assumptions can provide insight into how important these assumptions may be.

Behavioral responses are usually characterized as reactions to proposed policy options. However, the behavioral assumptions used in the baseline, when no regulatory action is taken, are also very important. Individuals may attempt to mitigate the affect of pollution (e.g., by buying bottled water, using masks, or purchasing medication), or prevent their exposure altogether through some type of averting behavior (e.g., keeping windows closed or relocating). Careful consideration of this behavior is important to correctly measure the costs and benefits of regulation. Analysts should make explicit all assumptions about firm and individual behavioral in both the baseline and policy scenarios so that a proper comparison between the two can be made.

5.7.1 Potential for Cost Savings

Predicting firm-level responses begins with a comprehensive list of possible response options. In addition to the possible compliance technologies (if the technology is not specified by the policy itself) or waste management methods, less obvious firm-level responses should be considered. These include changes in operations (e.g., input mixtures, re-use or recycling, and developing new markets for waste products) to avoid or reduce the need for new controls or the use of restricted materials, shutting down a production line or plant to avoid the investments required to achieve compliance, relocation of the firm, or even exiting the industry. The possibility of noncompliance should also be explored, including the use of lawsuits to delay the required investment. In general, affected parties are assumed to choose the option that minimizes their costs.

In some cases, compliance implies a reduction in costs from the baseline. In other words, choosing the least costly regulatory solution would provide cost savings to the firms. In this case, it is important to provide an analysis of why these cost-saving measures are not undertaken in the baseline. It is not always obvious why firms would actively choose to not undertake a change that results in cost savings. If firms will eventually voluntarily undertake these changes without the regulation, then the regulatory intervention cannot be credited with the cost savings.

One possibility is that firms may not adopt cost-saving measures because of market failures (e.g., informational asymmetries or transactions costs) and other circumstances. In these cases, regulation can motivate economically beneficial actions, but there should be a reasonable description of the market failure or circumstances that the regulation is correcting. A second possibility is that firms are actively choosing a higher cost option in order to reduce legal liabilities or to achieve compliance with other implemented or proposed rules. In this latter case, firms will continue to choose the higher cost solution in both the baseline and the policy scenario and the costs savings can only be achieved by relaxing the legal liability or eliminating the other rule. In other words, the additional costs of compliance in excess of a least-cost strategy would be attributed to these other causes, but the rule itself will not achieve the cost savings.

5.7.2 Voluntary Actions

Occasionally, polluting industries adopt voluntary measures to reduce emissions. This can be implemented through a formal, government-sponsored voluntary program or a firm or sector may independently adopt measures. Such voluntary measures are adopted for a variety of reasons, including public relations considerations, to avoid regulatory controls, or to gain access to incentives associated with joining a formal program. When this is the case, it is important to account for these voluntary actions in the baseline and to be explicit about the assumptions of firms' future actions.

Typically, the economic baseline should reflect current circumstances, which means that voluntary reductions in emissions should be included in the baseline assumptions. This is not always possible, however, as voluntary actions are often difficult to measure (Brouhle, Griffiths, and Wolverton 2005). In the case of data or resource limitations, analysts may be compelled to adopt a "current regulations" baseline, which effectively ignores these emission reductions.

For the policy scenario, analysts should generally not assume that the current trends in voluntary reductions will persist. If firms are required to reduce emissions below their current level, then it should be assumed that the firms would meet the new standard without over-complying. While firms that go beyond compliance are often "good actors" who will continue to make reductions beyond the regulatory threshold, there is no a priori reason to expect this without a formal model explaining the firms' motivation. If the regulatory threshold is set above the emissions of these "good actions" then it is important to hypothesize why the voluntary actions were taken in the first place. If firms were making voluntary reductions in anticipation of the regulation or to dissuade the Agency from passing the regulation, then the firm can probably be expected to increase emissions to the regulatory level. On the other hand, if firms were making the reduction for some other incentive that continues to be present after the regulation is passed, then the voluntary emissions level may remain unchanged.

In some cases, it may be appropriate to demonstrate the significance of voluntary actions in a sensitivity analysis. This might involve analyzing competing assumptions of voluntary behavior. In all cases, the potential impact of the regulation on formal voluntary programs should be discussed. If participation in voluntary programs was motivated by the threat of the proposed regulation, then that voluntary program will likely be affected. In the extreme case, the voluntary program may be curtailed or eliminated as a consequence of the regulation. These potential implications should be included in the economic analysis.

5.8 Conclusion

Developing a baseline plays a critical role in analyzing policy scenarios, because it is the basis for BCA and option selection. However, developing a baseline is not a straightforward process, and analysts must make many decisions on the basis of professional judgment.

As stated in this chapter, a well-specified baseline should address exogenous changes in the economy, industry compliance rates, other concurrent regulations, and behavioral responses. The assumptions used in the baseline will be derived from models, published literature, or government agencies and should be clearly referenced. In cases where the data are uncertain, or not easily quantified, but may have a significant influence on the results, the analyst should describe the weaknesses in the data and assumptions, and include some type of sensitivity analysis. In some cases, multiple baselines or alternative scenarios may be required.

Chapter 6

Discounting Future Benefits and Costs

Discounting renders benefits and costs that occur in different time periods comparable by expressing their values in present terms. In practice, it is accomplished by multiplying the changes in future consumption (broadly defined, including market and non-market goods and services) caused by a policy by a discount factor. At a summary level, discounting reflects that people prefer consumption today to future consumption, and that invested capital is productive and provides greater consumption in the future. Properly applied, discounting can tell us how much future benefits and costs are worth today.

Social discounting, the type of discounting discussed in this chapter, is discounting from the broad society-as-a-whole point of view that is embodied in benefit-cost analysis (BCA). *Private discounting*, on the other hand, is discounting from the specific, limited perspective of private individuals or firms. Implementing this distinction can be complex but it is an important distinction to maintain because using a given private discount rate instead of a social discount rate can bias results as part of a BCA.

This chapter addresses discounting over the relatively short term, what has become known as *intragenerational discounting,* as well as discounting over much longer time horizons, or *intergenerational* discounting. Intragenerational, or *conventional*, discounting applies to contexts that may have decades-long time frames, but do not explicitly confront impacts on unborn generations that may be beyond the private planning horizon of the current ones. Intergenerational discounting, by contrast, addresses extremely long time horizons and the impacts and preferences of generations to come. To some extent this distinction is a convenience as there is no discrete point at which one moves from one context to another. However, the relative importance of various issues can change as the time horizon lengthens.

Several sensitive issues surround the choice of discount rate. This chapter attempts to address those most important for applied policy analysis. In addition to the sensitivity of the discount rate to the choice of discounting approach, a topic discussed throughout this chapter, these issues include: the distinction and potential confounding of efficiency and equity considerations (Section 6.3.2.1); the difference between consumption and utility discount rates (Sections 6.2.2.2 and 6.3.1); "prescriptive" vs. "descriptive" approaches to discount rate selection (Section 6.3.1); and uncertainty about future economic growth and other conditions (Sections 6.3.2.1 and 6.3.2.2).

6.1 The Mechanics of Summarizing Present and Future Costs and Benefits

Discounting reflects: (1) the amount of time between the present and the point at which these changes occur; (2) the rate at which consumption is expected to change over time in the absence of the policy; (3) the rate at which the marginal value of consumption diminishes with increased consumption; and (4) the rate at which the future utility from consumption is discounted with time. Changes in these components or uncertainty about them can lead to a discount rate that changes over time, but for many analyses it may be sufficient to apply a fixed discount rate or rates without explicit consideration of the constituent components or uncertainty.[1]

There are several methods for discounting future values to the present, the most common of which involve estimating *net present values* and *annualized values*. An alternative is to estimate a *net future value*.

6.1.1 Net Present Value (NPV)

The NPV of a projected stream of current and future benefits and costs relative to the analytic baseline is estimated by multiplying the benefits and costs in each year by a time-dependent weight, or discount factor, d, and adding all of the weighted values as shown in the following equation:

$$NPV = NB_0 + d_1 NB_1 + d_2 NB_2 + \dots + d_{n-1} NB_{n-1} + d_n NB_n \qquad (1)$$

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

$$d_t = \frac{1}{(1+r)^t} \qquad (2)$$

where r is the discount rate. The final period of the policy's future effects is designated as time n.

The NPV can be estimated using real or nominal benefits, costs, and discount rates. The analyst can estimate the present value of costs and benefits separately and then compare them to arrive at net present value.

It is important that the same discount rate be used for both benefits and costs because nearly any policy can be justified by choosing a sufficiently low discount rate for benefits, by choosing sufficiently high discount rates for costs, or by choosing a sufficiently long time horizon. Likewise, making sufficiently extreme opposite choices could result in any policy being rejected.

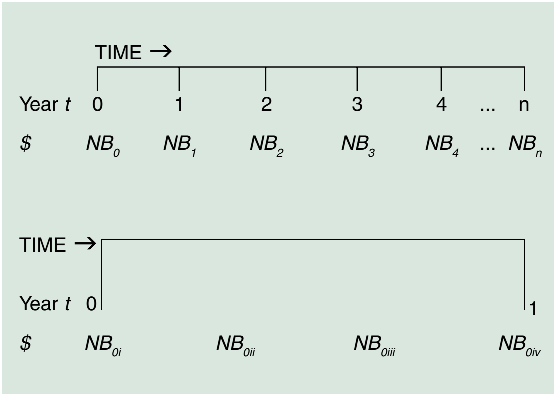
When estimating the NPV, it is also important to explicitly state how time periods are designated and when, within each time period, costs and benefits accrue. Typically time periods are years, but alternative time periods can be justified if costs or benefits accrue at irregular or non-annual intervals. The preceding formula assumes that t=0 designates the beginning of the first period. Therefore, the net benefits at time zero (NB_0) include a C_0 term that captures startup or one-time costs such as capital costs that occur immediately upon implementation of the policy. The formula further assumes that no additional costs are incurred until the end of the first year of regulatory compliance.[2] Any benefits also accrue at the end of each time period.

Figure 6.1 illustrates how net benefits (measured in dollars) are distributed over time. NB_1 is the sum of benefits and costs that may have been spread evenly across the four quarters of the first year (NB_{0i} through NB_{0iv}) as shown in the bottom part of the figure. There may be a loss of precision by "rounding" a policy's effects in a given year to the end or beginning of that year, but this is almost always extremely small in the scope of an entire economic analysis.

1 Note that accounting for changes in these components through discounting is distinct from accounting for inflation, although observed market rates reflect expected inflation. Both values (i.e., benefits and costs) and the discount rate should be adjusted for inflation; therefore most of the discussion in this chapter focuses on real discount rates and values.

2 See U.S. EPA (1995c) for an example in which operating and monitoring costs are assumed to be spread out evenly throughout each year of compliance. While the exponential function in equation (2) is the most accurate way of modeling the relationship between the 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 the distribution of monetary flows over time.

Figure 6.1 - Distribution of Net Benefits over Time



6.1.2 Annualized Values

An annualized value is the amount one would have to pay at the end of each time period t so that the sum of all payments *in present value terms* equals the original stream of values. Producing annualized values of costs and benefits is useful because it converts the time varying stream of values to a constant stream. 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. While the formulas below illustrate the estimation of annualized costs, the formulas are identical for benefits.[3]

To annualize costs, the present value of costs is calculated using the above formula for net benefits, except the stream of costs alone, not the net benefits, is used in the calculation. The exact equation for annualizing depends on whether or not there are any costs at time zero (i.e., at t=0).

Annualizing costs when there is no initial cost at t=0 is estimated using the following equation:

$$AC = PVC * \frac{r * (1+r)^n}{(1+r)^n - 1} \qquad (3)$$

where

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

PVC = present value of costs (estimated as in equation 1, above);

r = the discount rate per period; and

n = the duration of the policy.

Annualizing costs when there is initial cost at t=0 is estimated using the following slightly different equation:

$$AC = PVC * \frac{r * (1+r)^n}{(1+r)^{(n+1)} - 1} \qquad (4)$$

Note that the numerator is the same in both equations. The only difference is the "n+1" term in the denominator.

Annualization of costs is also useful when evaluating 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 or policy option 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.

As mentioned above, the same formulas would apply to estimating annualized benefits.

6.1.3 Net Future Value

Instead of discounting all future values to the present, it is possible to estimate value in some future time period, for example, at the end of the last year of the policy's effects, n. The net future value is estimated using the following equation:

$$NFV = d_0 NB_0 + d_1 NB_1 + d_2 NB_2 + \ldots + d_{n-1} NB_{n-1} + NB_n \qquad (5)$$

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

$$d_t = (1+r)^{(n-t)} \qquad (6)$$

3 Variants of these formulas may be common in specific contexts. See, for example, the Equivalent Uniform Annual Cost approach in EPA's *Air Pollution Control Cost Manual* (U.S. EPA 2002b).

where r is the discount rate. It should be noted that the net present value and net future value can be expressed relative to one another:

$$NPV = \frac{1}{(1+r)^n} \qquad (7)$$

6.1.4 Comparing the Methods

Each of the methods described above uses a discount factor to translate values across time, so the methods are not different ways to determine the benefits and costs of a policy, but rather are different ways to express and compare these costs and benefits in a consistent manner. NPV represents the present value of all costs and benefits, annualization represents the value as spread smoothly through time, and NFV represents their future value. For a given stream of net benefits, the NPV will be lower with higher discount rates, the NFV will be higher with higher discount rates, and the annualized value may be higher or lower depending on the length of time over which the values are annualized. Still, rankings among regulatory alternatives are unchanged across the methods.

Depending on the circumstances, one method might have certain advantages over the others. Discounting to the present to get a NPV 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. However, annualizing the costs of two machines with different service lives might reveal that the one with the higher total cost actually has a lower annual cost because of its longer lifetime.

Annualized values are sensitive to the annualization period; for any given present value the annualized value will be lower the longer the annualization period. Analysts should be careful when comparing annualized values from one analysis to those from another.

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 NPV estimates.

6.1.5 Sensitivity of Present Value Estimates to the Discount Rate

The impact of discounting streams of benefits and costs depends on the nature and timing of benefits and costs. The discount rate is not likely to affect the present value of the benefits and costs for those cases in which:

- All effects occur in the same period (discounting may be unnecessary or superfluous because net benefits are positive or negative regardless of the discount rate used);
- Costs and benefits are largely constant over the relevant time frame (discounting costs and benefits will produce the same conclusion as comparing a single year's costs and benefits); and/or
- Costs and benefits of a policy occur simultaneously and their relative values do not change over time (whether the NPV is positive does not depend on the discount rate, although the discount rate can affect the relative present value if a policy is compared to another policy).

Discounting can, however, substantially affect the NPV of costs and benefits when there is a significant difference in the timing of costs and benefits, such as with policies that require large initial outlays or that have long delays before benefits are realized. Many of EPA's policies fit these profiles. Text Box 6.1 illustrates a case in which discounting and the choice of the discount rate have a significant impact on a policy's NPV.

6.1.6 Some Issues in Application

There are several important analytic components that need to be considered when discounting: risk and valuation, placing effects in time, and the length of the analysis.

6.1.6.1 Risk and Valuation

There are two concepts that are often confounded when implementing social discounting, but should be treated separately. The first is the future value of environmental effects, which depends on many factors,

Text Box 6.1 - Potential Effects of Discounting

Suppose the benefits of a given program occur 30 years in the future and are valued (in real terms) at $5 billion at that time. The rate at which the $5 billion future benefits is discounted can dramatically alter the economic assessment of the policy: $5 billion 30 years in the future discounted at 1 percent is $3.71 billion, at 3 percent it is worth $2.06 billion, at 7 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 of difference in the present value of benefits. Longer time horizons will produce even more dramatic effects on a policy's NPV (see Section 6.3 on intergenerational discounting). For a given present value of costs, particularly the case where costs are incurred in the present and therefore not affected by the discount rate, it is easy to see that the choice of the discount rate can determine whether this policy is considered, on economic efficiency grounds, to offer society positive or negative net benefits.

including the availability of substitutes and the level of wealth in the future. The second is the role of risk in valuing benefits and costs. For both of these components, the process of determining their values and then translating the values into present terms are two conceptually distinct procedures. Incorporating the riskiness of future benefits and costs into the social discount rate not only imposes specific and generally unwarranted assumptions, but it can also hide important information from decision makers.

6.1.6.2 Placing Effects in Time

Placing effects properly in time is essential for NPV calculations to characterize efficiency outcomes. Analyses should account for implementation schedules and the resulting changes in emissions or environmental quality, including possible changes in behavior between the announcement of policy and compliance. Additionally, there may be a lag time between changes in environmental quality and a corresponding change in welfare. It is the change in welfare that defines economic value, and not the change in environmental quality itself. Enumerating the time path of welfare changes is essential for proper valuation and BCA.

6.1.6.3 Length of the Analysis

While there is little theoretical guidance on the time horizon of economic analyses, a guiding principle is that the time span should be sufficient to capture major welfare effects from policy alternatives. This principle is consistent with the underlying requirement that BCA reflect the welfare outcomes of those affected by the policy. Another way to view this is to consider that the time horizon, T, of an analysis should be chosen such that:

$$\sum_{t=T}^{\infty} (B_t - C_t)e^{-rt} \leq \varepsilon \,, \qquad (8)$$

where ε is a tolerable estimation error for the NPV of the policy. That is, the time horizon should be long enough that the net benefits for all future years (beyond the time horizon) are expected to be negligible when discounted to the present. In practice, however, it is not always obvious when this will occur because it may be unclear whether or when the policy will be renewed or retired by policy makers, whether or when the policy will become obsolete or "non-binding" due to exogenous technological changes, how long the capital investments or displacements caused by the policy will persist, etc.

As a practical matter, reasonable alternatives for the time span of the analysis may be based on assumptions regarding:

- The expected life of capital investments required by or expected from the policy;
- The point at which benefits and costs reach a steady state;
- Statutory or other requirements for the policy or the analysis; and/or
- The extent to which benefits and costs are separated by generations.

The choice should be explained and well-documented. In no case should the time horizon be arbitrary, and the analysis should highlight the extent to which the sign of net benefits or the relative rankings of policy alternatives are sensitive to the choice of time horizon.

6.2 Background and Rationales for Social Discounting

The analytical and ethical foundation of the social discounting literature rests on the traditional test of a "potential" Pareto improvement in social welfare; that is, the trade-off between the gains to those who benefit and the losses to those who bear the costs. This framework casts the consequences of government policies in terms of individuals contemplating changes in their own consumption (broadly defined) over time. Trade-offs (benefits and costs) in this context reflect the preferences of those affected by the policy, and the time dimension of those trade-offs should reflect the intertemporal preferences of those affected. Thus, social discounting should seek to mimic the discounting practices of the affected individuals.

The literature on discounting often uses a variety of terms and frameworks to describe identical or very similar key concepts. General themes throughout this literature are the relationship between consumption rates of interest and the rate of return on private capital, the need for a social rate of time preference for BCA, and the importance of considering the opportunity cost of foregone capital investments.

6.2.1 Consumption Rates of Interest and Private Rates of Return

In a perfect capital market with no distortions, the return to savings (the consumption rate of interest) equals the return on private sector investments. Therefore, if the government seeks to value costs and benefits in present day terms in the same way as the affected individuals, it should also discount using this single market rate of interest. In this kind of "first best" world, the market interest rate would be an unambiguous choice for the social discount rate.

Real-world complications, however, make the issue much more complex. Among other things, private sector returns are taxed (often at multiple levels), capital markets are not perfect, and capital investments often involve risks reflected in market interest rates. These factors drive a wedge between the *social rate* at which consumption can be traded through time (the pre-tax rate of return to private investments) and the rate at which *individuals* can trade consumption over time (the post-tax consumption rate of interest). Text Box 6.2 illustrates how these rates can differ.

A large body of economic literature analyzes the implications for social discounting of divergences between the social rate of return on private sector investment and the consumption rate of interest. Most of this literature is based on the evaluation of public projects, but many of the insights still apply to regulatory BCA. The dominant approaches in this literature are briefly outlined here. More complete recent reviews can be found in Spackman (2004) and Moore et al. (2004).

Text Box 6.2 - Social Rate and Consumption Rates of Interest

Suppose that the market rate of interest, net of inflation, is 5 percent, and that the taxes on capital income amount to 40 percent of the net return. In this case, private investments will yield 5 percent, of which 2 percent is paid in taxes to the government, with individuals receiving the remaining 3 percent. From a social perspective, consumption can be traded from the present to the future at a rate of 5 percent. But individuals effectively trade consumption through time at a rate of 3 percent because they owe taxes on investment earnings. As a result, the consumption rate of interest is 3 percent, which is substantially less than the 5 percent social rate of return on private sector investments (also known as the social opportunity cost of private capital).

6.2.2 Social Rate of Time Preference

The goal of social discounting is to compare benefits and costs that occur at different times based on the rate at which society is willing to make such trade-offs. If costs and benefits can be represented as changes in consumption profiles over time, then discounting should be based on the rate at which society is willing to postpone consumption today for consumption in the future. Thus, the rate at which society is willing to trade current for future consumption, or the social rate of time preference, is the appropriate discounting concept.

Generally a distinction is made between individual rates of time preference and that of society as a whole, which should inform public policy decisions. The individual rate of time preference includes factors such as the probability of death, whereas society can be presumed to have a longer planning horizon. 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. For these and other reasons, the social rate of time preference is not directly observable and may not equal any particular market rate.

6.2.2.1 Estimating a Social Rate of Time Preference Using Risk-Free Assets

One common approach to estimating the social rate of time preference is to approximate it from the market rate of interest from long-term, risk-free assets such as government bonds. The rationale behind this approach is that this market rate reflects how individuals discount future consumption, and government should value policy-related consumption changes as individuals do. In other words, the social rate of discount should equal the consumption rate of interest (i.e., an individual's marginal rate of time preference).

In principle, estimates of the consumption rate of interest could be based on either after-tax lending or borrowing rates. Because individuals may be in different marginal tax brackets, may have different levels of assets, and may have different opportunities to borrow and invest, the type of interest rate that best reflects marginal time preference will differ among individuals. 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.

The historical rate of return, post-tax and after inflation, is a useful measure because it is relatively risk-free, and BCA should address risk elsewhere in the analysis rather than through the interest rate. Also, because these are longer-term instruments, they provide more information on how individuals value future benefits over these kinds of time frames.

6.2.2.2 Estimating a Social Rate of Time Preference Using the 'Ramsey' Framework

A second option is to construct the social rate of time preference in a framework originally developed by Ramsey (1928) to reflect: (1) the value of additional consumption as income changes; and (2) a "pure rate of time preference" that weighs utility in one period directly against utility in a later period. These factors are combined in the equation:

$$r = \eta g + \rho \qquad (9)$$

where (r) is the market interest rate, the first term is the elasticity of marginal utility (η) times the consumption growth rate (g), and the second term is pure rate of time preference (ρ). Estimating a social rate of time preference in this framework requires information on each of these arguments, and while the first two of these factors can be derived from data, ρ is unobservable and must be determined.[4] A more detailed discussion of the Ramsey equation can be found in Section 6.3: Intergenerational Social Discounting.

4 The Science Advisory Board (SAB) Council defines discounting based on a Ramsey equation as the "demand-side" approach, noting that the value judgments required for the pure social rate of time preference make it an inherently subjective concept (U.S. EPA 2004c).

6.2.3 Social Opportunity Cost of Capital

The social opportunity cost of capital approach recognizes that funds for government projects, or those required to meet government regulations, have an opportunity cost in terms of foregone investments and therefore future consumption. When a regulation displaces private investments society loses the total pre-tax returns from those foregone investments. In these cases, ignoring such capital displacements and discounting costs and benefits using a consumption rate of interest (the post-tax rate of interest) does not capture the fact that society loses the higher, social (pre-tax) rate of return on foregone investments.

Private capital investments might be displaced if, for example, public projects are financed with government debt or regulated firms cannot pass through capital expenses, and the supply of investment capital is relatively fixed. The resulting demand pressure in the investment market will tend to raise interest rates and squeeze out private investments that would otherwise have been made.[5] Applicability of the social opportunity cost of capital depends upon full crowding out of private investments by environmental policies.

The social opportunity cost of capital can be estimated by the pre-tax marginal rate of return on private investments observed in the marketplace. There is some debate as to whether it is best to use only corporate debt, only equity (e.g., returns to stocks) or some combination of the two. In practice, average returns that are likely to be higher than the marginal return, are typically observed, given that firms will make the most profitable investments first; it is not clear how to estimate marginal returns. These rates also reflect risks faced in the private sector, which may not be relevant for public sector evaluation.

5 Another justification for using the social opportunity cost of capital 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. While it is true that social welfare will be improved if the government invests in projects that have higher values rather than lower ones, it does not follow that rates of return offered by these alternative projects define the level of the social discount rate. If individuals discount future benefits using the consumption rate of interest, the correct way to describe a project with a rate of return greater than the consumption rate is to say that it offers substantial present value net benefits.

6.2.4 Shadow Price of Capital Approach

Under the *shadow price of capital approach* costs are adjusted to reflect the social costs of altered private investments, but discounting for time itself is accomplished using the social rate of time preference that represents how society trades and values consumption over time.[6] The adjustment factor is referred to as the "shadow price of capital."[7] Many sources recognize this method as the preferred analytic approach to social discounting for public projects and policies.[8]

The shadow price, or social 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 that at which individuals discount them. 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 (generally considered to be consumption) for the government. Text Box 6.3 illustrates this idea of the shadow price of capital.

If compliance with environmental policies displaces private investments, the shadow price of capital approach suggests first adjusting the project or policy cost upward by the shadow price of capital, and then discounting all costs and benefits using a social rate of discount equal to the social rate of time preference. The most complete frameworks for the shadow price of capital also note that while the costs of regulation might displace private capital, the benefits could encourage additional private sector investments. In principle, a full analysis of shadow price of

6 Because the consumption rate of interest is often used as a proxy for the social rate of time preference, this method is sometimes known as the "consumption rate of interest – shadow price of capital" approach. However, as Lind (1982b) notes, what is really needed is the social rate of time preference, so more general terminology is used. Discounting based on the shadow price of capital is referred to as a "supply side" approach by EPA's SAB Council (U.S. EPA 2004c).

7 A "shadow price" can be viewed as a good's opportunity cost, which may not equal the market price. Lind (1982a) remains the seminal source for this approach in the social discounting literature.

8 See OMB *Circular A-4* (2003), Freeman (2003), and the report of EPA's Advisory Council on Clean Air Compliance Analysis (U.S. EPA 2004c).

Text Box 6.3 - Estimating and Applying the Shadow Price of Capital

To estimate the shadow price of capital, suppose that the consumption rate of interest is 3 percent, the pre-tax rate of return on private investments is 5 percent, the net-of-tax earnings from these investments are consumed in each period, and 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 under these conditions will produce a stream of private consumption of $.03 per year, and tax revenues of $.02 per year. Discounting the private post-tax stream of consumption at the 3 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 $.67. The social value of this $1 private investment – the shadow price of capital – is thus $1.67, which is substantially greater than the $1 private value that individuals place on it.

To apply this shadow price of capital *estimate*, we need additional information about debt and tax financing as well as about how investment and consumption are affected. Assume 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. 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 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; add to this the $.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 $.03, which is the present value of the future $5 benefit discounted at the 3 percent consumption rate of interest ($1.5328) minus the $1.5025 social cost.

capital adjustments would treat costs and benefits symmetrically in this sense.

The first step in applying this approach is determining whether private investment flows will be altered by a policy. 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 social rate of time preference. A simple illustration of this method applied to the costs of a public project and using the consumption rate of interest is shown in Text Box 6.3.[9]

9 An 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 social discount rate equal to a weighted average of the two. The weights would equal the proportions of project financing that displace private investment and consumption respectively. This approach has enjoyed considerable popularity over the years, but it is technically incorrect and can produce NPV results substantially different from the shadow price of capital approach. (For an example of these potential differences see Spackman 2004.)

6.2.4.1 Estimating the Shadow Price of Capital

The shadow price of capital approach is data intensive. It requires, among other things, estimates of the social rate of time preference, the social opportunity cost of capital, and estimates of the extent to which regulatory costs displace private investment and benefits stimulate it. While the first two components can be estimated as described earlier, information on regulatory effects on capital formation is more difficult. As a result empirical evidence for the shadow price of capital is less concrete, making the approach difficult to implement.[10]

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

10 Depending on the magnitudes of the various factors, shadow prices from about 1 to infinity can result (Lyon 1990). Lyon (1990) and Moore et al. (2004) contain excellent reviews of how to calculate the shadow price of capital and possible settings for the various parameters that determine its magnitude.

considered relative to the flow of investment capital from abroad.[11]

Some argue that early analyses implicitly assumed that capital flows into the nation were either nonexistent or very insensitive to interest rates, known as the "closed economy" assumption.[12] Some empirical evidence suggests, however, that international capital flows are quite large and are sensitive to interest rate changes. In this case, the supply of investment funds to the U.S. equity and debt markets may be highly elastic (the "open economy" assumption), thus private capital displacement would be much less important than previously thought.

Under this alternative view, it would be inappropriate to assume that financing a public project through borrowing would result in dollar-for-dollar crowding out of private investment. If there is 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 social rate of time preference alone. However, the literature to date is not conclusive on the degree of crowding out. There is little detailed empirical evidence as to the relationship between the nature and size of projects and capital displacement. While the approach is often recognized as being technically superior to simpler methods, it is difficult to implement in practice.

6.2.5 Evaluating the Alternatives

The empirical literature for choosing a social discount rate focuses largely on estimating the consumption rate of interest at which individuals translate consumption through time with reasonable certainty. Some researchers have explored other approaches that, while not detailed here, are described briefly in Text Box 6.4.

11 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.

12 See Lind (1990) for this revision of the shadow price of capital approach.

To estimate a consumption rate of interest that includes low risk, historical rates of return on "safe" assets (post-tax and after inflation), such as U.S. Treasury securities, are normally used. Some may use the rate of return to private savings. Recent studies and reports have generally found government borrowing rates in the range of around 2 percent to 4 percent.[13] Some studies have expanded this portfolio to include other bonds, stocks, and even housing. This generally raises the range of rates slightly. It should be noted that these rates are *realized* rates of return, not anticipated, and they are somewhat sensitive to the choice of time period and the class of assets considered.[14] Studies of the social discount rate for the United Kingdom place the consumption rate of interest at approximately 2 percent to 4 percent, with the balance of the evidence pointing toward the lower end of the range.[15]

Others have constructed a social rate of time preference by estimating the individual arguments in the Ramsey equation. These estimates necessarily require judgments about the pure rate of time preference. Moore et al. (2004) and Boardman et al. (2006) estimate the intragenerational rate to be 3.5 percent. Other studies base the pure rate of time preference on individual mortality risks in order to arrive at a discount rate estimate. As noted earlier, this may be useful for an individual, but is not generally appropriate from a societal standpoint. The Ramsey equation has been used more frequently in the context of intergenerational discounting, which is addressed in the next section.

13 OMB (2003) cites evidence of a 3.1 percent pre-tax rate for ten-year U.S. Treasury notes. According to the U.S. Congressional Budget Office (CBO) (2005), funds continuously reinvested in 10-year U.S. Treasury bonds from 1789 to the present would have earned an average inflation-adjusted return of slightly more than 3 percent a year. Boardman et al. (2006) suggest 3.71 percent as the real rate of return on ten-year U.S. Treasury notes. Newell and Pizer (2003) find rates slightly less than 4 percent for thirty-year U.S. Treasury securities. Nordhaus (2008) reports a real rate of return of 2.7 percent for twenty-year U.S. Treasury securities. The CBO estimates the cost of government borrowing to be 2 percent, a value used as the social discount rate in their analyses (U.S. CBO 1998).

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

15 Lind (1982b) offers some empirical estimates of the consumption rate of interest. Pearce and Ulph (1994) provide 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.

Text Box 6.4 - Alternative Social Discounting Perspectives

Some of the literature questions basic premises underlying the conventional social discounting analysis. For example, some studies of individual financial and other decision-making contexts suggest that even a single individual may appear to value and discount different actions, goods, and wealth components differently. This "mental accounts" or "self-control" view suggests that individuals may evaluate one type of future consequence differently from another type of future consequence. 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 the case if the future consequences in question are not tradable commodities. Some evidence from experimental economics indicates that discount rates appear to be lower the larger the magnitude of the underlying effect being valued. Experimental results have shown higher discount rates for gains than for losses, and show a tendency for discount rates to decline as the length of time to the event increases. Further, individuals may have preferences about whether sequences of environmental outcomes are generally improving or declining. Some experimental evidence suggests that individuals tend to discount hyperbolically rather than exponentially, a structure that raises time-consistency concerns. Approaches to social discounting based on alternative perspectives and ecological structures have also been developed, but these have yet to be fully incorporated into the environmental economics literature.[16]

The social opportunity cost of capital represents a situation where investment is crowded out dollar-for-dollar by the costs of environmental policies. This is an unlikely outcome, but it can be useful for sensitivity analysis and special cases. Estimates of the social opportunity costs of capital are typically in the 4.5 percent to 7 percent range depending upon the type of data used.[17]

The utility of the shadow price of capital approach hinges on the magnitude of altered capital flows from the environmental policy. If the policy will substantially displace private investment then a shadow price of capital adjustment is necessary before discounting consumption and consumption equivalents using the social rate of time preference. The literature does not provide clear guidance on the likelihood of this displacement, but it has been suggested that if a policy is relatively small and capital markets fit an "open economy" model, there is probably little displaced investment.[18] Changes in yearly U.S. government borrowing during the past several decades have been in the many billions of dollars. It may be 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. Primarily for these reasons, some argue that for most environmental regulations it is sufficient to discount using a government bond rate with some sensitivity analysis.[19]

6.3 Intergenerational Social Discounting

Policies designed to address long-term environmental problems such as global climate change, radioactive waste disposal, groundwater pollution, or biodiversity will likely involve significant impacts on future generations. This section focuses on social discounting in the context of policies with very long time horizons involving multiple generations, typically referred to in the literature as intergenerational discounting.

16 See Thaler (1990) and Laibson (1998) for more information on mental accounts; Guyse, Keller, and Eppell (2002) on preferences for sequences; Gintis (2000) and Karp (2005) on hyperbolic discounting; and Sumaila and Waters (2005) and Voinov and Farley (2007) for additional treatments on discounting.

17 OMB (2003) recommends a real, pre-tax opportunity cost of capital of 7 percent and refers to *Circular A-94* (1992) as the basis for this conclusion. Moore et al. (2004) estimate a rate of 4.5 percent based on AAA corporate bonds. In recent reviews of EPA's plans to estimate the costs and benefits of the Clean Air Act, the SAB Advisory Council (U.S. EPA 2004c and U.S. EPA 2007b) recommends using a single central rate of 5 percent as intermediate between 3 percent and 7 percent rates, based generally on the consumption rate of interest and the cost of capital, respectively.

18 Lind (1990) first suggested this.

19 See in particular Lesser and Zerbe (1994) and Moore et al. (2004).

Discounting over very long time horizons is complicated by at least three factors: (1) the "investment horizon" is longer than what is reflected in observed interest rates that are used to guide private discounting decisions; (2) future generations without a voice in the current policy process are affected; and (3) compared to intragenerational time horizons, intergenerational investment horizons involve greater uncertainty. Greater uncertainty implies rates lower than those observed in the marketplace, regardless of whether the estimated rates are measured in private capital or consumption terms. Policies with very long time horizons involve costs imposed mainly on the current generation to achieve benefits that will accrue mainly to unborn, future generations, making it important to consider how to incorporate these benefits into decision making. There is little agreement in the literature on the precise approach for discounting over very long time horizons.

This section presents a discussion of the main issues associated with intergenerational social discounting, starting with the Ramsey discounting framework that underlies most of the current literature on the subject. It then discusses how the "conventional" discounting procedures described so far in this chapter might need to be modified when analyzing policies with very long ("intergenerational") time horizons. The need for such modifications arises from several simplifying assumptions behind the conventional discounting procedures described above. Such conventional procedures will likely become less realistic the longer is the relevant time horizon of the policy. This discussion will focus on the social discount rate itself. Other issues such as shadow price of capital adjustments, while still relevant under certain assumptions, will be only briefly touched upon.

Clearly, economics alone cannot provide definitive guidance for selecting the "correct" social welfare function or social rate of time preference. In particular, the fundamental choice of what moral perspective should guide intergenerational social discounting — e.g., that of a social planner who weighs the utilities of present and future generations or those preferences of the current generations regarding future generations — cannot be made on economic grounds alone. Nevertheless, economics can offer important insights concerning discounting over very long time horizons, the implications and consequences of alternative discounting methods, and the systematic consideration of uncertainty. Economics can also provide some advice on the appropriate and consistent use of the social welfare function approach as a policy evaluation tool in an intergenerational context.

6.3.1 The Ramsey Framework

A common approach to intergenerational discounting is based upon methods economists have used for many years in optimal growth modeling. In this framework, the economy is assumed to operate as if a "representative agent" chooses a time path of consumption and savings that maximizes the NPV of the flow of utility from consumption over time.[20] Note that this framework can be viewed in normative terms, as a device to investigate how individuals should consume and reinvest economic output over time. Or it can be viewed in positive terms, as a description (or "first-order approximation") of how the economy actually works in practice. It is a first order approximation only from this positive perspective because the framework typically excludes numerous real-world departures from the idealized assumptions of perfect competition and full information that are required for a competitive market system to produce a Pareto-optimal allocation of resources. If the economy worked exactly as described by optimal growth models — i.e., there were no taxes, market failures, or other distortions — the social discount rate as defined in these models would be equal to the market interest rate. And the market interest rate, in turn, would be equal to the social rate of return on private investments and the consumption rate of interest.

It is worth noting that the optimal growth literature is only one strand of the substantial

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

body of research and writing on intertemporal social welfare. This literature extends from the economics and ethics of interpersonal and intergenerational 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.

As noted earlier, the basic model of optimal economic growth, due to Ramsey (1928), implies equivalence between the market interest rate (r), and the elasticity of marginal utility (η) times the consumption growth rate (g) plus the pure rate of time preference (ρ):

$$r = \eta g + \rho \qquad (10)$$

The first term, ηg, reflects the fact that the marginal utility of consumption will change over time as the level of consumption changes. The second term, ρ, the pure rate of time preference, measures the rate at which individuals discount their own utility over time (taking a positive view of the optimal growth framework) or the rate at which society should discount utilities over time (taking a normative view). Note that if consumption grows over time — as it has at a fairly steady rate at least since the industrial revolution (Valdés 1999) — then future generations will be richer than the current generation. Due to the diminishing marginal utility of consumption, increments to consumption will be valued less in future periods than they are today. In a growing economy, changes in future consumption would be given a lower weight (i.e., discounted at a positive rate) than changes in present consumption under this framework, even setting aside discounting due to the pure rate of time preference (ρ).

There are two primary approaches typically used in the literature to specify the individual parameters of the Ramsey equation: the "descriptive" approach and the "prescriptive," or more explicitly, the normative approach. These approaches are illustrated in Text Box 6.5 for integrated assessment models of climate change.

The descriptive approach attempts to derive likely estimates of the underlying parameters in the Ramsey equation. This approach argues that economic models should be based on actual behavior and that models should be able to predict this behavior. By specifying a given utility function and modeling the economy over time one can obtain empirical estimates for the marginal utility and for the change in growth rate. While the pure rate of time preference cannot be estimated directly, the other components of the Ramsey equation can be estimated, allowing ρ to be inferred.

Other economists take the prescriptive approach and assign parameters to the Ramsey equation to match what they believe to be ethically correct.[21] For instance, there has been a long debate, starting with Ramsey himself, on whether the pure rate of time preference should be greater than zero. The main arguments against the prescriptive approach are that: (1) people (individually and societally) do not make decisions that match this approach; and (2) using this approach would lead to an over-investment in environmental protection (e.g., climate change mitigation) at the expense of investments that would actually make future generations better off (and would make intervening generations better off as well). There is also an argument that the very low discount rate advocated by some adherents to the prescriptive approach leads to unethical shortchanging of current and close generations.

Other analyses have adopted at least aspects of a prescriptive approach. For example, the Stern Review (see Text Box 6.6) sets the pure rate of time preference at a value of 0.1 percent and the elasticity of marginal utility as 1.0. With an assumed population growth rate of 1.3 percent, the social discount rate is 1.4 percent. Guo et al. (2006) evaluate the effects of uncertainty and discounting on the social cost of carbon where the social discount rate is constructed from the Ramsey equation. A number of different discount rate schedules are estimated depending on the adopted parameters.

21 Arrow et al. (1996a).

Text Box 6.5 - Applying these Approaches to the Ramsey Equation

The Ramsey approach has been most widely debated in the context of climate change. Most climate economists adopt a descriptive approach to identify long-term real interest rates and likely estimates of the underlying parameters in the Ramsey equation. William Nordhaus argues that economic models should be based on actual behavior and that models should be able to predict this behavior. His Dynamic Integrated model of Climate and the Economy (DICE), for example, uses interest rates, growth rates, etc., to calibrate the model to match actual historic levels of investment, consumption, and other variables. In the most recent version of the DICE model (Nordhaus 2008), he specifies the current rate of productivity growth to be 5.5 percent per year, the rate of time preference to be 1.5 percent per year, and the elasticity of marginal utility to be 2. In an earlier version (Nordhaus 1993) he estimates the initial return on capital (and social discount) to be 6 percent, the rate of time preference to be 2 percent, and the elasticity of marginal utility to be 3. Because the model predicts that economic and population growth will slow, the social discount rate will decline.

While use of the Ramsey discounting framework is quite common and is based on an intuitive description of the general problem of trading off current and future consumption, it has some limitations. In particular, it ignores differences in income within generations (at least in the basic single representative agent version of the model). Arrow (1996a) contains detailed discussion of descriptive and prescriptive approaches to discounting over long time horizons, including examples of rates that emerge under various assumptions about components of the Ramsey equation.

6.3.2 Key Considerations

There are a number of important ways in which intergenerational social discounting differs from intragenerational social discounting, essentially due to the length of the time horizon. Over a very long time horizon it is much more difficult, if not impossible, for analysts to judge whether current generation preferences also reflect those of future generations and how per capita consumption will change over time. This section discusses efficiency and intergenerational equity concerns, and uncertainty in this context.

6.3.2.1 Efficiency and Intergenerational Equity

A principal problem with policies that span long time horizons is that many of the people affected are not yet alive. While the preferences of each affected individual are knowable (if perhaps unknown in practice) in an intragenerational context, the preferences of future generations in an intergenerational context are essentially unknowable. This is not always a severe problem for practical policy making, especially when policies impose relatively modest costs and benefits, or when the costs and benefits begin immediately or in the not too distant future. Most of the time, it suffices to assume 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. 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. In these cases social discounting can no longer be thought of as a process of consulting the preferences of all affected parties concerning today's valuation of effects they will experience in future time periods.

Moreover, compounding interest over very long time horizons can have profound impacts on the intergenerational distribution of welfare. An extremely large benefit or cost realized far into the future has essentially a present value of zero, even when discounted at a low rate. But a modest sum invested today at the same low interest rate 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 implications is not advised.[22]

For example, in the climate change context, Pearce et al. (2003) show that decreasing the discount rate from a constant 6 percent to a constant 4 percent nearly doubles the estimate of the marginal benefits from carbon dioxide (CO_2) emission reductions. Weitzman (2001) shows that moving from a constant 4 percent discount rate to a declining discount rate approach nearly doubles the estimate again. Newell and Pizer (2003) show that constant discounting can substantially undervalue the future given uncertainty in economic growth and the overall investment environment. For example, Newell and Pizer (2003) show that a constant discount rate could undervalue net present benefits by 21 percent to 95 percent with an initial rate of 7 percent, and 440 percent to 700 percent with an initial rate of 4 percent, depending upon the model of interest rate uncertainty.

Using observed market interest rates for intergenerational discounting in the representative agent Ramsey framework essentially substitutes the pure rate of time preference exhibited by individuals for the weight placed on the utilities of future generations relative to the current generation (see OMB 2003 and Arrow et al. 1996). Many argue that the discount rate should be below market rates — though not necessarily zero — to: (1) correct for market distortions and inefficiencies in intergenerational transfers; and (2) so that generations are treated equally based on ethical principles (Arrow et al. 1996, and Portney and Weyant 1999).[23]

Intergenerational Transfers

The notion of Pareto compensation attempts to identify the appropriate social discount rate in an intergenerational context by asking whether the distribution of wealth across generations could be adjusted to compensate the losers under an environmental policy and still leave the winners better off than they would have been absent the policy. Whether winners could compensate losers across generations hinges on the rate of interest at which society (the United States 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. Some authors also consider the infeasibility of intergenerational transfers to be a fundamental problem for discounting across generations.[24]

Equal Treatment Across Generations

Environmental policies that affect distant future generations can be considered to be altruistic acts.[25] As such, some argue that they should be valued by current generations in exactly the same way as other acts of altruism are valued. Under this logic, the relevant discount rate is not based on an individual's own consumption, but instead on an individual's valuation of the consumption (or welfare) of someone else. These altruistic values can be estimated through either revealed or stated preference methods.

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 as temporal, cultural, geographic, and other measures of "distance" increase. The implied discount rates survey respondents appear to apply in trading off present and future lives also is relevant under this approach. One such survey (Cropper, Aydede, and Portney 1994) suggests that these rates are positive on average, which is consistent with the rates at which people discount monetary outcomes. The rates decline as the time horizon involved lengthens.

22 OMB's *Circular A-4* (2003) requires the use of constant 3 percent and 7 percent for both intra- and intergenerational discounting for benefit-cost estimation of economically significant rules but allows for lower, positive consumption discount rates, perhaps in the 1 percent to 3 percent range, if there are important intergenerational values.

23 Another issue is that there are no market rates for intergenerational time periods.

24 See Lind (1990) and a summary by Freeman (2003).

25 Schelling (1995), and Birdsall and Steer (1993) are good references for these arguments.

6.3.2.2 Uncertainty

A longer time horizon in an intergenerational policy context also implies greater uncertainty about the investment environment and economic growth over time, and a greater potential for environmental feedbacks to economic growth (and consumption and welfare), which in turn further increases uncertainty when attempting to estimate the social discount rate.

This additional uncertainty has been shown to imply effective discount rates lower than those based on the observed average market interest rates, regardless of whether or not the estimated investment effects are predominantly measured as private capital or consumption terms (Weitzman 1998, 2001; Newell and Pizer 2003; Groom et al. 2005; and Groom et al. 2007).[26] The rationale for this conclusion is that consideration of uncertainty in the discount rate should be based on the average of discount factors (i.e., $1/(1+r)^t$) rather than the standard discount rate (i.e., r). From the expected discount factor over any period of time a constant, certainty-equivalent discount rate that yields the discount factor (for any given distribution of r) can be inferred. Several methods for accounting for uncertainty into intergenerational discounting are discussed in more detail in the next section.

6.3.3 Evaluating Alternatives

There is a wide range of options available to the analyst for discounting intergenerational costs and benefits. Several of these are described below, ordered from simplest to most analytically complex. Which option is utilized in the analysis is left to expert judgment, but should be based on the likely consequences of undertaking a more complex analysis for the bottom-line estimate of expected net benefits. This will be a function of the proportion of the costs and benefits occurring far out on the time horizon and the separation of costs and benefits over the planning horizon. When it is unclear which method should be utilized, the analyst is encouraged to explore a variety of approaches.

26 Gollier and Zeckhauser (2005) reach a similar result using a model with decreasing absolute risk aversion.

6.3.3.1 Constant Discount Rate

One possible approach is to simply make no distinction between intergenerational and intragenerational social discounting. For example, models of infinitely-lived individuals suggest the consumption rate of interest as the social discount rate. Of course, individuals actually do not live long enough to experience distant future consequences of a policy and cannot report today the present values they place on those effects. However, it is equally sufficient to view this assumption as a proxy for family lineages in which the current generation treats the welfare of all its future generations identically with the current generation. It is not so much that the individual lives forever as that the family spans many generations (forever) and that the current generation discounts consumption of future generations at the same rate as its own future consumption.

Models based on constant discount rates over multiple generations essentially ignore potential differences in economic growth and income and/or preferences for distant future generations. Since economic growth is unlikely to be constant over long time horizons, the assumption of a constant discount rate is unrealistic. Interest rates are a function of economic growth; thus, increasing (declining) economic growth implies an increasing (decreasing) discount rate.

A constant discount rate assumption also does not adequately account for uncertainty. Uncertainty regarding economic growth increases as one goes further out in time, which implies increasing uncertainty in the interest rate and a declining certainty equivalent rate of return to capital (Hansen 2006).

6.3.3.2 Step Functions

Some modelers and government analysts have experimented with varying the discount rate with the time horizon to reflect non-constant economic growth, intergeneration equity concerns, and/or heterogeneity in future preferences. For instance, in the United Kingdom the Treasury recommends the use of a 3.5 percent discount rate for the first 30 years followed by a declining rate over future time periods until it reaches 1 percent for 301 years and beyond.[27] This method acknowledges that a constant discount rate does not adequately reflect the reality of fluctuating and uncertain growth rates over long time horizons. However, application of this method also raises several potential analytic complications. First, there is no empirical evidence to suggest the point(s) at which the discount rate declines, so any year selected for a change in the discount rate will be necessarily ad-hoc. Second, this method can suffer from a time inconsistency problem. Time inconsistency means that an optimal policy today may look sub-optimal in the future when using a different discount rate and vice versa. Some have argued that time inconsistency is a relatively minor problem relative to other conditions imposed (Heal 1998, Henderson and Bateman 1995, and Spackman 2004).

6.3.3.3 Declining or Non-Constant Discount Rate

Using a constant discount rate in BCA is technically correct only if the rate of economic growth will remain fixed over the time horizon of the analysis. If economic growth is changing over time, then the discount rate, too, will fluctuate. In particular, one may assume that the growth rate is declining systematically over time (perhaps to reflect some physical resource limits), which will lead to a declining discount rate. This is the approach taken in some models of climate change.[28] In principle, any set of known changes to income growth, the elasticity of marginal utility of consumption, or the pure rate of time preference will lead to a discount rate that changes accordingly.

6.3.3.4 Uncertainty-Adjusted Discounting

If there is uncertainty about the future growth rate, then the correct procedure for discounting must account for this uncertainty in the calculation of the expected NPV of the policy. Over the long time horizon, both investment uncertainty and risk will naturally increase, which results in a decline in the imputed discount rate. If the time horizon of the policy is very long, then eventually a low discount rate will dominate the expected NPV calculations for benefits and costs far in the future (Weitzman 1998).

Newell and Pizer (2003) expand on this observation, using historical data on U.S. interest rates and assumptions regarding their future path to characterize uncertainty and compute a certainty equivalent rate. In this case, uncertainty in the individual components of the Ramsey equation is not being modeled explicitly. Their results illustrate that a constant discount rate could substantially undervalue net present benefits when compared to one that accounts for uncertainty. For instance, a constant discount rate of 7 percent could undervalue net present benefits by between 21 percent and 95 percent depending on the way in which uncertainty is modeled.

A key advantage of this treatment of the discount rate over the step function and simple declining rate discounting approaches is that the analyst is not required to arbitrarily designate the discount rate transitions over time, nor required to ignore the effects of uncertainty in economic growth over time. Thus, this approach is not subject to the time inconsistency problems of some other approaches. Another issue that has emerged about the use of discount rates that decline over time due to uncertainty is that they could generate inconsistent policy rankings NPV versus NFV.[29] Because the choice between NPV and NFV is arbitrary, such an outcome would be problematic for applied policy analysis. More recent work, however, appears to resolve this seeming inconsistency, confirming the original findings and providing sound conceptual rationale for the approach.[30]

27 The guidance also requires a lower schedule of rates, starting with 3 percent for zero to 30 years, where the pure rate of time preference in the Ramsey framework (the parameter ρ in our formulation) is set to zero. For details see HM Treasury (2008) *Intergenerational wealth transfers and social discounting: Supplementary Green Book Guidance.*

28 See, for example, Nordhaus (2008).

29 See Gollier (2004) for a technical characterization of this concern, and Hepburn and Groom (2007) for additional exploration of the issues.

30 See Gollier and Weitzman (2009) provide a concise and clear treatment. Freeman (2009) and Gollier (2009) also propose solutions.

Text Box 6.6 - What's the Big Deal with *The Stern Review*?

In autumn 2006, the U.K. government released a detailed report titled *The Economics of Climate Change: The Stern Review*, headed by Sir Nicholas Stern (2006). The report drew mainly on published studies and estimated that damages from climate change could result in a 5 percent to 20 percent decline in global output by 2100. The report found that costs to mitigate these impacts were significantly less (about 1 percent of GDP). Stern's findings led him to say that "climate change is the greatest and widest-ranging market failure ever seen," and that "the benefits of strong early action considerably outweigh the cost." *The Stern Review* recommended that policies aimed towards sharp reduction in GHG emissions should be enacted immediately.

While generally lauded for its thoroughness and use of current climate science, *The Stern Review* drew significant criticism and discussion of how future benefits were calculated, namely targeting Stern's assumptions about the discount rate (Tol and Yohe 2006 and Nordhaus 2008). *The Stern Review* used the Ramsey discounting equation (see Section 6.3.1), applying rates of 0.1 percent for the annual pure rate of time preference, 1.3 percent for the annual growth rate, and a elasticity of marginal utility of consumption equal to 1. Combining these parameter values reveals an estimated equilibrium real interest rate of 1.4 percent, a rate arguably lower than most returns to standard investments, but not outside the range of values suggested in these *Guidelines* for intergenerational discount rates.

So why is the issue on the value of the discount rate so contentious? Perhaps the biggest concern is that climate change is expected to cause significantly greater damages in the far future than it is today, and thus benefits are sensitive to discounting assumptions. A low social discount rate means *The Stern Review* places a much larger weight on the benefits of reducing climate change damages in 2050 or 2100 relative to the standard 3 percent or 7 percent commonly observed in market rates. Furthermore, Stern's relatively low values of ρ and η imply that the current generation should operate at a higher savings rate than what is observed, thus implying that society should save more today to compensate losses incurred by future generations.

Why did Stern use these particular parameter values? First, he argues that the current generation has an ethical obligation to place similar weights on the pure rate of time for future generations. Second, a marginal elasticity of consumption of unity implies a relatively low inequality aversion, which reduces the transfer of benefits between the rich and the poor relative to a higher elasticity. Finally, there are significant risks and uncertainties associated with climate change, which could imply using a lower-than-market rate. Stern's (2006) concluding remarks for using a relatively low discount rate are clear, "However unpleasant the damages from climate change are likely to appear in the future, any disregard for the future, simply because it is in the future, will suppress action to address climate change."

6.4 Recommendations and Guidance

As summed up by Freeman (2003 p. 206), "economists have not yet reached a consensus on the appropriate answers" to all of the issues surrounding intergenerational discounting. And while there may be more agreement on matters of principle for discounting in the context of intragenerational policies, there is still some disagreement on the magnitude of capital displacement and therefore the importance of accounting for the opportunity costs of capital in practice.[31] The recommendations provided here are intended as practical and plausible default assumptions rather than comprehensive and precise estimates of social discount rates that must be applied without adjustment in all situations. That is, these recommendations should be used as a starting point for BCA, but if the

31 This chapter summarizes some key aspects from the core literature on social discounting; it is not a detailed review of the vast and varied social discounting literature. Excellent sources for additional information are: Lind (1982a, b; 1990; 1994), Lyon (1990, 1994), Kolb and Scheraga (1990), Scheraga (1990), Arrow et al. (1996), Pearce and Turner (1990), Pearce and Ulph (1994), Groom et al. (2005), Cairns (2006), Frederick et al. (2002), Moore et al. (2004), Spackman (2004), and Portney and Weyant (1999).

analysts can develop a more realistic model and bring to bear more accurate empirical estimates of the various factors that are most relevant to the specific policy scenario under consideration, then they should do so and provide the rationale in the description of their methods. With this caveat in mind, our default recommendations for discounting are below.

- Display the time paths of benefits and costs as they are projected to occur over the time horizon of the policy, i.e., without discounting.
- The shadow price of capital approach is the analytically preferred method for discounting, but there is some disagreement on the extent to which private capital is displaced by EPA regulatory requirements. EPA will undertake additional research and analysis to investigate important aspects of this issue, including the elasticity of capital supply, and will update guidance accordingly. In the interim analysts should conduct a bounding exercise as follows:
  - Calculate the NPV using the consumption rate of interest. This is appropriate for situations where all costs and benefits occur as changes in consumption flows rather than changes in capital stocks, i.e., capital displacement effects are negligible. As of the date of this publication, current estimates of the consumption rate of interest, based on recent returns to Government-backed securities, are close to 3 percent.
  - Also calculate the NPV using the rate of return to private capital. This is appropriate for situations where all costs and benefits occur as changes in capital stocks rather than consumption flows. The OMB estimates a rate of 7 percent for the opportunity cost of private capital.
  - EPA intends to periodically review the empirical basis for the consumption discount rate and the rate of return to private capital.

  In most cases the results of applying the more detailed "shadow price of capital" approach will lie somewhere between the NPV estimates ignoring the opportunity costs of capital displacements and discounting all costs and benefits using these two alternative discount rates.

- If the policy has a long time horizon (more than 50 years or so) where net benefits vary substantially over time (e.g., most benefits accrue to one generation and most costs accrue to another) then the analysis should use the consumption rate of interest as well as additional approaches. These approaches include calculating the expected present value of net benefits using an estimated time-declining schedule of discount factors (Newell and Pizer 2003, Groom et al. 2007, and Hepburn et al. 2009). This approach accounts for discount rate uncertainty and variability, which are known to have potentially large effects on NPV estimates for policies with long time horizons. If a time-declining approach cannot be implemented, it is possible to capture part of its empirical effect by discounting at a constant rate somewhat lower than those used in the conventional case. For example, the current Interagency guidance for valuing CO_2 emission reductions includes treatment with certainty-equivalent constant discount rates of 2.5 percent, 3 percent, and 5 percent. (See Text Box 7.1 for more discussion of the Interagency guidance.)

Other more detailed alternatives, such as constructing discounts rate from estimates of the individual parameters in the Ramsey equation, may merit inclusion in the analysis. In any case, all alternatives should be fully described, supported, and justified.

When implementing any discounting approach the following principles should be kept in mind:

- In all cases social benefits and costs should be discounted in the same manner, although private discount rates may be used to predict behavior and to evaluate economic impacts.
- The discount rate should reflect marginal rates of substitution between consumption in different time periods and should not be confounded with factors such as uncertainty in benefits and costs or the value of environmental goods or other commodities in the future (i.e., the "current price" in future years).
- The lag time between a change in regulation and the resulting welfare impacts should be accounted for in the economic analysis. The monetary benefits from the expected future impacts should be discounted at the same rate as other benefits and costs in the analysis. This includes changes in human health, environmental conditions, ecosystem services, etc.

Chapter 7
Analyzing Benefits

The aim of an economic benefits analysis is to estimate the benefits, in monetary terms, of proposed policy changes in order to inform decision making. Estimating benefits in monetary terms allows the comparison of different types of benefits in the same units, and it allows the calculation of net benefits — the sum of all monetized benefits minus the sum of all monetized costs — so that proposed policy changes can be compared to each other and to the baseline scenario.

The discussion in this chapter focuses on methods and approaches available to monetize benefits in the context of a "typical" EPA policy, program, or regulation that reduces emissions or discharges of contaminants. This is not to say that those benefits that cannot be monetized due to lack of available values or quantification methods are not important. Chapter 11 on the "Presentation of Analysis and Results" discusses how to carry forward information on non-monetized benefits to help inform the policy-making process. In addition, this chapter includes a discussion of several alternatives to monetization that may add some context to this category of benefits. The general monetization methods and principles discussed here should apply to other types of EPA polices as well, such as those that provide regulatory relief, encourage reuse of remediated land, or provide information to the public to help people avoid environmental risks.[1]

7.1 The Benefits Analysis Process

Ideally, benefits analyses would consist of comprehensive assessments of all environmental effects attributable to the rule in question. However, it is seldom possible to analyze all effects simultaneously in an integrated fashion. In most cases, analysts will need to address each effect individually, and then aggregate the individual values to generate an estimate of the total benefits of a policy. A constant challenge in employing an effect-by-effect approach is to balance potential trade-offs between inclusion and redundancy.

Ideally, each effect will be measured once and only once. Techniques intended to bring additional effects into the analysis may run the risk of double counting effects already measured. For example, stated preference methods may be the only way to measure non-use values, but they may double count use values already reflected in hedonic or travel cost analyses. Therefore, the analyst should be careful in interpreting and combining the results of different methods.

There are of course exceptions to this "effect-by-effect" approach to benefits analysis (e.g., efforts to estimate the social benefits of reducing CO_2 emissions — see Text Box 7.1), but the remainder of the discussion below is framed with this approach in mind.

A second challenge analysts often face is the difficulty of conducting original valuation research in support

1 Other methods, such as cost-effectiveness analysis (CEA), can also be used to evaluate policies. CEA does not require monetization of benefits but rather divides the costs of a policy by a particular effect (e.g., number of lives saved). CEA can be used to compare proposed policy changes on an effect-by-effect basis, but, unlike BCA, cannot be used to calculate a single, comprehensive measure of the net effects of a policy, nor can it compare proposed policy changes to the status quo. Other methods for evaluating policies (e.g., distributional analyses) are covered in Chapter 10.

Text Box 7.1 - Estimating Benefits from Reducing Carbon Dioxide Emissions: The Social Cost of Carbon

Monetized estimates of the damages associated carbon dioxide (CO_2) emissions allows the social benefits of regulatory actions that are expected to reduce these emissions to be incorporated into BCA. One way to accomplish this is through the estimation of the "social cost of carbon" (SCC). The SCC is the present value of the stream of future economic damages associated with an incremental increase (by convention, one metric ton) in CO_2 emissions in a particular year. It is intended to be a comprehensive measure and includes economic losses due to changes in agricultural productivity, human health risks, property damages from increased flood frequencies, the loss of ecosystem services, etc. The SCC is a marginal value so it may not be accurate for valuing large changes in emissions. However, many U.S. government regulations will lead to relatively small reductions in cumulative global emissions, so for these regulations the SCC is the appropriate shadow value for estimating the economic benefits of CO_2 reductions.

Most published estimates of the SCC have been derived from "integrated assessment models" (IAMs) that combine climate processes, economic growth, and feedbacks between the two in a single modeling framework. These models include a reduced form representation of the potential economic damages from climate change. Therefore IAMs used to estimate the SCC are necessarily highly simplified and limited by the current state of the climate economics literature, which continues to expand rapidly. Despite the inherent uncertainties in models such as these, they are the best tools currently available for estimating the SCC.

The Interagency SCC Workgroup. In 2009, an interagency workgroup composed of members from six federal agencies and various White House offices was convened to improve the accuracy and consistency in how agencies value reductions in CO_2 emissions in regulatory impact analyses. The resulting range of values is based on estimates from three integrated assessment models applied to five socioeconomic and emissions scenarios, all given equal weight. To reflect differing expert opinions about discounting, the present value of the time path of global damages in each model-scenario combination was calculated using discount rates of 5 percent, 3 percent, and 2.5 percent. Finally, in a step toward more formal uncertainty analysis, all model runs employed a probabilistic representation of climate sensitivity (in addition to other parameters in two of the models).

The workgroup selected four SCC estimates from the model runs to reflect the global damages caused by CO_2 emissions: $5, $21, $35, and $65 for 2010 emission reductions (in 2007 U.S. dollars). The first three estimates are based on the average SCC across the three models and five socioeconomic and emissions scenarios for the 5 percent, 3 percent, and 2.5 percent discount rates, respectively. The fourth value, the 95^{th} percentile of the SCC distribution at a 3 percent discount rate, was chosen to represent potential higher-than-expected impacts from temperature change. The SCC estimates grow over time at rates endogenously determined by the models. For instance, with a discount rate of 3 percent, the mean SCC estimate increases to $24 per ton of CO_2 in 2015 and $26 per ton of CO_2 in 2020.

Going Forward. The Interagency SCC Workgroup presented the SCC estimates with a clear acknowledgement of the many uncertainties involved and the final report outlined a number of limitations to the analysis. The Interagency SCC Workgroup is committed to re-visiting these estimates on a regular basis and revising them as needed to reflect the growing body of scientific knowledge regarding climate change impacts and the potential economic damages from those impacts.

Further Reading: U.S. Interagency Working Group on Social Cost of Carbon (2010). Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866, www.epa.gov/otaq/climate/regulations/scc-tsd.pdf.

of specific policy actions. Because budgetary and time constraints often make performing original research infeasible, analysts regularly need to draw upon existing value estimates for use in benefits analysis. The process of applying values estimated in previous studies to new policy cases is called *benefit transfer*. The benefit transfer method is discussed in detail in Section 7.4, but much of this chapter is written with benefit transfer in mind. In particular, the descriptions of revealed and stated preference valuation methods in Sections 7.3.1 and 7.3.2 include recommendations for evaluating the quality and suitability of published studies for use in benefit transfer.

A general "effect-by-effect" approach to benefits analysis

This approach consists of separately evaluating the major effects of a given policy, and then summing these individual estimates to arrive at an overall estimate of total benefits. The effect-by-effect approach for benefits analysis requires three fundamental steps:

1. Identify benefit categories potentially affected by the policies under consideration;
2. Quantify significant endpoints to the extent possible by working with managers, risk assessors, ecologists, physical scientists, and other experts; and
3. Estimate the values of these effects using appropriate valuation methods for new studies or existing value estimates from previous studies that focus on the same or sufficiently similar endpoints.

Each step in this approach is discussed in more detail below. Analysts also should consider whether this general framework is appropriate for assessing a specific policy or whether a more integrated approach that incorporates all of the relevant effects simultaneously can be applied. When applying the effect-by-effect approach it is important to avoid double counting benefits across effects as much as possible. Collaboration with appropriate experts will be necessary to execute these steps meaningfully.

Step1: Identify potentially affected benefit categories

The first step in a benefits analysis is to determine the types of benefits associated with the policy options under consideration. More information on benefits categories can be found in Section 7.2. To identify benefit categories, analysts should, to the extent feasible:

Develop an initial understanding of policy options of interest by working with other analysts and policy makers. Initially, the range of options considered may be very broad. Resources should be focused on benefit categories that are likely to influence policy decisions. Collaboration between all parties involved in the policy analysis can help ensure that all potential effects are recognized and that the necessary and appropriate information and endpoints are collected and evaluated at each step in the process. Analysts should take care to think through potential secondary or indirect effects of the policy options as well, as these may prove to be important.

Research the physical effects of the pollutants on human health and the environment by reviewing the literature and consulting with other experts. This step requires considering the transport of the pollutants 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 pollutants can have detrimental effects on natural resources, such as affecting oxygen availability in surface water or reducing crop yields. Pollutants can also have direct or indirect effects on human health, for example affecting cancer incidence through direct inhalation or through ingestion of contaminated food.

Consider the potential change in these effects as a result of each policy option. 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 are increased or remain unchanged. Evaluating how physical effects change under each policy option requires evaluation of how the pathways differ in the "post-policy" world.

Determine which benefit categories to include in the overall benefits analysis using at least the following three criteria:

1. Which benefit categories are likely to differ across policy options, including the baseline option? Analysts should conduct an assessment of how the physical effects of each policy option will differ and how each physical effect will impact each benefit category.
2. Which benefit categories are likely to account for the bulk of the total benefits of the policy? The cutoff point here should be based on an assessment of the magnitude and precision of the estimates of each benefit category, the total social costs of each policy option, and the costs of gathering further information on each benefit category. A benefit category should not be included if the cost of gathering the information necessary to include it is greater than the expected increase in the value of the policy owing to its inclusion. The analyst should make these preliminary assessments using the best quantitative information that is readily available, but as a practical matter these decisions may often have to be based on professional judgments.
3. Which benefit categories are especially salient to particular stakeholders? Monetized benefits in this category are not necessarily large and so may not be captured by the first two criteria.[2]

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(s), identifies the benefit categories associated with these effects, and identifies the effects that warrant further investigation.

The list of physical effects under each benefit category may be lengthy at first, encompassing all of those that reasonably can be associated with the policy options under consideration. Analysts should preserve and refine this list of physical effects as the analysis proceeds. Maintaining the full list of potential effects — even though the quantitative analysis will (at least initially) focus on a sub-set of them — will allow easy revision of the analysis plan if new information warrants it.

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 information (U.S. EPA 2009a). No specific guidance exists for assessing changes in amenities or material damages. Analysts should consult relevant experts and existing literature to determine the "best practices" appropriate for these categories of benefits.

Step 2: Quantify significant endpoints

The second step is to quantify the physical endpoints related to each category, focusing on changes attributable to each policy option relative to the baseline. Data are usually needed on the extent, timing, and severity of the endpoints. For example, if the risk of lung cancer is an endpoint of concern, required information will usually include the change in risk associated with each option, the timing of the risk changes, the age distribution of affected populations, and fatality rates. If visibility is the attribute of concern, required information will usually include the geographical areas affected and the change in visibility resulting from each policy option.

Analysts should keep the following issues in mind while quantifying significant physical effects.

Work closely with analysts in other fields. Estimating physical effects is largely, but not completely, the domain of other experts, including human health and ecological risk assessors and other natural scientists. These experts generally are responsible for evaluating the likely transport of the pollutant through the environment and its potential effects on humans, ecological systems, and manufactured materials.

2 This third criterion relates to distributional considerations detailed in Chapter 10.

Text Box 7.2 - Integrating Economics and Risk Assessment

Historically, health and ecological risk assessments have been designed not to support benefits analyses per se but rather to support the setting of standards or to rank the severity of different hazards. Traditional measures of risk can be difficult or impossible to incorporate into benefits analyses. For example, traditional measures of risk are often based on endpoints not directly related to health outcomes or ecological services that can be valued using economic methods. These measures are often based on outcomes near the tails of the risk distribution for highly sensitive endpoints, which would lead to biased benefits estimates if extrapolated to the general population.

Because economists rely on risk assessment outcomes as key inputs into benefits analysis, it is important that risk assessments and economic valuation studies be undertaken together. Economists can contribute information and insights into how behavioral changes may affect realized risk changes. For example, if health outcomes in a particular risk assessment are such that early medical intervention could reduce the chances of illness, economists may be able to estimate changes in the probability that individuals will seek preventative care. Even in cases where the economists' contribution to the risk characterization is not direct, economists and risk assessors should communicate frequently to ensure that economic analyses are complete. Specifically risk assessors and economists should:

- Identify a set of human health and ecological endpoints that are economically meaningful. The endpoints should be linked to human well-being and monetized using economic valuation methods. This may require risk assessors to model more or different outcomes than they would if they were attempting to capture only the most sensitive endpoint. This also may require risk assessors and economists to convert specific human health or ecological endpoints measured in laboratory or epidemiological studies to other effects that can be valued in the economic analysis.
- Estimate changes in the probabilities of human health or ecological outcomes rather than "safety assessment" measures such as reference doses and reference concentrations.
- Work to produce expected or central estimates of risk, rather than bounding estimates as in safety assessments. At a minimum, any expected bias in the risk estimates should be clearly described.
- Attempt to estimate the "cessation lag" associated with reductions in exposure. That is, the analysis should characterize the time profile of changes in exposures and risks.
- Attempt to characterize the full uncertainty distribution associated with risk estimates. Not only does this contribute to a better understanding of potential regulatory outcomes, it also enables economists to incorporate risk assessment uncertainty into a broader analysis of uncertainty. Formal probabilistic assessment is required for some regulations by *Circular A-4* (OMB 2003). Also refer to EPA's guidance and reference documents on Monte Carlo methods and probabilistic risk assessment, including EPA's Policy for Use of Probabilistic Analysis in Risk Assessments (U.S. EPA 1997e), and the 1997 Guiding Principles for Monte Carlo Analysis (U.S. EPA 1997d).

The principal role of the economist at this stage is to ensure that the information provided is useful for the subsequent economic valuation models that may be used later in the benefits analysis. The analyst should give special care to ensuring that the endpoints evaluated are appropriate for use in benefits estimation. Effects that are described too broadly or that cannot be linked to human well-being limit the ability of the analysis to capture the full range of a policy's benefits. Text Box 7.2 provides examples and a more detailed discussion.

Another important role for economists at this stage is to provide insights, information, and analysis on behavioral changes that can affect the results of the risk assessment as needed. Changes in behavior due to changes in environmental quality (e.g., staying indoors to avoid detrimental effects of air pollution) can be significant and care should be taken to account for such responses in risk assessments and benefit estimations.

Step 3: Estimate the values of the effects

The next step is to estimate willingness to pay (WTP) of all affected individuals for the quantified benefits in each benefit category, and then to aggregate these to estimate the total social benefits of each policy option. Typically, a representative agent approach is used when deriving estimates of benefits. The analyst calculates WTP for an "average" individual in a sample of people from the relevant population and then multiply that average value by the number of individuals in the exposed population to derive an estimate of total benefits. As discussed earlier, markets do not exist for many of the types of benefits expected to result from environmental regulations. Details on the economic valuation methods suitable for this step and examples of how they can be applied can be found in Section 7.3. In applying these methods, analysts should:

Consider using multiple valuation methods when possible. Different methods often address different subsets of total benefits and the use of multiple methods allows for comparison of alternative measures of value when applied to the same category of benefits. Double counting is a significant concern when applying more than one method. Any potential overlap should be noted when presenting the results. The discussion of benefit transfer in Section 7.4 describes many of the issues involved in applying value estimates from previous studies to new policy cases, including various meta-analysis techniques for combining estimates from multiple studies.

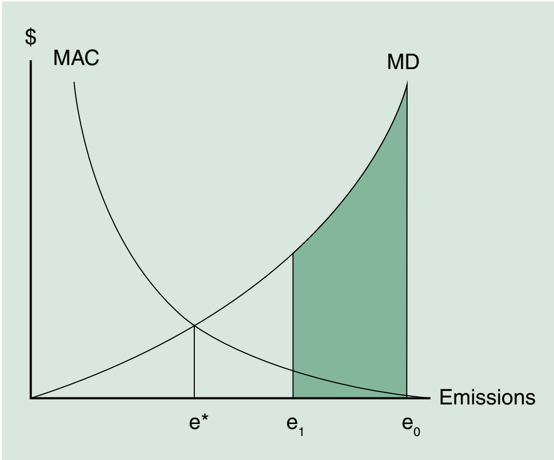
Describe the source of estimates and confidence in those sources. Valuation estimates always contain a degree of uncertainty. Using them in a context other than the one in which they were initially estimated can only increase that uncertainty. If many high-quality studies of the same effect have produced comparable values, analysts can have more confidence in using these estimates in their benefits calculations. In other cases, analysts may have only a single study, or even no directly comparable study, to draw from. In all cases, the benefits analysis should clearly describe the sources of the value estimates used and provide a qualitative discussion of the reliability of those sources. The analyst should include a quantitative uncertainty assessment when possible. Guiding principles for presenting uncertainty are addressed in Chapter 11.

7.2 Economic Value and Types of Benefits

Economic valuation is based on the traditional economic theory of human behavior and preferences, which centers on the concept of "utility" (or "satisfaction" or "welfare") that people realize from goods and services, both market and non-market. Different levels and combinations of goods and services afford different levels of utility for any one person. Because different people have different preferences, different sets of goods and services will appeal more or less to different people. Utility is inherently subjective and cannot be measured directly. Therefore, in order to give "value" an operational definition it must be expressed in a quantifiable metric. Money generally is used as the metric, but this choice for the unit of account has no special theoretical significance. One could use "apples," "bananas," or anything else that is widely valued and consumed by individuals. The crucial assumption is that a person could be compensated for the loss of some additional quantity of any good by some quantity of another good that is selected as the metric. Table 7.1 summarizes the types of benefits associated with environmental protection policies and provides examples of each of the benefits types, as well as valuation methods commonly used to monetize the benefits for each type.

When goods and services are bought and sold in competitive markets, the ratio of the marginal utility (the utility afforded by the last unit purchased) of any two goods that a person consumes must be equal to the ratio of the prices of those goods. If it were otherwise, that person could reallocate her budget to buy a little more

Figure 7.1 - Benefits of an Environmental Improvement



of one good and a little less of the other good to achieve a higher level of utility. Thus, market prices can be used to measure the value of market goods and services directly. A practical rationale for using money as the metric for non-market valuation is that money is the principal medium of exchange for the wide variety of market goods and services among which people choose on a daily basis.

The benefits of an environmental improvement are shown graphically in Figure 7.1. Reducing emissions from e_0 to e_1 produces benefits equal to the shaded area under the marginal damages (MD) curve. Many environmental goods and services, such as air quality and biological diversity, are not traded in markets. The challenge of valuing non-market goods that do not have prices is to relate them to one or more market goods that do. This can be done either by determining how the non-market good contributes to the production of one or more market goods (often in combination with other market good inputs), or by observing the trade-offs people make between non-market goods and market goods. One way or another, this is what each of the revealed and stated preference valuation methods discussed in Section 7.3 is designed to do. Of course, some methods will be more suitable than others in any particular case for a variety of reasons, and some will be better able to capture certain types of benefits than others. In principle, though, they are all different ways of measuring the same thing, which is the total amount of money required to make all individuals indifferent between the baseline and policy scenarios.

The economic valuation of an environmental improvement is the dollar value of the private goods and services that individuals would be willing to trade for the improvement at prevailing market prices. The willingness to trade compensation for goods or services can be measured either as *willingness to pay* (WTP) or *willingness to accept* (WTA). WTP is the maximum amount of money an individual would voluntarily pay to obtain an improvement. WTA is the least amount of money an individual would accept to forego the improvement.[3] The key theoretical distinction between WTP and WTA is their respective reference utility levels. For environmental improvements, WTP uses the level of utility *without* the improvement as the reference point while WTA uses the level of utility *with* the improvement as the reference point. Because of their different reference points, one relevant factor to consider when deciding whether WTP or WTA is the appropriate value measure to use in a BCA is the property rights for the environmental resource(s) in question. WTP is consistent with individuals or firms having rights to pollute and the affected parties needing to pay them to desist. WTA is consistent with individuals being entitled to a clean environment and needing to be compensated for any infringements of that right (Freeman 2003).

Economists generally expect that the difference between WTP and WTA will be small, provided the amounts in question are a relatively small proportion of income and the goods in question are not without substitutes, either market or non-market. However, there may be instances in which income and substitution effects are important.[4] To simplify the presentation, the term WTP is used throughout the remainder of this chapter to refer

3 For simplicity, the discussion in this section is restricted to the case of environmental improvements, but similar definitions hold for environmental damages. For a more detailed treatment of WTP and WTA and the closely related concepts of compensating variation, equivalent variation, and Hicksian and Marshallian consumer surplus, see Hanley and Spash (1993), Freeman (2003), Just et al. (2005), and Appendix A of these *Guidelines*.

4 For more information see Appendix A and Hanemann (1991).

to the underlying economic principles behind both WTA and WTP, but the analyst should keep the potential differences between the two measures in mind.

Based on the connection to individual welfare just described, estimates of WTP are needed for the Kaldor and Hicks potential compensation tests that form the basis of BCA (Boadway and Bruce 1984, Just et al. 1982, and Freeman 2003). To carry out these tests, sum the WTP for all affected individuals and compare the summed WTP value to the estimated costs of the proposed policy. Because environmental policy typically deals with improvements rather than deliberate degradation of the environment, WTP is generally the relevant measure.[5]

The types of benefits that may arise from environmental policies can be classified in multiple ways (Freeman 2003). As shown in Table 7.1, these *Guidelines* categorize benefits as human health improvements, ecological improvements, and other types of benefits, including aesthetic improvements and reduced materials damages, and list commonly used valuation methods for reference. The list is not meant to be exhaustive, but rather to provide examples and commonly used methods for estimating values.[6] The sections below provide a more detailed discussion of each of the benefit categories listed in Table 7.1.

7.2.1 Human Health Improvements

Human health improvements from environmental policies include effects such as reduced mortality rates, decreased incidence of non-fatal cancers, chronic conditions and other illnesses, and reduced adverse reproductive or developmental effects. While the most appropriate approach to valuation would consider mortality and morbidity together, in practice these effects are typically valued separately, and are therefore discussed separately in these *Guidelines*.

7.2.1.1 Mortality

Some EPA policies will lead to decreases in human mortality risks due to potentially fatal health conditions such as cancers. In considering the impact of environmental policy on mortality, it is important to remember that environmental policies do not assure that particular individuals will not die of environmental causes. Rather, they lead to small changes in the probability of death for many people.

EPA currently recommends a default central "value of statistical life" (VSL) of $7.9 million (in 2008 dollars) to value reduced mortality for all programs and policies.[7] This value is based on a distribution fitted to 26 published VSL estimates. The distribution itself can be used in uncertainty analysis. The underlying studies, the distribution parameters, and other useful information are available in Appendix B.

As a general matter, the impact of risk and population characteristics should be addressed qualitatively. In some cases, the analysis may include a quantitative sensitivity analysis. Analysts should account for latency and cessation lag when valuing reduced mortality risks, and should discount appropriately.

Valuing mortality risk changes in children is particularly challenging. EPA's *Handbook for Valuing Children's Health Risks* (2003b) provides some information on this topic, including key benefit transfer issues when using adult-based studies. *Circular A-4* also recognizes this subject, specifically advising: "For rules where health gains are expected among both children and adults and you decide to perform a BCA, the monetary values for children should be at least as large as the values for adults (for the same probabilities and outcomes) unless there is specific and compelling evidence to suggest otherwise" (OMB 2003). OMB guidance applies to risk of mortality and of morbidity.

5 See Section A.3 of Appendix A for further explanation of Kaldor-Hicks conditions.

6 In very rare cases with employment implications for the structurally unemployed, analysts may need to include job creation as a benefits category. See Appendix C for more detail.

7 This value is adjusted from the base value reported in U.S. EPA 2000d ($4.8 million in 1990 dollars) using the Consumer Price Index (CPI). The value is not adjusted for income growth over time.

Table 7.1 - Types of Benefits Associated With Environmental Policies: Categories, Examples, and Commonly Used Valuation Methods

Benefit Category	Examples	Commonly Used Valuation Methods
Human Health Improvements		
Mortality risk reductions	Reduced risk of: Cancer fatality Acute fatality	Averting behaviors Hedonics Stated preference
Morbidity risk reductions	Reduced risk of: Cancer Asthma Nausea	Averting behaviors Cost of illness Hedonics Stated preference
Ecological Improvements		
Market products	Harvests or extraction of: Food Fuel Fiber Timber Fur and Leather	Production function
Recreation activities and aesthetics	Wildlife viewing Fishing Boating Swimming Hiking Scenic views	Production function Averting behaviors Hedonics Recreation demand Stated preference
Valued ecosystem functions	Climate moderation Flood moderation Groundwater recharge Sediment trapping Soil retention Nutrient cycling Pollination by wild species Biodiversity, genetic library Water filtration Soil fertilization Pest control	Production function Averting behaviors Stated preference
Non-use values	Relevant species populations, communities, or ecosystems	Stated preference
Other Benefits		
Aesthetic improvements	Visibility Taste Odor	Averting behaviors Hedonics Stated preference
Reduced materials damages	Reduced soiling Reduced corrosion	Averting behaviors Production / cost functions

Note: "Stated preference" refers to all valuation studies based on hypothetical choices, as distinguished from "revealed preference," which refers to valuation studies based on observations of actual choices.

Methods for valuing mortality risk changes

Because individuals appear to make risk-income trade-offs in a variety of ways, the value of mortality risk changes are estimated using three primary methods. The most commonly used method is the hedonic wage, or wage-risk, method in which value is inferred from the income-risk trade-offs made by workers for on-the-job risks. Averting behavior studies value risk changes by examining purchases of goods that can affect mortality risk (e.g., bicycle helmets). Finally, stated preference studies are increasingly used to estimate WTP for reduced mortality risks. Key considerations in all of these studies include the extent to which individuals know and understand the risks involved, and the ability of the study to control for aspects of the actual or hypothetical transaction that are not risk-related. Because the value of risk reduction may depend on the risk context (e.g., work-related vs. environmental), results from any single study may not be directly applicable to a typical environmental policy case.

There are additional methods that can be used to derive information on risk trade-offs. Van Houtven et al. (2008) use a risk-risk trade-off model to examine preferences for avoiding fatal cancers. Carthy et al. (1999) examine trade-offs between fatal and non-fatal risks to indirectly estimate a WTP. This approach may make the task more manageable for the respondent, but the analyst should consider and evaluate the complexity of the additional steps and the indirect nature of the resulting estimates.

At one time, reduced mortality risk was valued under a human capital approach that equated the value of a statistical life with foregone earnings. This has largely been rejected as an inappropriate measure of the value of reducing mortality risks because it is not based on WTP for small risk reductions and as such does not capture the value associated with avoided pain and suffering, dread, and other risk factors that are thought to affect value (Viscusi 1993).

Previous studies

While there are many unresolved issues in valuing mortality risks, the field is relatively rich in empirical estimates and several substantial reviews of the literature are available. A general overview of common approaches and issues in mortality risk valuation can be found in Hammitt (2003). Viscusi (1993) and Viscusi and Aldy (2003) provide detailed reviews of the hedonic wage literature. Black, Galdo, and Liu (2003) provide a technical review of the statistical issues associated with hedonic wage studies. Blomquist (2004) provides a review of the averting behavior literature. Some key issues related to stated preference studies are included in Alberini (2004). Recently, some researchers have begun to use meta-analysis to combine study results and examine the impact of study design. Recent examples include Viscusi and Aldy (2003), Mrozek and Taylor (2002), and Kochi et al. (2006). EPA applications of VSL are numerous, and include the Clean Air Interstate Rule (CAIR), the Non-Road Diesel Rule, and the Stage 2 Disinfection By-Products Rule (DBP).[8]

Important considerations

The analyst should keep three important considerations in mind when estimating mortality benefits:

- Characterizing and measuring mortality effects;
- Heterogeneity in risk and population characteristics; and
- The timing of health risk changes.

Characterizing and measuring mortality effects

Reduced mortality risks are typically measured in terms of "statistical lives." This measure is the

8 The economic analyses for these three rules are available electronically as follows (accessed May 23, 2008):

CAIR (http://www.epa.gov/air/interstateairquality/pdfs/finaltech08.pdf);

Non-Road Diesel (http://www.epa.gov/nonroad-diesel/2004fr.htm#ria); and

Stage 2 DBP (http://www.epa.gov/safewater/disinfection/stage2/pdfs/anaylsis_stage2_ecconomic_main.pdf).

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 shows that the policy leads to 10 premature fatalities averted, or 10 statistical lives "saved."

Alternative measures attempt to capture the remaining life expectancy associated with the risk reductions. This is sometimes referred to as the "quantity of life" saved (Moore and Viscusi 1988) and is typically expressed as "statistical life years." Looking again at the policy described above, suppose the risks were spread over a population where each individual had 20 years of remaining life expectancy. The policy would then save 200 statistical life years (10 statistical lives times 20 life years each). In practice, estimating statistical life years saved requires risk information for specific subpopulations (e.g., age groups or health status). It is typical to use statistical life years saved in CEA, but valuing a statistical life year remains a subject of debate in the economics literature. Theoretical models show that the relationship between WTP and factors such as age, baseline risk, and the presence of co-morbidities is ambiguous and empirical findings are generally mixed (U.S. EPA 2006e).

Heterogeneity in risk and population characteristics

The value of mortality risks can vary both by risk characteristics and by the characteristics of the affected population. Key risk characteristics include voluntariness (i.e., whether risks are voluntarily assumed), timing (immediate or delayed), risk source (e.g., natural vs. man-made), and the causative event (e.g., cancer vs. accidents). Population characteristics include those generally expected to influence WTP for any good (e.g., income and education) as well as those more closely related to mortality risks such as baseline risk or remaining lifespan, health status, risk aversion, and familiarity with the type of risk. The empirical and theoretical literature on many of these characteristics is incomplete or ambiguous. For example, some studies suggest that older populations are willing to pay less for risk reductions (Jones-Lee et al. 1993), but others find this effect to be small if it exists at all (Alberini et al. 2004). Still others suggest older populations have higher WTP (Kniesner, Viscusi, and Ziliak 2006). Smith et al. (2004) and Viscusi and Aldy (2007a) discuss the relationship between age and VSL in the context of hedonic wage studies. Appendix B contains a more complete discussion of risk and population characteristics and how they may affect WTP.

Timing of health risk changes

Environmental contamination can cause immediate or delayed health effects. If individuals typically prefer health improvements earlier in time rather than later, all else equal, then the WTP for reductions in exposure to environmental pollutants will depend on when the resulting health risk changes will occur. The description here focuses on mortality risk, but the same principles apply to non-fatal health risks.

The effects of timing on the present or annualized value of reduced mortality risk can be considered in the context of a lifecycle consumption model with uncertain lifetime (Cropper and Sussman 1990, Cropper and Portney 1990, and U.S. EPA 2007g). In this framework reductions in mortality risk are represented as a shift in the survival curve — the probability an individual will survive to all future ages — which leads to a corresponding change in life expectancy and future utility.

If the basis for benefit transfer is a marginal WTP for contemporaneous risk reductions, then calculating the benefits of a policy with delayed risk reductions requires three steps: (1) estimating the time path of future mortality risk reductions; (2) estimating the annual WTP in all future years; and (3) calculating the present value of these annual WTP amounts. The first step should account for all the factors that ultimately relate changes in exposure to changes in mortality risk as defined by shifts in the survival curve.

7.2.1.2 Morbidity

Morbidity benefits consist of reductions in the risk of non-fatal health effects ranging from mild illnesses, such as headaches and nausea, to very serious illnesses such as cancer (see Table 7.1). Non-fatal health effects also include conditions such as birth defects or low birth weight. Non-fatal health effects differ with respect to the availability of existing value estimates. Values for reducing the risks of some of these health effects have been estimated multiple times using a variety of different methods, while others have been the subject of only a few or no valuation studies.

WTP to reduce the risk of experiencing an illness is the preferred measure of value for morbidity effects. As described in Freeman (2003), this measure consists of four components:

- "Averting costs" to reduce the risk of illness;
- "Mitigating costs" for treatments such as medical care and medication;
- Indirect costs such as lost time from paid work, maintaining a home, and pursuing leisure activities; and
- 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. For example, cost-of-illness (COI) estimates generally only capture mitigating and indirect costs, and omit averting expenditures and lost utility associated with pain and suffering. [9]

Methods for valuing morbidity

Researchers have developed a variety of methods to value changes in morbidity risks. Some methods measure the theoretically preferred value of individual WTP to avoid a health effect. Others can provide useful data, but that data must be interpreted carefully if it is to inform economically meaningful measures. Methods also differ in the perspective from which values are measured (e.g., before or after the incidence of morbidity), whether they control for the opportunity to mitigate the illness (e.g., before or after taking medication) and the degree to which they account for all of the components of total WTP. The three primary methods most often used to value morbidity in an environmental context are stated preference (Section 7.3.2), averting behavior (Section 7.3.1.4), and COI (Section 7.3.1.5). Hedonic methods (Section 7.3.1.3) are used less frequently to value morbidity from environmental causes.

Many other approaches do not estimate WTP and their ability to inform benefits analyses consequently varies. Risk-risk trade-offs, for example, do not directly estimate dollar values for risk reductions, but rather provide rankings of relative risks based on consumer preferences. Risk-risk trade-offs can be linked to WTP estimates for related risks.[10]

Other methods suffer from certain methodological limitations and are therefore generally less useful for policy analysis. For example, health-state indices, composite metrics that combine information on quality and quantity of life lived under various scenarios, are often used for cost-effectiveness or cost-utility analyses. These methods cannot be directly related to WTP estimates as the indices were developed using very different paradigms than those for WTP values. As such, they should not be used for deriving monetary estimates for use in BCA [Hammitt 2003, and Institute of Medicine (IOM) 2006], although there is evidence that components of these indices may still be useful in a benefit-transfer context (Van Houtven et al. 2006). Another commonly suggested alternative is jury awards, but these generally should *not* be used in benefits analysis, for reasons explained in Text Box 7.3.

9 This is why COI estimates generally understate WTP to reduce the same risk or avoid a given health effect. Some studies have estimated that total WTP can be two to four times as large as COI even for minor acute respiratory illnesses (Alberini and Krupnick 2000). Still, there is not any broadly applicable "scaling factor" that relates COI to WTP generally.

10 EPA analyses have used risk-risk trade-offs for non-fatal cancers in conjunction with VSL estimates as one method to assess the benefits of reduced carcinogens in drinking water (U.S. EPA 2005a).

Text Box 7.3 - Non-Willingness to Pay Measures

Economic measures of value calculate willingness to pay (WTP) for environmental changes. WTP is defined as that amount of money that, if taken away from income, would make an individual exactly indifferent between experiencing an environmental improvement and not experiencing either the improvement or any change in income. (An analogous measure can also be constructed for "not experiencing degradation" rather than "experiencing an improvement"). WTP is a valid measure of "economic value" because it is directly useful for applying the potential compensation tests of Kaldor and Hicks.

Some measures of economic value are not valid, as they do not measure WTP, and cannot be related to changes in utility. Others should be used only in a limited set of circumstances. Some examples are provided below.

Replacement cost. One of the common consequences of environmental deterioration is damage to assets. Some analysts have suggested that the economic value of the damage is the cost of replacing the asset. This will only be true if: (1) damage to the asset is the only cost of the environmental deterioration; and (2) the least expensive way to achieve the level of satisfaction realized before the deterioration would be to replace the asset. If the first condition is not met, consideration of replacement costs alone might underestimate the economic consequences of environmental degradation. If the second condition is not met, replacement costs might overestimate the consequences. Suppose that water pollution kills fish in a pond. Replacing those fish with healthy, edible ones might prove extremely expensive: the pond might need to be dredged and restocked. However, people who are no longer able to catch fish in the pond might be compensated simply by giving them enough money to purchase substitutes at their local supermarket.

Proxy costs. A closely related concept to replacement cost is the cost of a substitute for the damaged asset. In widely cited work, ecologist H.T. Odum (1996) calculated the number of barrels of petroleum that would be required to provide the energy to replace the services of wetland ecosystems. However, this number is economically irrelevant. There is no reason to suppose that people would choose to replace services of damaged wetlands with those of purchased oil. A similar argument can be made against the interpretation of "ecological footprints" as an estimate of economic consequences (Wackernagel and Rees 1996). Dasgupta (2002) interprets these approaches as single-factor theories of value (Karl Marx's labor theory of value is the best known example), fallacies that were disproved in general by Samuelson's (1951) "non-substitution theorem."

Cost-of-illness (COI). Health effects are often proxied by the "cost of illness," which are the total costs of treatment and time lost due to illness. Although COI is discussed in greater detail in Section 7.3.1.5, note here that: (1) COI does not record other expenses incurred in efforts to avoid illness; (2) health insurance may drive a wedge between the costs incurred to treat illness and WTP to avoid it; and (3) COI ignores factors such as discomfort and dread that patients would also be willing to pay to avoid.

Jury awards. Another approach sometimes taken to measure environmental damages is derived from the awards made by juries. Using such awards may also prove problematic for at least two reasons. First, cases only go to trial if both sides prefer the risk of an adverse outcome to the certainty of a pre-trial settlement. Cases that go to juries are "atypical" by definition. Second, since adjudication does not always occur and can never be infallible, jury awards often do, and arguably should (Shavell 1979), embody "punitive" as well as "compensatory" elements. Juries make examples of guilty defendants in an attempt to deter others from committing similar offenses. For this reason, jury awards may overstate typical damages. Finally, jury awards reflect a certain outcome and not the probability of experiencing an adverse event and therefore include the influence of characteristics typically not included in statistical analysis, such as pain, suffering, and likeability. These estimates are not appropriate for application to ex ante evaluation of the value associated with a statistical probability.

Previous studies

A comprehensive summary of existing studies of morbidity values is beyond the scope of these *Guidelines*. Below is a short list of references that can serve as a starting point for reviewing available morbidity value estimates for benefit transfer or for designing a new study. Some recent estimates for particular health effects include Hammitt and Haninger (2007), who examine food-related illnesses, and Chestnut et al. (2006), who examine respiratory and cardiovascular effects. Tolley et al. (1994) and Johanneson (1995) are useful general references for valuing non-fatal health effects. EPA's *Handbook for Non-Cancer Valuation* (U.S. EPA 2000c) provides published estimates for many illnesses and reproductive and developmental effects. 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. EPA's *Cost of Illness Handbook* (U.S. EPA 2007c) includes estimates for many cancers, developmental illnesses, disabilities, and other conditions. EPA analyses of regulations and policies, including EPA's two comprehensive studies of the benefits and costs of the Clean Air Act (U.S. EPA 1997a and U.S. EPA 1999) draw upon a number of existing studies to obtain values for reductions of a variety of health effects. These sources describe how the central estimates were derived, and attempt to quantify the uncertainty associated with using the estimates.

At least two meta-analyses have attempted to examine how the value of non-fatal risk reductions varies with characteristics of the condition, the affected population, and the approach to valuation. Vassanadumrongdee et al. (2004) focus on air pollution-related morbidity risks and posit a meta-regression based benefit transfer function. Van Houtven et al. (2006) evaluate more than 230 WTP estimates from 17 stated preference studies, finding evidence that illness severity, measured systematically, is a significant factor explaining variation in WTP. The authors also illustrate how a meta-regression-based function can facilitate benefit transfer based on duration and severity of acute illnesses, along with population characteristics. While the specific benefit-transfer functions in these articles might not be suitable for application in any particular context, the estimates contained in them can be helpful. Other studies are available through the Environmental Valuation Reference Inventory (EVRI). EVRI is maintained by Environment Canada and contains more than 1,100 studies that can be referenced according to medium, resource, stressor, method, and country.[11]

Important considerations

The analyst should keep two important considerations in mind when estimating morbidity benefits:

- Characterizing and measuring morbidity effects; and
- Incomplete estimates of WTP.

Characterizing and measuring morbidity effects

The key characteristics that will influence the values of morbidity effects are their severity, frequency, duration, and symptoms. Severity defines the degree of impairment associated with the illness. Examples of how researchers have measured severity include "restricted activity days," "bed disability days," and "lost work days."[12] Severity can also be described in terms of health state indices that combine multiple health dimensions into a single measure.[13] For duration, 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 illnesses. The

11 See www.evri.ca for more information.

12 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 require 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.

13 The difference in the indices is intended to reflect the relative difference in disutility associated with symptoms or illnesses. There are serious questions about the theoretic and empirical consistency between these "health-related quality of life" index values and WTP measures for improved health outcomes (Hammitt 2002). Still the inclusion of some aspects of these indices may prove useful in valuation studies (Van Houtven et al. 2006). Examples of economic analyses that have employed some form of health state index include Desvousges et al. (1998) and Magat et al. (1996).

frequency of effects also can vary widely across illnesses. Some effects are one-time events that are unlikely to recur, such as a gastrointestinal illness. Other effects, such as asthma, do recur or can be aggravated regularly, causing disruptions in work, school, or recreational activities.

For chronic conditions or more serious outcomes, morbidity effects are usually measured in terms of the number of expected cases of a particular illness. Given the risks faced by each individual and the number of people exposed to this risk, an estimate of "statistical cases" can be defined analogously to "statistical lives." In contrast, morbidity effects that are considered acute or mild in nature can be estimated as the expected number of times a particular symptom associated with an illness occurs. These estimates of "symptom days" may be used in benefits analysis when appropriate estimates of economic value are available, although a richer characterization of combinations of symptoms, severity, duration, and episode frequency would be an improvement over much of the existing literature. Some studies have attempted to deal with these complexities in a more systematic manner, but the results have not yet been widely applied and interpreted for policy analysis (Cameron and DeShazo 2008). (Refer to Section 7.3.1.5 and Text Box 7.3 on the use of COI versus WTP measures of value.)

Incomplete estimates of WTP

The widespread availability of health insurance and paid sick leave shift some of the costs of illness from individuals to others. While this cost-shifting can be addressed explicitly in COI studies, it may lead to problems in estimating total WTP. If the researcher does not adequately address these concerns, individuals may understate their WTP, assuming that some related costs would be borne by others. However, to the extent that these costs represent diversions from other uses in the economy, they represent real costs to society and should be accounted for in the analysis.

More information on these and other issues to consider when conducting or evaluating morbidity value studies is provided in EPA's *Handbook for Non-Cancer Health Effects Valuation* (U.S. EPA 2000c).

7.2.2 Ecological Benefits

In addition to human health benefits, many EPA policies will produce ecological benefits by increasing the delivery of "ecosystem services," which are the end products of ecological functions that are valued by people (Daily 1997, National Research Council 2005, and Millennium Ecosystem Assessment 2005). There is a large and growing literature on the valuation of ecosystem services. Fisher et al. (2009) document an exponentially increasing number of published articles on ecosystem services, growing from essentially none in the early 1980s to around 250 in 2007. Much of this literature focuses on the impacts of habitat loss and other land use changes on ecosystem service flows. Because EPA has only limited authority over private and public land use decisions, analysts may find that only a subset of the results in these studies will be directly transferable to traditional EPA regulations. Nevertheless, this growing literature can provide a useful conceptual framework and potentially transferable methods for analyzing a wide range of EPA policies that may affect ecological services.

In principle, once the pollutants (or other environmental stressors) whose emissions will be altered by the regulation have been identified, the same general approach used to estimate human health benefits can be used to estimate ecological benefits: identify the endpoints that are affected by those pollutants and that are valuable to society; estimate dose-response relationships between stressors and endpoints; and estimate people's WTP for changes in the endpoints using revealed or stated preference valuation methods. In the case of ecological benefits estimation, the relevant endpoints will include measures of ecosystem health rather than human health, and the methods and data required to estimate the dose-response functions and WTP will differ accordingly. As in the human health case, the estimation of dose-response relationships between pollutants and endpoints will fall mainly to natural scientists, although collaboration between scientists and economists often is needed to help focus the analysis on the most important endpoints. [The Agency's *Ecological Benefits Assessment Strategic Plan* describes an interdisciplinary approach for conducting ecological benefits assessments, as well as research priorities for improving such assessments (U.S. EPA 2006a)]. Even though the basic approach for valuing ecological benefits is similar to that used to value human health benefits, an entirely different set of complications may arise when estimating ecological benefits (Freeman 2003 pp. 457-460). Some of these complications are explored below.

A hypothetical policy

To illustrate some of the complications that can arise when assessing ecological benefits, consider a hypothetical policy that would control the emissions of an industrial chemical that are believed to decrease survival and reproductive rates in one or more fish species. First, compared to the commonly accepted individual-level mortality and morbidity endpoints used in human health benefit assessments, it may be more difficult to identify or define the relevant endpoints in an ecological benefits assessment (de Groot et al. 2002, Boyd and Banzhaf 2007, Wallace 2007, and Fisher and Turner 2008). Identifying endpoints for estimating use values may be relatively straightforward. For example, endpoints for this hypothetical policy would include the abundances and distributions of species that are directly or indirectly affected by the chemical and are harvested or targeted for wildlife viewing or other non-consumptive outdoor activities. Identifying relevant endpoints for non-use values, on the other hand, can be more complicated. Even for this simplified hypothetical policy, it may not be clear which among the wide variety of measureable ecosystem attributes — beyond those previously identified as relevant for use values — would provide an adequate basis for eliciting non-use values in a stated preference survey. Evans et al. (2008) discuss some of the challenges they faced in defining endpoints for a stated preference survey to value reductions in acid rain in the Adirondacks. Boyd and Krupnick (2009) discuss problems of identifying ecological endpoints more generally.

After relevant endpoints are identified, there may be additional complications in modeling the effects of the chemical on those endpoints. For example, the emissions-transport-exposure pathway(s) — i.e., the "ecological production function" (U.S. EPA 2009b) — may involve complex food web linkages that are less direct or have more convoluted feedbacks than in the human health context. Furthermore, some of the important feedbacks may involve human responses to the changed ecological conditions. For example, if some of the fish species in our hypothetical policy scenario are harvested by recreational or commercial fishers, then the nature of the management regime in the fisheries may influence the response of the fish stocks to the policy. In an extreme case, if the commercial fisheries are completely unregulated and subject to open access conditions, then any increases in the stock sizes from the policy may be completely offset in the long run by new entrants to the fishery (Freeman 1991, Barbier et al. 2002, Smith 2007, and Newbold and Iovanna 2007). Therefore, an integrated bio-economic modeling approach may be needed to accurately project the bio-physical effects of the policy. Some examples of such an approach include Smith and Crowder (2006), Massey et al. (2006), and Finnoff and Tschirhart (2008).

After the ecological effects of the policy are characterized, there may be further complications in valuing those effects. For this hypothetical policy, the main requirement for revealed preference valuation methods might be data on commercial and recreational fishing activities in the affected water bodies. Other recreational activities also might be affected, and water-related amenities might influence property values. As with human health benefits, care must be taken to avoid double counting when using multiple datasets and methods that could include overlapping values (McConnell 1990, and Phaneuf et al. 2008). Furthermore, if a significant portion of the benefits for ecological changes are thought to consist of non-use values rather than use values, analysts may need to rely more heavily on stated preference methods when estimating ecological benefits. Considering the challenges in conducting reliable stated preference valuation studies even for well-defined and familiar commodities (described in detail in Section 7.3.2), this compounds the extra complications already discussed. This also points to a larger potential role for non-monetized and non-quantified benefits in the overall analysis (U.S. EPA 2006a, and U.S. EPA 2009b).

Application of economic valuation methods to ecological changes

Extensive treatments of the valuation of ecosystem services can be found in recent reports from the National Academy of Science (NAS) (2005) and EPA's SAB Committee on the Valuation of Ecological Systems and Services (U.S. EPA 2009b). Analysts are referred to these reports for more detailed discussions on the application of economic valuation methods to ecological benefits than are provided in these *Guidelines*. In this section are examples of studies that apply traditional valuation methods (discussed more generally in the following sections of this chapter) to ecosystem goods and services. Some of the special complications that can arise in such studies are highlighted.

Production functions

A number of recent contributions to the literature on valuing of ecosystem services emphasize the importance of understanding the production functions relating natural systems to the provision of products that are valuable to people (Polasky et al. 2008a, 2008b; Boyd and Banzhaf 2007; and U.S. EPA 2009b). Some simple examples have long been known: commercially valuable species "produce" themselves. Early work such as Faustmann's 1848 analysis of optimal rotations in forestry (see also Samuelson 1976), Clark's (1990) work in fisheries, and Hammack and Brown's (1974) work on wetlands and waterfowl have provided templates for later studies. It may be possible to value the effects of pollution on the exploitation of renewable resources when biological production possibilities are affected by environmental conditions — for example, when fish stocks are affected by water quality, or when waterfowl populations are affected by the extent and configuration of wetlands (Bell 1997, Ellis and Fischer 1987, and Massey et al. 2006). As discussed above, analysts should keep in mind that institutional features such as open access to renewable resources may dissipate values that might otherwise be realized from environmental improvement.

Ecological resources also can contribute to the production of other useful goods and services, such as crop yields, groundwater quality, and surface water flow characteristics. Hence the degradation of supporting ecological resources should be reflected in diminished outputs of these commodities. Direct application of production function approaches often is hampered by data and methodological limitations. Specifically, it can be difficult to measure the flow of non-market ecosystem services that a particular production process receives, as well as to statistically control for the effects of unobserved characteristics of climate and topography. One approach is to design observational studies to mimic controlled experiments as closely as possible. Ricketts et al. (2004) use this approach in a study of the value of pollination services to coffee crops. In some cases production functions might plausibly be derived from first principles. For example, Weitzman (1992), Simpson et al. (1996), Rausser and Small (2000), and Costello and Ward (2006) use simple probability models to examine the role of biodiversity in the development of new pharmaceutical products. Further examples of studies relating ecological conditions to economic outputs through production processes include Acharya and Barbier (2002), who examine ground water recharge as a function of surrounding land cover, and Pattanayak and Kramer (2001), who examine stream flow as a function of land cover.

Hedonic models

Econometricians generally have favored estimating cost or profit functions to estimating production functions. This is because the prices that are the arguments of the former will be uncorrelated with unobserved factors, whereas input choices will not (see Varian 1992). While a cost or profit function approach could be adopted in the estimation of ecosystem service values, a more common, and theoretically equivalent, approach is to estimate a hedonic price function. In theory, the rental price of land is equal to the earnings that could be derived from its use, while the purchase price is equal to the net discounted value of the stream of such earnings. A number of authors have estimated hedonic models relating the value of residential properties to the proximity and attributes of nearby forests (Anderson and Cordell 1988, Tyrväinen and Miettinen 2000, and Willis and Garrod 1991), wetlands (Lupi et al. 1991, Mahan et al. 2000, Woodward and Wui 2001, Bin and Polasky 2005, and Costanza et al. 2008), or other varieties of "open space" (Geoghegan et al. 1997, Benson et al. 1998, Irwin and Bockstael 2002, Irwin 2002, and Thorsnes 2002).

Travel cost models

A large number of studies use travel cost models to value ecological endpoints. The predominant activity in the recreational use value literature has been fishing; where the ecological endpoint is expected fish catch (or one or more proxy measures thereof) at one or more recreation sites. For example, 122 of 325 studies in the recreational use value database assembled by Rosenberger and Stanley (2007) focused on either freshwater or saltwater recreational fishing. The remaining studies in the database focus on one of 25 other categories of activities, including bird watching (Hay and McConnell 1979), wildlife hunting (Creel and Loomis 1990, Coyne and Adamowicz 1992, Boxall 1995, Peters et al. 1995, and Adamowicz et al. 1997), beach use (Bockstael et al. 1987a, and Parsons and Massey 2003), backcountry recreation (Boxall et al. 1996), rock-climbing (Shaw and Jakus 1996), and kayaking (Phaneuf and Siderelis 2003).

Stated preference methods

Revealed preference methods cannot capture non-use values, such as those associated with the existence of biological diversity. This is because it is not possible to use data on market transactions or any other observed choices to estimate the value of goods that leave no "behavioral trail" (Larson 1993) in their enjoyment. In such cases only stated preference methods can provide estimates of WTP or WTA (Freeman 2003). More generally, stated preference methods may be employed when researchers want to identify the widest possible spectrum of values, both use and non-use (Loomis et al. 2000).

Stated preference studies have been used to value a number of ecosystem services. Examples include the protection of endangered species (Brown and Shogren 1998), the ecological consequences of water quality improvements in Europe (Hanley et al. 2006), improved ecological conditions resulting from reduced air pollution in the United States (Banzhaf et al. 2006), and restoration of the Florida Everglades (Milon and Scrogin 2006). In some instances researchers may want to combine results of stated preference valuation studies of particular ecological endpoints with other data on the effects of pollution, land use, or other factors on the production of ecosystem services. See Boyd and Krupnick (2009) for an extended discussion.

Complications that may apply to all methods

When using these valuation methods or when transferring the results of previous valuation studies to assess ecological benefits for new policy cases, analysts should be prepared to confront several complications. For example:

For new studies, it may be difficult to identify and/or measure the ecological endpoints that are most relevant for the policy case. Without a set of observable measures of ecological conditions (or measures that can be linked to ecological conditions through supplemental bio-physical modeling) thought to be relevant for outdoor recreation behavior, housing decisions, etc., it will not be possible to use revealed preference methods to value ecological effects. For example, users may care mainly about water clarity for a certain type of recreational activity, while the most readily available data might measure nutrient loading in the water bodies that would be affected by a policy change. Under such circumstances it may be difficult to relate revealed preferences regarding housing decisions, recreational behavior, etc., to the available nutrient loading data, as those data are imperfect proxies for water clarity. There are well-known statistical pitfalls associated both with specifying the wrong "right-hand side" variables in an econometric relationship, as well as with "data mining" by including right-hand side variables in the absence of theoretical justification. The best, if not always practicable, advice that can be given is to think as carefully as possible about which variables should motivate choices before running any regressions.

For benefit transfers, it may be difficult to find existing studies that value ecological endpoints that are the same as, or sufficiently similar to, those of interest in the policy case. This problem is likely to be more common for ecological benefits than for human health benefits because the latter has a larger set of studies to draw from and a smaller set of common endpoints that have been used in multiple studies. The less similar are the commodities valued in the existing ecological benefit studies, the more difficult it will be to synthesize those studies in a meta-analysis or preference calibration exercise, and the less valid will be the transfer of the resulting value estimate or function.

Estimation difficulties are likely to arise in many cases of interest. In particular, explanatory variables may not meet the exogeneity requirement for estimating their associated coefficients. For example, in performing hedonic regressions of property prices on, among other things, the development status of nearby properties, it is likely that both the price of the property in question and the use made of nearby properties would be determined by factors that cannot be observed by the econometrician (Irwin and Bockstael 2002, and Irwin 2002). Similarly, in estimating recreation demand models in which a recreationist's decision to visit a particular site depends on, among other things, congestion (i.e., how many others decide to visit the site at the same time), it is likely that *all* recreationists' site visit choices will be influenced by the same unobserved factors (Timmins and Murdock 2007). Similar difficulties arise in other areas of economics; for example Durlauf's (2004) survey of empirical approaches to "neighborhood effects" in urban economics. The solution in each instance is to identify appropriate instrumental variables, but this can be difficult in many cases. One way around such problems may be to identify "natural experiments." Thorsnes (2002), for example, identifies instances in which historical accidents influenced land use patterns independently of the later realization of adjacent land value in order to conduct a hedonic study of the effects of open space.

For resources subject to consumptive use, such as harvested fish or wildlife species, expected harvest levels are endogenous variables, which can lead to biases similar to that introduced by congestion effects. If the policy of interest leads to spatially heterogeneous environmental quality improvements, then it may lead to a re-sorting not only of recreators but also of the target species among the recreation sites. Ignoring this spatial re-sorting effect can give biased welfare estimates (Newbold and Massey 2010). This can complicate both the estimation of preference parameters and the transfer of the estimated preference function to the policy case.

A basic goal of any benefits assessment is to count all categories of benefits, but to count each only once. This may be particularly important for ecological benefits assessments since stated preference studies employed to estimate intangible values, such as existence values of biodiversity, might also capture use values that are already covered by revealed preference studies such as recreation demand or hedonic studies. When combining values estimated using multiple methods, the analyst should take care to avoid double counting.

It is important to identify and discuss any omitted benefit categories that are thought to be important but that cannot be monetized, or possibly even quantified. There may be circumstances in which provision of some additional information may be helpful, even if does not rise to the level of presenting an explicit comparison of benefits with costs. For example, analysts may be able to identify the most cost-effective approach among different alternatives, or to present natural science information that can convey the biophysical impact of a policy even if it does not quantify the WTP or WTA for such a policy. It is better to acknowledge gaps in information by discussing them qualitatively or by reporting physical measures (if available) than to employ conceptually flawed methods of monetization. In particular, analysts should avoid the use of replacement cost, embodied energy-based evaluation methods, or ecological footprint analysis to derive estimates of WTP or WTA.

7.2.3 Other Benefits

Other types of potential benefits from environmental policies include aesthetic improvements and reduced material damages.

Aesthetic improvements include effects such as improved taste and odor of tap water resulting from water treatment requirements and enhanced visibility resulting from reduced air pollution. EPA typically considers two types of benefits from increased visibility due to improvements in air quality: residential visibility benefits and recreational visibility benefits. Improvements in residential visibility are typically assumed to only benefit residents living in the areas in which the improvements are occurring, while all households in the United States are usually assumed to derive some benefit from improvements in visibility in areas such as National Parks. The benefits received, however, are assumed to decrease with the distance from the recreational area in which the improvements occur.

Reduced materials damages include welfare impacts that arise from changes in the provision of service flows from human-made capital assets such as buildings, roads, and bridges. Materials damages can include changes in both the quantity and quality of such assets. Benefits from reduced material damages typically involve cost savings from reduced maintenance or restoration of soiled or corroded buildings, machinery, or monuments.

Methods and previous studies

Changes in the stock and quality of human-made capital assets are assessed in a manner similar to their "natural capital" counterparts. Analytically, the valuation of reduced materials damages parallels the methods for valuing the tangible end products from managed ecosystems such as agriculture or forestry. Effects from changes in air quality on the provision of the service flows from physical resources 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.

Linking changes in environmental quality with the provision of service flows from materials can be difficult because of the limited scientific understanding of the physical effects, the timing of the effects, and the behavioral responses of producers and consumers. An analysis of reduced materials damages typically begins with an environmental fate and transport model to determine the direct effects of the policy on the stocks and flows of pollutants in the environment. Then stressor-response functions are used to relate local concentrations of pollutants to corrosion, soiling, or other physical damages that affect the production (inputs) or consumption (output) of the material service flows. The market response to these impacts serves as the basis for the final stage of the assessment, in which some type of structural or reduced-form economic model that relates averting or mitigating expenditures to pollution levels is used to value the physical impacts. The degree to which behavioral adjustments are considered when measuring the market response is important, and models that incorporate behavioral responses are preferred to those that do not. Adams and Crocker (1991) provide a detailed discussion of this and other features of materials damages benefits assessment. Also see EPA's benefits analysis of household soiling for an example that employs a reduced-form economic model relating defensive expenditures to ambient pollution (U.S. EPA 1997f).

7.3 Economic Valuation Methods for Benefits Analysis

For goods bought and sold in undistorted markets, the market price indicates the marginal social value of an extra unit of the good. There are virtually no markets for environmental goods. While some natural products are sold in private markets, such as trees and fish, these are "products of the environment" and not the types of "environmental goods and services" analysts typically need to value. The analyst's concern is typically with *non-market* inputs, which are, by definition, not traded in markets.[14] To overcome this lack of market data, economists have developed a number of methods to value environmental quality changes. Most of these methods can be broadly categorized as either revealed preference or stated preference methods.

In cases where markets for environmental goods do not exist, WTP can often be inferred from choices people make in related markets. Specifically, because environmental quality is often a characteristic or component of a private good or service, it is sometimes possible to disentangle the value a consumer places on environmental quality from the overall value of a good. Methods that employ this general approach are referred to as *revealed preference methods* because values are estimated using data gathered from observed choices that reveal the preferences of individuals. Revealed preference methods include production or cost functions, travel cost models, hedonic pricing models, and averting behavior models. This section also discusses COI methods, which are sometimes used to value human health effects when estimates of WTP are unavailable.

In situations where no markets for environmental or related goods exist to infer WTP, economists sometimes rely on survey techniques to gather choice data from hypothetical markets. The methods that use this type of data are referred to as *stated preference methods* because they rely on choice data that are stated in response to hypothetical situations, rather than on choice behavior observed in actual markets. Stated preference methods include contingent valuation, conjoint analysis, and contingent ranking.

Each of these revealed and stated preference methods is discussed in detail below. Included are an overview of each method, a description of its general application to environmental benefits analysis, and a discussion of issues involved in interpreting and understanding valuation studies. The discussion concludes with a separate overview of benefit-transfer methods. It is important to keep in mind that research on all of these methods is ongoing. The limitations and qualifications described here are meant to characterize the state of the science at the time these *Guidelines* were written. Analysts should consult additional resources as they become available.

7.3.1 Revealed Preference Methods

A variety of revealed preference methods for valuing environmental changes have been developed and are widely used by economists. The following common types of revealed preference methods are discussed in this section:

- Production or cost functions;
- Travel cost models;
- Hedonic models;
- Averting behavior models; and
- Cost of Illness (COI).[15]

7.3.1.1 Production and Cost Functions

Discrete changes in environmental circumstances generally cause both consumer and producer effects, and it is common practice to separate the welfare effects brought about by changes in environmental circumstances into consumer surplus and producer surplus.[16] Marginal changes can be evaluated by considering the production side of the market alone.

14 There are examples in which environmental goods have been traded in markets. The Clean Air Act Amendments of 1990, for example, initiated a market in sulfur dioxide (SO_2). However prices in such markets are determined by regulation-induced scarcity, and not by considerations of marginal utilities or marginal products.

15 Although not a revealed preference method (as it does not measure WTP) COI methods are discussed in this section since estimates are based on observable data.

16 See Appendix A for more detail.

Economic foundations of production and cost functions

Inputs to production contribute to welfare indirectly. The marginal contribution of a productive input is calculated by multiplying the marginal product of the input by the marginal utility obtained from the consumption good, in whose production the input is employed. The marginal utility of a consumption good is recorded in its price. While marginal products are rarely observed, the need to observe them is obviated when both inputs and outputs are sold in private markets because *prices* can be observed. Environmental goods and services are typically not traded in private markets, and therefore the values of environmental inputs must be estimated indirectly.

Production possibilities can be represented in three equivalent ways:

- As a production function relating output to inputs;
- As a cost function relating production expenses to output and to input prices; and
- As a profit function relating earnings to the prices of both output and inputs (see Varian 1992, for an explication of the relationships among these functions).

The value of a marginal change in some environmental condition can be represented as a marginal change in the value of production, as a marginal change in the cost of production, or as a marginal change in the profitability of production.[17] It should be noted, however, that problems of data availability and reliability often arise. Such problems may motivate the choice among these conceptually equivalent approaches, or in favor of another approach.

Note that derivation of values *on the margin* does not require any detailed understanding of consumer demand conditions. To evaluate marginal effects via the production function approach, the analyst needs to know the price of output and the marginal product of the environmental input. To derive the equivalent measure using a cost function approach, the analyst needs to know the derivative of the cost function with respect to the environmental input. In the profit function approach, the analyst needs to know the derivative of the profit function with respect to the environmental input.[18]

In the statements note the emphasis that *marginal* effects are being estimated. Estimating the net benefits of larger, non-marginal, changes represent a greater challenge to the analyst. In general this requires consideration of changes in both producer and consumer surplus. The latter necessitates application of techniques such as travel cost, hedonics, and stated preference, which are discussed elsewhere in this chapter.

Before moving on to those topics, note a fourth equivalent way to estimate environmental effects on production possibilities. Such effects are reflected in the profitability of enterprises engaged in production. That profitability also can be related to the return on fixed assets such as land. The value of a parcel of land is related to the stream of earnings that can be achieved by employing it in its "highest and best use." Its rental value is equal to the profits that can be earned from it over the period of rental (the terms "rent" and "profit" are often used synonymously in economics). The purchase price of the land parcel is equal to the expected discounted present value of the stream of earnings that can be realized from its use over time. Therefore, the production, cost, and profit function approaches described above are also equivalent to inferences drawn from the effects of environmental conditions on asset values. This fourth approach is known as "hedonic pricing," and will be discussed in detail in Section 7.3.1.3.

17 For a good review of statistical procedures used for estimating production, cost, and profit functions see Berndt (1991).

18 Derivation of marginal values often involves an application of the "envelope theorem" that states that effects from variables that are already optimized are negligible. In determining the effect of an improvement in a particular environmental input on welfare arising from the consumption of a particular product using the cost function approach, the analyst would determine how $\int_0^Q p(q)dq - C(Q, e)$ varies with e, the environmental variable. The integral is consumer surplus, i.e., the area under the demand curve, and the second term is the cost of producing quantity Q given environmental conditions, e. Differentiating with respect to e yields $[p(Q) - \partial C/\partial Q]\, dQ/de - \partial C/\partial e = -\partial C/\partial e$, where the last equality results because competitive firms set price equal to marginal cost, i.e., $p(Q) = \partial C/\partial Q$. This is the basis for the general proposition that marginal values can be estimated by looking solely at the production side of the market.

It is introduced now to show that production, cost, or profit function approaches are generally equivalent to hedonic approaches.

"Production" as a term is broad in meaning and application, especially with regard to hedonic pricing. While businesses produce goods and services in their industrial facilities, one might also say that developers "produce" housing services when they build residences. Therefore, hedonic pricing approaches can measure the value of the environment in "production," whether they are focusing on commercial or residential properties. Similarly, households may "produce" their health status by combining inputs such as air and water filtration systems and medical services along with whatever environmental circumstances they face. Or they "produce" recreational opportunities by combining "travel services" from private vehicles, their own time, recreational equipment purchases, and the attributes of their destination. Much of what is discussed elsewhere in this section is associated with this "production" analysis. This is not to say that estimation of production, cost, or profit functions is necessarily the best way to approach such problems, but rather, that all of these approaches are conceptually consistent.

General application of production and cost functions

Empirical applications of production and cost function approaches are diverse. Among other topics, the empirical literature has addressed the effects of air quality changes on agriculture and commercial timber industries. It also has 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.[19] Production, cost, or profit functions have found interesting applications to the estimation of some ecological benefits.[20] Probabilistic models of new product discovery from among diverse collections of natural organisms can also be regarded as a type of "production."[21] Finally, work in ecology points to "productive" relationships among natural systems that may yield insights to economists as well.[22]

Considerations in evaluating and understanding production and cost functions

The analyst should consider the following factors when estimating the values of environmental inputs into production:

Data requirements and implications. Estimating production, cost, or profit functions requires data on *all* inputs and/or their prices. Omitted variable bias is likely to arise absent such information, and may motivate the choice of one form over another. Econometricians have typically preferred to estimate cost or, better yet, profit functions. Data on prices are often more complete than are data on quantities and prices are typically uncorrelated to unobserved conditions of production, whereas input quantities are not.

The model for estimation. Standard practice involves the estimation of "flexible functional forms," i.e., functions that can be regarded as second-order approximations to any production technology. The translog and generalized Leontief specifications are examples. Estimation often will be more efficient if a system of equations is estimated (e.g., simultaneous estimation of a cost function and its associated factor demand equations), although data limitations may impose constraints.

Market imperfections. Most analyses assume perfectly competitive behavior on the part of producers and input suppliers, and assume an absence of other distortions. When these assumptions do not hold, the interpretation of welfare results becomes more problematic. While there is an extensive literature on the regulation of externalities under imperfect competition, originating with Buchanan (1969), analysts should exercise caution and restraint in attempting to correct for departures from competitive behavior.

19 Refer to Adams et al. (1986), Kopp and Krupnick (1987), Ellis and Fisher (1987), Taylor (1993), and U.S. EPA (1997a) for examples.

20 See, for example, Acharya and Barbier (2002) on groundwater recharge, and Pattanayak and Kramer (2001) on water supply.

21 For example, see Weitzman (1992), Simpson et al. (1996), and Rausser and Small (2000).

22 For example, see Tilman, Lehman, and Polasky 2005.

The issues can become quite complex and, as is the case with environmental externalities, there is typically no direct evidence of the magnitude of departures from perfectly competitive behavior. Moreover, in many circumstances it might reasonably be argued that departures from perfect competition are not of much practical concern (Oates and Strassman 1984). Perhaps a more pressing concern in many instances will be the wedge between private and social welfare consequences that arise with taxation. An increase in the value of production occasioned by environmental improvement typically will be split between private producers and the general public through tax collection. The issues here also can become quite complex (see Parry et al. 1997), with interactions among taxes leading to sometimes surprising implications. While it is difficult to give general advice, analysts may wish to alert policy makers to the possibility that the benefits of environmental improvements in production may accrue to different constituencies.

7.3.1.2 Travel Costs

Recreational values constitute a potentially large class of environmental use benefits. However, measuring these values is complicated by the fact that the full benefits of access to recreation activities are rarely reflected in admission prices. Travel cost models address this problem by inferring the value of changes in environmental quality through observing the trade-offs recreators make between environmental quality and travel costs. A common situation recreators may face is choosing between visiting a nearby lake with low water quality and a more distant lake with high water quality. The outcome of the decision of whether to incur the additional travel cost to visit the lake with higher water quality reveals information about the recreator's value for water quality. Travel cost models are often referred to as recreation demand models because they are most often used to value the availability or quality of recreational opportunities.

Economic foundation of travel cost models

Travel cost models of recreation demand focus on the choice of the number of trips to a given site or set of sites that a traveler makes for recreational purposes. Because there is no explicit market or price for recreation trips, travel cost models are frequently based on the assumption that the "price" of a recreational trip is equal to the cost of traveling to and from the site. These costs include both participants' monetary cost and opportunity cost of time. Monetary costs include all travel expenses. For example, when modeling day trips taken primarily in private automobiles, travel expenses would include roundtrip travel distance in miles multiplied by an estimate of the average cost per mile of operating a vehicle, plus any tolls, parking, and admission fees.

A participant's opportunity cost of time for a recreational day trip is the value of the participant's time spent traveling to and from the recreation site plus the time spent recreating.[23] A variety of approaches have been used in the literature to define the opportunity cost of time. Most commonly, researchers have used a fixed fraction ranging from one third to one whole of a person's hourly wage as an estimate of participants' hourly opportunity cost of time. In most cases, the fraction used depends on how freely individuals are assumed to be able to substitute labor and leisure. If a person can freely choose their work hours then their opportunity cost of time will be equal to their full wage rate. However, if a person cannot freely substitute labor for leisure (for example if they have a set 40 hour work week), then the opportunity cost of the time they have available for recreation is unobservable and may be less or more than the full wage rate. Many other factors can influence recreators' opportunity cost of time, including the utility received from traveling, non-wage income, and other non-work time constraints. A number of researchers have developed methods for estimating recreators' endogenous opportunity cost of time although no one method has yet been fully embraced in the literature. For examples, see McConnell and Strand (1981); Smith,

23 If the amount of time spent recreating or doing something else (not including the time spent traveling to and from the sites) is assumed to be the same across all alternatives then it will not be identifiable in estimation and therefore it is not necessary to include it in the estimation of the participant's opportunity cost of time. See Smith, Desvousges, and McGivney (1983); and McConnell (1992) for discussions of the implication of and the methods for allowing time onsite to vary across trip and alternatives.

Desvousges, and McGivney (1983); Bockstael et al. (1987b); McConnell (1992); and Feather and Shaw (1999). Hourly opportunity costs are multiplied by round trip travel time and time on-site to calculate a person's full opportunity cost of time. Total travel costs are the sum of monetary travel costs and full opportunity costs. Following the law of demand, as the cost of a trip increases the quantity of trips demanded generally falls, all else equal. This means that participants are more likely to visit a closer site than a site farther away.

While travel costs are the driving force of the model, they do not completely determine a participant's choice of sites to visit. Site characteristics, such as parking, restrooms, or boat ramps; participant characteristics, such as age, income, experience, and work status; and environmental quality also can affect demand for sites. The identification and specification of the appropriate site and participant characteristics are generally determined by a combination of data availability, statistical tests, and the researcher's best judgment. Ultimately, every recreation demand study strikes a compromise in defining sites and choice sets, balancing data needs and availability, costs, and time.[24]

General application by type of travel cost model

Travel cost models can logically be divided into two groups: single-site models and multiple-site models. Apart from the number of sites they address, the two types of models differ in several ways. The basic features of both model types are discussed below.

Single-site models. Single-site travel cost models examine recreators' choice of *how many trips to make to a specific site over a fixed period of time* (generally a season or year). It is expected that the number of trips taken will increase as the cost of visiting the site decreases and/or as the benefits realized from visiting increase. Site, participant, and environmental attributes, as well as the prices of substitute sites, act as demand curve shifters. For example, sites with good water quality are likely to be visited more often than sites with poor water quality, all else equal. Most current single-site travel cost models are estimated using count data models because the dependent variable (number of trips taken to a site) is a non-negative integer. See Haab and McConnell (2003) and Parsons (2003a) for detailed discussions and examples of recreation demand count data models.

Single-site models are most commonly used to estimate the value of a change in access to a site, particularly site closures (e.g., the closure of a lake due to unhealthy water quality). The lost access value due to a site closure is the difference between the participant's WTP for the option of visiting the site, which is given by the area between the site's estimated demand curve and the implicit "price" paid to visit it. Estimating the value of a change in the cost of a site visit, for example the addition or increase of an admission fee, is another common application of the model.

A weakness of the single-site model is its inability to deal with large numbers of substitute sites. If, as is often the case, a policy affects several recreation sites in a region, then traditional single-site models are required for each site. In cases with large numbers of sites, defining the appropriate substitute sites for each participant and estimating individual models for each site can impose overwhelming data collection and computational costs. Because of these difficulties, most researchers have opted to refrain from using single-site models when examining situations with large numbers of substitute sites.[25]

Multiple-site models. Multiple-site models examine a recreator's choice of *which site to visit from a set of available site (known as the choice set) on a given choice occasion* and in some cases can also examine *how many trips to make to each specific site*

24 For a comprehensive treatment of the theoretical and econometric properties of recreation demand models see Phaneuf and Smith (2005).

25 Researchers have developed methods to extend the single-site travel cost model to multiple sites. These variations usually involve estimating a system of demand equations. One example is the Kuhn-Tucker (KT) model discussed in the following multiple-site model section. See Bockstael, McConnell, and Strand (1991) and Shonkwiler (1999) for more discussion and other examples of extensions of the single-site model.

over a fixed period of time. Compared to the single-site model, the strength of multiple-site models lies in their ability to account for the availability and characteristics of substitute sites. By examining how recreators trade the differing levels of each site characteristic and travel costs when choosing among sites it is possible to place a per trip (or choice occasion) dollar value on site attributes or on site availability for single sites or multiple sites simultaneously.

The two most common multiple-site models are the random utility maximization (RUM) travel cost model and Kuhn-Tucker (KT) system of demand models. Both models may be described by a similar utility theoretic foundation, but they differ in important ways. In particular, the RUM model is a choice occasion model while the KT model is a model of seasonal demand.

Random utility maximization models. In a RUM model each alternative in the recreator's choice set is assumed to provide the recreator with a given level of utility, and on any given choice occasion the recreator is assumed to choose the alternative that provides the highest level of utility on that choice occasion.[26] The attributes of each of the available alternatives, such as the amenities available, environmental quality, and the travel costs, are assumed to affect the utility of choosing each alternative. Because people generally do not choose to recreate at every opportunity, a non-participation option is often included as a potential alternative.[27] From the researcher's perspective, the observable components of utility enter the recreator's assumed utility function. The unobservable portions of utility are captured by an error term whose assumed distribution gives rise to different model structures. Assuming that error terms have type 1 extreme values distribution leads to the closed form logit probability expression and allows for maximum likelihood estimation of utility function parameters. Using these estimated parameters it is then possible to estimate WTP for a given change in sites quality or availability.

However, because the RUM model examines recreation decisions on a choice occasion level, it is less suited for predicting the number of trips over a time period and measuring seasonal welfare changes. A number of approaches have been used to link the RUM model's estimates of values per choice occasion to estimates of seasonal participation rates. See Parsons, Jakus, and Tomasi (1999) for a detailed discussion of methods of incorporating seasonal participation estimates into the RUM framework.

The nested logit and mixed logit models are extensions of the basic logit. The nested logit model groups similar alternatives into nests where alternatives within a nest are more similar with each other than they are with alternatives outside of the nest. In very general terms, recreators are first assumed to choose a nest and then, conditional on the choice of nest, they then choose an alternative within that nest. Nesting similar alternatives allows for more realistic substitution patterns among sites than is possible with a basic logit. The mixed logit is a random parameter logit model that allows for even more flexible substitution patterns by estimating the variation in preferences (or correlation in errors) across the sample. If preferences do not vary across the sample then the mixed logit collapses to a basic logit.[28]

The Kuhn-Tucker (KT) model. The KT model is a seasonal demand model that estimates recreators' *choice of which sites to visit (like a multiple-site model) and how often to visit them over a season (like a single-site model).* The model is built on the theory that people maximize their seasonal utility subject to their budget constraint by purchasing

26 While the standard logit recreation demand model treats each choice occasion as an independent event, the model can also be generalized to account for repeated choices by an individual.

27 In a standard nested logit RUM model, recreators are commonly assumed to first decide whether or not to take a trip, and then conditional on taking a trip, to next choose which site to visit. By not including a non-participation option, the researcher in effect assumes that the recreator has already decided to take a trip, or in other words, that the utility of taking a trip is higher than the utility of doing something else for that choice occasion. Another way to think of it is that models lacking a participation decision only estimate the recreation values of the segment of the population that participates in recreation activities (i.e., recreators), while models that allow for non-participation incorporate the recreation values of the whole population (i.e., recreators and non-recreators combined). Because of this, recreation demand models without participation decisions tend to predict larger per person welfare changes than models allowing non-participation.

28 See Train (1998) and Train (2003) for detailed descriptions of the nested and mixed logit models.

the quantities of recreation and other goods that give them the greatest overall utility. Similar to the RUM model, the researcher begins by specifying the recreator's utility function. Taking the derivative of this utility function with respect to the number of trips taken, subject to a budget and non-negative trip constraint, yields the "Kuhn-Tucker" conditions. The KT conditions show that trips will be purchased up to the point that the marginal rate of substitution between trips and other spending is equal to the ratio of their prices. In cases where the price of a good exceeds its marginal value none will be purchased. Given assumptions on the form of the utility function and the distribution of the error term, probability expressions can be derived and parameter estimates may then be recovered. While recent applications have shown that the KT model is capable of accommodating a large number of substitute sites (von Haefen, Phaneuf, and Parsons 2004) the model is computationally intensive compared to traditional models. For a basic application of the KT model see Phaneuf and Siderelis (2003). For more advanced treatments of the models see Phaneuf, Kling, and Herriges (2000), and von Haefen and Phaneuf (2005).

Considerations in evaluating and understanding recreation demand studies

Definition of a site and the choice set. The definition of what constitutes a unique site has been shown to have a significant effect on estimation results. Ideally, one could estimate a recreation demand model in which sites are defined as specific points such as exact fishing location, campsites, etc. The more exact the site definition, the more exact the measure of travel costs and site attributes, and therefore WTP, that can be calculated. However, in situations with a large number of potential alternatives, the large data requirements may be cost and time prohibitive, estimation may be problematic, and aggregation may be required. The method of aggregation has been shown to have a significant effect on estimated values. The direction of the effect will depend on the situation being evaluated and the method of aggregation chosen (Parsons and Needleman 1992; Feather 1994; Kaoru, Smith, and Liu 1995; and Parsons, Plantinga, and Boyle 2000).

In addition to the definition of what constitutes a site, the number of sites included in a recreator's choice set can have a significant effect on estimated values. When defining choice sets, the most common practice in the literature has been to include all possible alternatives available to the recreator. In many cases availability has been defined by location with a given distance or travel time.[29] This strategy has been criticized on the grounds that people may not know about all possible sites, or even if they do know they exist they may not seriously consider them as alternatives. In response to this, a number of researchers have suggested methods that either restrict choice sets to include only those sites that the recreators seriously consider visiting (Peters et al. 1995, and Haab and Hicks 1997) or that weight seriously-considered alternatives more heavily than less-seriously-considered alternatives (Parsons, Massey, and Tomasi 2000).

Multiple-site or multipurpose trips. Recreation demand models assume that the particular recreation activity being studied is the sole purpose for a given trip. If a trip has more than one purpose, it almost certainly violates the travel cost model's central assumption that the "price" of a visit is equal to the travel cost. The common strategy for dealing with multipurpose trips is simply to exclude them from the data used in estimation.[30] See Mendelsohn et al. (1992) and Parsons (2003b) for further discussion.

Day trips versus multi-day trips. The recreation demand literature has focused almost exclusively on single-day trip recreation choices. One main reason researchers have focused mostly on day trips is that adding the option to stay longer than one day adds another choice variable in estimation,

29 Parsons and Hauber (1998) explore the implication of this strategy by expanding the choice set geographically and find that beyond some threshold the effect of additional sites is negligible.

30 Excluding any type or class of trip (like multiple-site or multipurpose) will produce an underestimate of the population's total use value of a site. The amount by which benefits will be underestimated will depend on the number and type of trips excluded.

thereby greatly increasing estimation difficulty. A second reason is that as trip length increases multipurpose trips become increasingly more likely, again casting doubt on the assumption that trip's travel costs represent the "price" of one single activity (see previous paragraph). A few researchers have estimated models that allow for varying trip length. The most common strategy has been to estimate a nested logit model in which each choice nest represents a different trip length option. See Kaoru (1995) and Shaw and Ozog (1999) for examples. The few multi-day trip models in the literature find that the *per-day* value of multi-day trips is generally less than the value of a single-day trip, which suggests that estimating the value of multi-day trips by multiplying a value estimated for single-day trips value by the number of days of will overestimate the multi-day trip value.

7.3.1.3 Hedonics

Hedonic pricing models use statistical methods to measure the contribution of a good's characteristics to its price. Cars differ in size, shape, power, passenger capacity, and other features. Houses differ in size, layout, and location. Even labor hours can be thought of as "goods" differing in attributes like risk levels, and supervisory nature, that should be reflected in wages. Hedonic pricing models use variations in property prices or wages and are commonly used to value the characteristics of properties or jobs. The models are based on the assumption that heterogeneous goods and services (e.g., houses or labor) consist of "bundles" of attributes (e.g., size, location, environmental quality, or risk) that are differentiated from each other by the quantity and quality of these attributes. Environmental conditions are among the many attributes that differ across neighborhoods and job locations.

Economic foundations of hedonic models

Hedonic pricing studies estimate economic benefits by weighing the advantages against the costs of different choices. A standard assumption underlying hedonic pricing models is that markets are in equilibrium, which means that no individual can improve her welfare by choosing a different home or job. For example, if an individual changed location she might move to a larger house, or one in the midst of a cleaner environment. However, to receive such amenities, the individual must pay for a more expensive house and incur transaction costs to move. The more the individual spends on her house, the less she has to spend on food, clothing, transportation, and all the other things she wants or needs. Thus, individuals are assumed to choose a better available option such that the benefits derived from it are exactly offset by the increased cost. So, if the difference in prices paid to live in a cleaner neighborhood is observable, then that price difference can be interpreted as the WTP for a better environment.

One key requirement in conducting a hedonic pricing study is that the available options differ in measurable ways. To see why, suppose that all locations in a city's housing market were polluted to the same degree, or all jobs in a particular labor market expose workers to the same risks. Homeowners and workers would, of course, be worse off due to their exposure to pollution and job risks, but their losses could not be measured unless a comparison could be made to purchasers of more expensive houses in less polluted neighborhoods, or wages in lower-paying but safer jobs. However, there is also a practical limit on the heterogeneity of the sample. Workers in different countries earn very different wages and face very different job risks, but this does not mean it is possible to value the difference in job risks by reference to international differences in wages. This is because: (1) there are many other factors that differ between widely separated markets; and (2) people simply are not mobile between very disparate sites. For these reasons it is important to exercise care in defining the market in which choices are made.[31]

Another aspect of the heterogeneity in locations required to make hedonic pricing studies work is that people must *be able to perceive* the differences among their options. If homeowners are unable to recognize differences in health outcomes, visibility, and other consequences of differences

31 Michaels and Smith (1990) offer guidance for defining the extent of the market.

in air quality at different locations, or if workers are unaware of differences in risks at different jobs, then a hedonic pricing study would not be suitable for estimating the values for those attributes.

Hedonic pricing studies can be used in different ways in environmental economics. Some are intended to provide direct evidence of the value of environmental improvements. Hedonic housing price studies are good examples. House prices are related to environmental conditions. The most frequent example is probably air quality (see Smith and Huang 1995 for a meta-analysis of many studies), although water quality (Leggett and Bockstael 2000), natural amenities (Thorsnes 2002), land contamination (Messer et al. 2006) and other examples have been studied. Other hedonic studies evaluate endpoints other than environmental conditions. A good example would be hedonic wage studies that are used in the computation of the VSL. (See Viscusi 2004 for a recent example.)

General application by type of hedonic pricing study

Hedonic wage studies, also known as wage-risk or compensating wage studies, are based on the premise that individuals make trade-offs between wages and occupational risks of death or injury. Most analysts assume that workers understand on-the-job risks, but others argue that workers generally underestimate them (Viscusi 1993). Some studies attempt to account for workers' perceived risks, but the results of these studies are not markedly different from those that do not (Gerking, de Haan, and Schulze 1988). Two of the most frequently used data sources for hedonic wage studies are the National Institute of Occupational Safety and Health (NIOSH) and Bureau of Labor Statistics (BLS) Survey on Working Conditions (SWC) data. The NIOSH data are state-level data of fatalities by occupation or industry, while the SWC data provide a finer resolution of occupation or industry fatalities, but do not vary by location. Black and Kneiser (2003), however, question the ability of hedonic wage studies using these data sources to measure job risks accurately due to severe measurement error. They find that the measurement error in the fatality rates reported from these sources is correlated with covariates commonly used in the wage equations, making the consistent estimation of the coefficient on risk in the standard hedonic wage equation a challenge. More recent hedonic wage studies have used the BLS Census of Fatal Occupational Injuries (CFOI) as the source for workplace risk information (Viscusi 2004; Viscusi and Aldy 2007b; Aldy and Viscusi 2008; Kniesner, Viscusi, and Ziliak 2006; Leeth and Ruser 2003; Viscusi 2003; and Scotton and Taylor 2009). These data are considered the most comprehensive data on workplace fatalities available (Viscusi 2004), compiling detailed information since 1992 from all states and the District of Columbia. Not only are the counts of fatal events reported by 3-digit occupation and 4-digit industry classifications, but the circumstances of the fatal events, as well as worker characteristics like age, gender and race, are also captured.[32] To ensure the veracity and completeness of the reported data, multiple sources, including death certificates, workers' compensation reports and federal and state administration reports are consulted and cross-referenced.

Although questions still persist about the applicability of hedonic wage study results to environmental benefits assessment, hedonic wage studies have been used most frequently in benefits assessments to estimate the value of fatal risk reductions.[33] When a benefits assessment requires a VSL estimate, hedonic wage estimates are a good source of information. Historically, EPA has used a VSL estimate primarily derived from hedonic wage studies. For more information on the Agency's VSL estimate, see Section 7.1.1 and Appendix C.[34] The VSL determined by a hedonic wage study, for example, typically relates WTA higher wages in exchange for the increased likelihood of accidental death during a person's working years. However,

32 More information on the CFOI data is available at: http://www.bls.gov/iif/oshfat1.htm.

33 For example, EPA's SAB has recognized the limitations of these estimates for use in estimating the benefits of reduced cancer incidence from environmental exposure. Despite these limitations, however, the SAB concluded that these estimates were the best available at the time (U.S. EPA 2000d).

34 As part of the revision of this document, EPA is revisiting the VSL estimate used in policy analysis; further guidance will be forthcoming.

analysts should take care when applying results from one hedonic study to a new policy case, for example, if there are differences in the age groups facing mortality risks from longer-term conditions.

Hedonic property value studies measure the different contributions of various characteristics to the value of property. These studies are typically conducted using residential housing data, but they have also been applied to commercial and industrial property, agricultural land, and vacant land.[35] Bartik (1988) and Palmquist (1988, 1991) provide detailed discussions of benefits assessment using hedonic methods. Property value studies require large amounts of disaggregated data. To avoid aggregation problems, market transaction prices on individual parcels or housing units are preferred to aggregate data such as census tract information on average housing units. Problems can arise from errors in measuring prices (aggregated data) and errors in measuring product characteristics (particularly those related to the neighborhood and the environment). There are numerous statistical 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, endogeneity, and spatial correlation. Refer to Palmquist (1991) for a thorough treatment of the main econometric issues. Recently, advances have been made in modeling spatial correlation in hedonic models (see Text Box 7.4 on spatial correlation for more information).

Other hedonic studies. Applicability of the hedonic pricing method is not limited to the property and labor markets. For example, hedonic pricing methods can be combined with travel cost methods to examine the implicit price of recreation site characteristics (Brown and Mendelsohn 1984). Results from other studies can be used to infer the value of reductions in mortality, cancer, or injury risks. For example, Dreyfus and Viscusi (1995) use a hedonic analysis to determine the trade-offs between automobile price and safety features to infer the VSL.

Considerations in evaluating and understanding hedonic pricing studies

Unobservable factors. A concern common to hedonic pricing studies is that it is impossible to observe all factors that go into a decision. People will choose among different jobs or houses not only because they can trade off differences in amenities and risks against differences in prices or wages, but also because they have different preferences for risks. Idiosyncratic personal tastes that cannot be observed may be responsible for a substantial portion of differences in observed choices. For example, mountain climbers have been known to pay tens of thousands of dollars to undertake expeditions that substantially increase their likelihood of early death.

Source of risks. Similarly, analysts need to be careful in distinguishing the source of the risks used to estimate risk premia. Consider an individual who both works a dangerous job and lives in unhealthy circumstances. Such a person may be at greater risk of premature death than someone who works a different job or lives elsewhere. Analysts risk underestimating the wage premium demanded on the job if they fail to distinguish between causes of death — for example between on-the-job accidents and environmentally induced conditions acquired at home — when relating the wage premium paid on dangerous jobs to the statistics on premature mortality. Conversely, if the same job poses multiple risks — say the risk of both accidental death and serious, but nonfatal injury were higher on a particular job — the wage premium the job offers would overstate WTP for reductions in mortality risks if the injury risks were not properly controlled for in the analysis. See Eeckhoudt and Hammitt (2001), and Evans and Smith (2006) for more discussion of competing versus specific risks.

Marginal changes. As with many results in economics, hedonic pricing models are best suited to the valuation of small, or marginal, changes in attributes. Under such circumstances, the slope

35 See Xu, Mittlehammer, and Barkley (1993), and Palmquist and Danielson (1989) for hedonic values of agricultural land; Ihlanfeldt and Taylor (2004) for commercial property; Dale, Murdoch, Thayer, and Waddell (1999), and McCluskey and Rausser (2003) for residential property; and Clapp (1990), and Thorsnes (2002) for vacant land.

Text Box 7.4 - Spatial Correlation

Real property, such as buildings and land, and their associated characteristics are spatially distributed over the landscape. As such, the characteristics of some of the properties may be spatially correlated. If some of these characteristics are unobserved or for any other reason are not incorporated into the econometric model, there may be dependence across the error terms of the model. Spatial econometrics is a subfield of econometrics that has gained more attention as the capability for assessing such locational relationships within hedonic property data has improved. Such improvements are primarily due to the increasing use of geographic information systems (GIS) technology and geographically referenced data sets.

The nature of the correlation in the data can manifest itself so that there is either spatial heterogeneity across observations, or more importantly, so that the characteristic values (e.g., price of homes) are correlated with those of nearby observations. Standard econometric techniques can readily deal with the former, but are not well equipped to handle the latter case. The econometric techniques allow for testing for the presence of spatial correlation, and specifically modeling and correcting the correlation between observations and correcting for the biasing effect it can have on parameter estimates. In practice, a relationship is defined between every variable at a given location and the same variable at other, usually nearby, locations in the data set. In most cases this relationship is based on common boundaries or is some specified function based on the distances between observations. This relationship between observations is then accounted for in the econometric model in order to correct the error terms and obtain unbiased model estimates. For more details on the fundamentals of spatial statistics see Anselin (1988).

of the hedonic price function can be interpreted as WTP for a small change in the attribute. Public policy, however, is sometimes geared to larger, discrete changes in attributes. When this is the case, calculation of benefits can become significantly more complicated. Hedonic price functions typically reflect equilibria between consumer demands and producer supplies for fixed levels of the attributes being evaluated. The demand and supply functions are tangent to the hedonic price function only in the immediate neighborhood of an equilibrium point. Palmquist (1991) describes conditions under which exact welfare measures can be calculated for discrete changes. See Freeman (2003) and Ekeland, Heckman, and Nesheim (2004) for recent treatments.

7.3.1.4 Averting Behaviors

The averting behavior method infers values for environmental quality from observations of actions people take to avoid or mitigate the increased health risks or other undesirable consequences of reductions in ambient environmental quality conditions. Examples of such defensive actions can include the purchase and use of air filters, boiling water prior to drinking it, and the purchase of preventative medical care or treatment. By analyzing the expenditures associated with these averting behaviors economists can attempt to estimate the value individuals place on small changes in risk (Shogren and Crocker 1991, and Quiggin 1992).

Economic foundations of averting behavior methods

Averting behavior methods can be best understood from the perspective of a household production framework. Households can be thought of as producing health outcomes by combining an exogenous level of environmental quality with inputs such as purchases of goods that involve protection against health and safety risks (Freeman 2003). To the extent that averting behaviors are available, the model assumes that a person will continue to take protective action as long as the expected 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 marginal cost just equals her marginal WTP for these reductions. Thus, the value of a small change in health risks can be estimated from two primary pieces of information:

- The cost of the averting behavior or good; and
- Its effectiveness, as perceived by the individual, in offsetting the loss in environmental quality.

Blomquist (2004) provides a detailed description of the basic household production model of averting behavior. 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: changes in averting expenditures, changes in mitigating expenditures, lost time, 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 calculating changes in these costs 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 (Shogren and Crocker 1991, and Quiggin 1992).

General application of averting behavior method

Although the first applications of the method were directed toward values for benefits of reduced soiling of materials from environmental quality changes (Harford 1984), recent research has primarily focused on health risk changes. Conceptually, the averting behavior method can provide WTP estimates for a variety of other environmental benefits such as damages to ecological systems and materials.

Some averting behavior studies focus on behaviors that prevent or mitigate the impact of particular symptoms (e.g., shortness of breath or 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 could potentially be more useful, however, for assessing the benefits of a regulation expected to reduce the probability of similar contamination episodes.

Considerations in evaluating and understanding averting behavior studies

Perceived versus actual risks. Analysts should remember that consumers base their actions on perceived benefits from defensive behaviors. Many averting behavior studies explicitly acknowledge that their estimates rest on consistency between the consumer's perception of risk reduction and actual risk reduction. While there is some evidence that consumers are rational with regard to risk — for example, consumer expenditures to reduce risk vary positively with risk increases — there is also evidence that there are predictable differences between consumers' perceptions and actual risks. Thus, averting behavior studies can produce biased WTP estimates for a given change in objective risk. Surveys may be necessary 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. For example, if surveys reveal that perceived risks are lower than expert risk estimates, then WTP can be estimated with the lower, perceived risk (Blomquist 2004).

Data requirements and implications. Data needed for averting behavior studies 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, like age, health status, or chronic conditions.

Separability of joint benefits. Analysts should exercise caution in interpreting the results of studies that focus on goods in which there may be significant joint benefits (or costs). Many defensive behaviors not only avert or mitigate 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 taste better. Conversely, it also is 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 these "joint" benefits and costs associated with averting behaviors will result in biased estimates of WTP.

Modeling assumptions. Restrictive assumptions are sometimes needed to make averting behavior models tractable. Analysts drawing upon averting behavior studies will need to review and assess the implications of these assumptions for the valuation estimates.

7.3.1.5 Cost of Illness

A frequently encountered alternative to WTP estimates is the avoided cost of illness (COI). The COI method estimates the financial burden of an illness based on the combined value of direct and indirect costs associated with the illness. Direct costs represent the expenditures associated with diagnosis, treatment, rehabilitation, and accommodation. Indirect costs represent the value of illness-related lost income, productivity, and leisure time. COI is better suited as a WTP proxy when the missing components (e.g., pain and suffering) are relatively small as they usually are in cases of in minor, acute illnesses. However, there are usually better medical treatment and lost productivity estimates for more severe illnesses.

The COI method is straightforward to implement and explain to policy makers, and has a number of other advantages. The method has been used for many years and is well developed. Collecting data to implement it often is less expensive than for other methods, improving the feasibility of developing original COI estimates in support of a specific policy.

Economic foundations of COI studies

Two conditions must be met for the COI method to approximate a market value of reduced health risk. First, the direct costs of morbidity must reflect the economic value of goods and services used to treat illness. Second, a person's earnings must reflect the economic value of lost work time, productivity, and leisure time. Because of distortions in medical and labor markets, these assumptions do not routinely hold. Further, COI estimates are not necessarily equal to WTP. The method generally does not attempt to measure the loss in utility due to pain and suffering, and does not account for the costs of any averting behaviors that individuals have taken to avoid an illness. When estimates of WTP are not available, the potential bias inherent in relying on COI estimates should be acknowledged and discussed. A second shortcoming of the COI method is that by focusing on ex post costs, it does not capture the risk attitudes associated with ex ante measures of reduced health risk.

Although COI estimates do not adequately capture several components of WTP, COI does not necessarily serve as a lower bound estimate of WTP. This is because, for some illnesses, the cost of behaviors that allow one to avoid an illness might be far lower than the cost of the illness itself. Depending on the design of the research question, WTP could reflect the lower avoidance costs while COI would reflect the higher costs of treating the illness once it has been contracted. In addition, COI estimates capture medical expenses passed on to third parties such as health insurance companies and hospitals, whereas WTP estimates generally do not. Finally, COI estimates capture the value of lost productivity (see Text Box 7.4 above), whereas these costs may be overlooked in WTP estimates — especially when derived from consumers or employees covered by sick leave.

Available comparisons of COI and total WTP estimates suggest that the difference can be large (Rowe et al. 1995). This difference varies greatly across health effects and across individuals.

General application by type of COI study

Prevalence-based estimates. Prevalence-based COI estimates are derived from the costs faced by all individuals who have a sickness in a specified time period. For example, an estimate of the total number of individuals who currently have asthma, as diagnosed by a physician, reflects the current prevalence of physician-diagnosed asthma. Prevalence-based COI estimates for asthma include all direct and indirect costs associated with asthma within a given time period, such as a year. Prevalence-based COI estimates are a measure of the full financial burden of a disease, but generally will be lower bound estimates of the total WTP for avoiding the disease altogether. They are useful for evaluating the financial burden of policies aimed at improving the effectiveness of treatment or at reducing the morbidity and mortality associated with a disease.

Incidence-based estimates. By contrast, incidence-based COI estimates reflect expected costs for *new* individuals who develop a disease in a given time period. For example, the number of individuals who receive a *new* diagnosis of asthma from a physician in a year reflects the annual incidence of physician-diagnosed asthma. Incidence-based COI estimates reflect the expected value of direct medical expenditures and lost income and productivity associated with a disease from the time of diagnosis until recovery or death. Because these expenses can occur over an extended time period, incidence-based estimates are usually discounted to the year the illness is diagnosed and expressed in present value terms. Incidence-based COI estimates are useful for evaluating the financial burden of policies that are aimed at reducing the incidence of new cases of disease.

Most existing COI 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 estimates the total medical costs of the disease by using a database of actual costs incurred for patients with the illness. The "expert elicitation" 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.

COI estimates for many illnesses are readily available from existing studies and span a wide range of health effects. EPA's *Cost of Illness Handbook* (U.S. EPA 2007c) provides estimates for many cancers, developmental illnesses and disabilities, and other illnesses.

Considerations in evaluating and understanding COI studies

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 COI 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. 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 she would otherwise qualify for as a healthy person, or drop out of the labor force altogether. A second issue 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 her family is not

Text Box 7.5 - Value of Time

Estimating the cost of an illness by examining only medical costs clearly understates the true costs experienced by an individual with ill health. Not only does the individual incur medical expenditures, they also miss production and consumption opportunities. In particular they miss opportunities to work for wages, produce household goods and services (e.g., laundry, home-cooked meals), and enjoy leisure activities. These latter two categories are jointly referred to as non-work time. The value of these lost opportunities has typically been estimated by examining the value of time.

EPA has developed an approach for valuing time losses based on the opportunity cost of time. For paid work, the approach is relatively straightforward. It rests on the assumption that total compensation (wages and employment benefits) is equal to the employers' valuation of the worker's output. Therefore, if a worker is absent due to illness, society loses the value of the foregone output, which can be estimated by examining the worker's wages and employment benefit values. To value time spent on non-market work and leisure activities, the assumption is made that an individual will engage in such unpaid activities only if, at the margin, the value of these activities is greater than the wages that could be earned in paid employment. Hence after-tax wages provide a lower bound estimate of the value of non-work time.

The loss of work time and leisure activities due to illness need not be complete. When an illness reduces but does not eliminate productivity at work or enjoyment of leisure time, estimates of the value of the diminishments in these opportunities are legitimate components of the cost of the illness.

Valuing time lost due to illness experienced by children and other subpopulations that do not earn wages is more difficult. Examples of such subpopulations include the elderly, unemployed, or individuals who are out of the work force. Analysts could surmise the post-tax wage if such individuals were employed; however, the situation involves less certainty. For example, the time loss of children who suffer illness is sometimes estimated by considering the effect of the illness, if any, on future earnings. For this case, however, *Circular A-4* (OMB 2003) currently suggests that, in the absence of better data, monetary values for children should be at least be as large as the values for adults (for the same risk probabilities and health outcomes).

Accounting for time losses in COI estimates comes closer to a full accounting of the losses borne by individuals suffering illness than simply assessing medical costs. However, a third cost category remains neglected — the value of pain and suffering. When an individual is sick, she not only misses opportunities to produce or relax, she also would be willing to pay some amount to avoid the pain or discomfort of the illness. In most economic models, these costs are represented as declines in utility and as such are inherently difficult to estimate. To date, there are no good estimates, or methods for obtaining good estimates, of the value of avoiding pain.

included in most COI studies. (See Text Box 7.5 for a discussion of the value of time.)

7.3.2 Stated Preference

The distinguishing feature of stated preference methods compared to revealed preference methods is that stated preference methods rely on data drawn from people's responses to hypothetical questions while revealed preference methods rely on observations of actual choices. Stated preference methods use surveys that ask respondents to consider one or a series of hypothetical scenarios that describe a potential change in a non-market good. The advantages of stated preference methods include their ability to estimate non-use values and to incorporate hypothetical scenarios that closely correspond to a policy case. The main disadvantage of stated preference methods is that they may be subject to systematic biases that are difficult to test for and correct.

National Oceanic and Atmospheric Administration's (NOAA) *The Report of the NOAA Panel on Contingent Valuation* is often cited as a primary source of information on stated preference techniques. Often referred to as the "NOAA Blue Ribbon Panel," this panel, comprised of five distinguished economists including two Nobel Laureates, deliberated on the usefulness of stated preference studies for policy analysis (Arrow et al. 1993). While their findings generally mirror the recommendations offered below, since the release of their report a number of changes in the survey administration "landscape" have occurred including the advent of internet surveys, the decline in representativeness of telephone surveys, and the growth in popularity of stated choice experiments.

7.3.2.1 Economic Foundation of Stated Preference Methods

The responses elicited from stated preference surveys, if truthful, are either direct expressions of WTP or can be used to estimate WTP for the good in question. However, the "if truthful" caveat is paramount. While many environmental economists believe that respondents can provide truthful answers to hypothetical questions and therefore view stated preference methods as useful and reliable if conducted properly, a non-trivial fraction of economists are more skeptical of the results elicited from stated preference surveys. Due to this skepticism, it is important to employ validity and reliability tests of stated preference results when applying them to policy decisions.

If the analyst decides to conduct a stated preference survey or use stated preference results in a benefit transfer exercise, then a number of survey design issues should be considered. Stated preference researchers have attempted to develop methods to make individuals' choices in stated preference studies as consistent as possible with market transactions. Reasonable consistency with the framework of market transactions is a guiding criterion for ensuring the validity of stated preference value estimates. Three components of market transactions need to be constructed in stated preference surveys: the commodity, the payment, and the scenario (Fischoff and Furby 1988).

Stated preference studies need to carefully define the commodity to be valued, including characteristics of the commodity such as the timing of provision, certainty of provision, and availability of substitutes and complements. The definition of the commodity generally involves identifying and characterizing attributes of the commodity that are relevant to respondents. Commodity definition also includes defining or explaining baseline or current conditions, property rights in the baseline, and the policy scenarios, as well as the source of the change in the environmental commodity.[36]

Respondents also must be informed about the transaction context, including the method, timing, and duration of payment. The transaction must not be coerced and the individual should be aware of her budget constraint. The payment vehicle should be described as a credible and binding commitment should the respondent decide to purchase the good. The timing and duration of a payment involves individuals implicitly discounting payments and calculating expected utility for future events. The transaction context and the commodity definition should describe and account for these temporal issues.

The hypothetical scenario(s) should be described so as to minimize potential strategic behavior such as "free riding" or "overpledging." In the case of free riding, respondents will underbid their true WTP for a good if they feel they will actually be made to pay for it but believe the good will be provided nevertheless. In the case of overpledging, respondents pledge amounts greater than their true WTP with the expectation that they will not be made to pay for the good, but believing that their response could influence whether or not the good will be provided. Incentive-compatible choice scenarios and attribute-based response formats have been shown to mitigate strategic responses. Both are discussed below.

It is recognized in both the experimental economics literature and the survey methodology

36 Depending on the scenario, the description of the commodity may produce strong reactions in respondents and could introduce bias. In these cases, the detail with which the commodity of the change is specified needs to be balanced against the ultimate goals of the survey. Regardless, the commodity needs to be specified with enough detail to make the scenario credible.

literature that different survey formats can elicit different responses. Changing the wording or order of questions also can influence the responses. Therefore, the researcher should provide a justification for her choice of survey format and include a discussion of the ramifications of that choice.

7.3.2.2 General Application by Type of Stated Preference Study

Two main types of stated preference survey format are currently used: direct WTP questions and stated choice questions. Stated choice questions can be either dichotomous choice questions or multi-attribute choice questions. Following a general discussion of survey format, each of the stated preference survey formats is described in detail below.

Goals that should guide selection of the survey format include the minimization of survey costs, of non-responsiveness, of unexplained variance, and of complications associated with WTP estimation. For example, open-ended questions require smaller sample sizes and are simpler to analyze than other methods of asking the valuation question. These advantages could lead to significant cost reductions. However, these advantages may be mitigated by higher non-response rates and large unexplained variance in the responses. Moreover, there remains a great deal of uncertainty over the effect of the choice mechanism (i.e., open-ended, dichotomous choice, etc.) on the ability and willingness of respondents to provide accurate and well-considered responses.

Because survey formats are still evolving and many different approaches have been used in the literature, no definitive recommendations are offered here regarding selection of the survey format. Rather, the following sections describe some of the most commonly used formats and discuss some of their known and suspected strengths and weaknesses. Researchers should select a format that suits their topic, and should strive to use focus groups, pretests, and statistical validity tests to address known and suspected weaknesses in the selected approach.

Direct/open-ended WTP questions

Direct/open-ended WTP questions ask respondents to indicate their maximum WTP for the specific quantity or quality changes of a good or service that has been described to them. An important advantage of open-ended stated preference questions is that the answers provide direct, individual-specific estimates of WTP. Although this is the measure that economists want to estimate, early stated preference studies found that some respondents had difficulty answering open-ended WTP questions and non-response rates to such questions were high. Such problems are more common when the respondent is not familiar with the good or with the idea of exchanging a direct dollar payment for the good. An example of a stated preference study using open-ended questions is Brown et al. (1996).

Various modifications of the direct/open-ended WTP question format have been developed in an effort to help respondents arrive at their maximum WTP estimate. In *iterative bidding* respondents are asked if they would pay some initial amount, and then the amount is changed up or down depending on whether the respondent says "yes" or "no" to the first amount. This continues until a maximum WTP is determined for that respondent. Iterative bidding has been shown to suffer from "starting point bias," wherein respondents' maximum WTP estimates are systematically related to the dollar starting point in the iterative bidding process (Rowe and Chestnut 1983, Boyle et al. 1988, and Whitehead 2002). A *payment card* is a list of dollar amounts from which respondents can choose, allowing respondents an opportunity to look over a range of dollar amounts while they consider their maximum WTP. Mitchell and Carson (1989) and Rowe et al. (1996) discuss concerns that the range and intervals of the dollar amounts used in payment card methods may influence respondents' WTP answers.

Stated choice questions

While direct/open-ended WTP questions are efficient in principle, researchers have generally turned to other stated preference techniques in recent years. This is largely due to the difficulties respondents face in answering direct WTP questions and the lack of easily implemented procedures to mitigate these difficulties. Researchers also have noted that direct WTP questions with various forms of follow-up bidding may not be "incentive compatible." That is, the respondents' best strategy in answering these questions is not necessarily to be truthful (Freeman 2003).

In contrast to direct/open-ended WTP questions, stated choice questions ask respondents to choose a single preferred option or to rank options from two or more choices. When analyzing the data the dependent variable will be continuous for open-ended WTP formats and discrete for stated choice formats.[37] In principle, stated choice questions can be distinguished along three dimensions:

- *The number of alternatives each respondent can choose from in each choice scenario* — surveys may offer only two alternatives (e.g., yes/no, or "live in area A or area B); two alternatives with an additional option to choose "don't know" or "don't care;" or multiple alternatives (e.g., "choose option A, B, or C").
- *The number of attributes varied across alternatives in each choice question (other than price)* — alternatives may be distinguished by variation in only a single attribute (e.g., mortality risk) or by variation in multiple attributes (e.g., price, water quality, air quality, etc.).
- *The number of choice scenarios an individual is asked to evaluate through the survey.*

Any particular stated choice survey design could combine these dimensions in any given way. For example, a survey may offer two options to choose from in each choice scenario, vary several attributes across the two options, and present each respondent with multiple choice scenarios through the course of the survey. Using the taxonomy presented in these *Guidelines*, a complete (though cumbersome) description of this format would be a dichotomous choice/multi-attribute/multi-scenario survey. The statistical strategy for estimating WTP is largely determined by the survey format adopted, as described below.

The earliest stated choice questions were simple yes/no questions. These were often called *referendum* questions because they were often posed as, "Would you vote for . . ., if the cost to you were $X?" However, these questions are not always posed as a vote decision and are now commonly called *dichotomous choice* questions.

In recent years, stated preference researchers have been adapting a choice question approach used in the marketing literature called *conjoint analysis*. These are more complex choice questions in which the respondent is asked repeatedly to pick her preferred option from a list of two or more options. Each option represents a package of product attributes. By incorporating a dollar price or cost in each option, stated preference researchers are able to extract WTP estimates for incremental changes in the attributes of the good, based on the preferences expressed by the respondents. Holmes and Adamowicz (2003) refer to this as *attribute-based stated choice*.

Dichotomous choice WTP questions. Dichotomous choice questions present respondents with a specified environmental change costing a specific dollar amount and then ask whether or not they would be willing to pay that amount for the change. The primary advantage of dichotomous choice WTP questions is that they are easier to answer than direct WTP questions, because the respondent is not required to determine her exact WTP, only whether it is above or below the stated amount. Sample mean and median WTP values can be derived from analysis of the frequencies of the yes/no responses to each dollar amount. Bishop and Heberlein (1979), Hanemann (1984), and Cameron and James (1987) describe the necessary statistical procedures for analyzing dichotomous choice responses using logit or probit models. Dichotomous choice responses will reveal an interval containing WTP and in the case of a 'yes' response this interval will be unbounded from above. As a result, significantly larger sample sizes are needed for

37 Some researchers use the term "contingent valuation" to refer to direct WTP and dichotomous choice/referendum formats and "stated preference" to refer to other stated choice formats. In these *Guidelines* the term "stated preference" is used to refer to all valuation studies based on hypothetical choices (including open-ended WTP and stated choice formats), as distinguished from "revealed preference."

dichotomous choice questions to obtain the same degree of statistical efficiency in the sample means as direct/open-ended responses that reveal point-values for WTP (Cameron and James 1987).

To increase the estimation efficiency of dichotomous choice questions, recent applications have commonly used what is called a double-bounded approach. In double-bounded questions the respondent is asked whether she would be willing to pay a second amount, higher if she said yes to the first amount, and lower if she said no to the first amount.[38] Sometimes multiple follow-up questions are used to try to narrow the interval around WTP even further. These begin to resemble iterative bidding style questions if many follow-up questions are asked. Similar to starting point bias in iterative bidding questions, the analyses of double-bounded dichotomous choice question results suggest that the second responses may not be independent of the first responses (Cameron and Quiggin 1994, 1998; and Kanninen 1995).

Multi-attribute choice questions. In multi-attribute choice questions, respondents are presented with alternative choices that are characterized by different combinations of goods and services attributes and prices. Multi-attribute choice questions ask respondents to choose the most preferred alternative (a partial ranking) from multiple alternative goods (i.e., a choice set), in which the alternatives within a choice set are differentiated by their attributes including price (Johnson et al. 1995 and Roe et al. 1996). The analysis takes advantage of the differences in the attribute levels across the choice options to determine how respondents value marginal changes in each of the attributes. To measure WTP, a price (often a tax or a measure of travel costs), is included in multi-attribute choice questions as one of the attributes of each alternative. This price and the mechanism by which the price would be paid need to be explained clearly and plausibly, as with any payment mechanism in a stated preference study. Boyle and Özdemir (2009) examine the impact of question design choices, such as the ordering of attributes and the number of alternatives in a single question, on the mean WTP estimate.

38 Alberini (1995) illustrated an analysis approach for deriving WTP estimates from such responses and demonstrates the increased efficiency of double-bounded questions. The same study showed that the most efficient range of dollar amounts in a dichotomous choice study design was one that covered the mid-range of the distribution and did not extend very far into the tails at either end.

There are many desirable aspects of multi-attribute choice questions, including the nature of the choice being made. To choose the most preferred alternative from some set of alternatives is a common decision experience in posted-price markets, especially when one of the attributes of the alternatives is a price. One can argue that such a decision encourages respondents to concentrate on the trade-offs between attributes rather than taking a position for or against an initiative or policy. This type of repeated decision process may also diffuse the strong emotions often associated with environmental goods, thereby reducing the likelihood of yea-saying or of rejecting the premise of having to pay for an environmental improvement.[39] Presenting repeated choices also gives the respondent some practice with the question format, which may improve the overall accuracy of her responses, and gives her repeated opportunities to express support for a program without always selecting the highest price option.

Some applications of multi-attribute survey formats include Opaluch et al. (1993), Adamowicz et al. (1994), Viscusi et al. (1991), Adamowicz et al. (1997), Adamowicz et al. (1998a), Layton and Brown (2000), Johnson and Desvousges (1997), Boyle et al. (2001), and Morey et al. (2002). Studies that investigate the effects of multi-attribute choice question design parameters include Johnson et al. (2000) and Adamowicz et al. (1997).

7.3.2.3 Considerations in Evaluating Stated Preference Results

Survey mode. The mode used to administer a survey is an important component of survey research design because it is the mechanism by

39 Yea-saying refers to the behavior of respondents when they overstate their true WTP in order to show support for a situation described in survey questions.

which information is conveyed to respondents, and likewise determines the way in which individuals can provide responses for analysis. Until recently there were three primary survey modes: telephone, in-person, and mail. Telephone surveys are primarily conducted with a trained interviewer using random digit dialing (RDD) to contact households. In-person surveys are conducted in a variety of ways, including door-to-door, intercepts at public locations, and via telephone recruiting to a central facility. Mail surveys are conducted by providing written survey materials for respondents to self-administer. As technology and society has changed, so has the preference for one mode over the other. With the influx of market research and telemarketing, the telephone has become a less convenient way to administer surveys. Many people refuse to answer the phone, or to answer questions over the phone. The same can be said of mail surveys. People are quick to ignore unsolicited mail. In recent years the Internet has emerged as a possible mode for conducting surveys. Internet access and email accounts are more prevalent and computer literacy is high in the United States and other developed countries. As with all of the survey modes mentioned, there are inherent biases. These biases are generally classified as social desirability bias, sample frame bias, avidity bias, and non-response bias. See Maguire (2009), Loomis and King (1994), Mannesto and Loomis (1991), Lindberg et al. (1997), and Ethier et al. (2000) for a discussion of different biases in survey mode.

Framing issues. An important issue regarding survey formats is whether information provided in the questions influences the respondents' answers in one way or another. For example, Cameron and Huppert (1991) and Cooper and Loomis (1992) find that mean WTP estimates based on dichotomous choice questions may be sensitive to the ranges and intervals of dollar amounts included in the WTP questions. Kanninen and Kriström (1993) show that the sensitivity of mean WTP to bid values can be caused by model misspecification, failure to include bid values that cover the middle of the distribution, or inclusion of bids from the extreme tails of the distribution.

Selection of payment vehicle. The payment vehicle in a stated preference study refers to the method by which individuals or households would pay for the good described in a particular survey instrument. Examples include increases in electricity prices, changes in cost of living, a one-time tax, or a donation to a special fund. It is imperative that the payment vehicle is incentive compatible and does not introduce any strategic or other bias. Incentive compatibility means that the individual is motivated to respond truthfully and does not use their responses to try to influence a particular outcome (e.g., state a WTP value that is higher than their true WTP to try to make sure a particular outcome succeeds).

Strategic behavior. Adamowicz et al. (1998a) also suggests that respondents may be less likely to behave strategically when responding to multi-attribute choice experiments. Repeatedly choosing from several options gives the respondent some practice with the question format that may improve the overall accuracy of her responses, and gives her repeated opportunities to express support for a program without always selecting the highest price option.

Yea-saying. As mentioned above, yea-saying refers to the behavior of respondents when they overstate their true WTP in order to show support for situation described in survey questions. For example, Kanninen (1995) finds some evidence of yea-saying in dichotomous choice responses through testing in follow-up questions. The extent of this potential problem is not well established, but it may provide an explanation for the fact that mean WTP values based on dichotomous choice responses tend to be equal to or higher than values from direct WTP questions for the same good (Cummings et al. 1986, Boyle et al. 1993, Brown et al. 1996, Ready et al. 1996, and Balistreri et al. 2001). It has not been determined whether yea-saying can be reduced by double-bounded dichotomous choice because in this case the respondent has more than one opportunity to say yes.

Treatment of "don't know" or neutral responses. Based on recommendations from the NOAA Blue Ribbon panel (Arrow et al. 1993), many surveys now include "don't know" or "no preference" options for respondents to choose from. There have been questions about how such responses should enter the empirical analysis. Examining referendum-style dichotomous choice questions, Carson et al. (1998) found that when those who chose not to vote were coded as "no" responses, the mean WTP values were the same as when the "would not vote" option was not offered. Offering the "would not vote" option did not change the percentage of respondents saying "yes". Thus, they recommend that if a "would not vote" option is included, it should be coded as a "no" vote, a practice that has become widespread. Stated preference studies should always be explicit about how they treat "don't know," "would not vote," or other neutral responses.

Reliability, in general terms, means consistency or repeatability. If a method is used numerous times to measure the same commodity, then the method is considered more reliable the lower the variability in the results.

- **Test-retest approach.** Possibly the most widely applied approach for assessing reliability in stated preference studies has been the test-retest approach. Test-retest assesses the variability of a measure between different time periods. Loomis (1989), Teisl et al. (1995), McConnell et al. (1998), and Hoban and Whitehead (1999) all provide examples of the test-retest method for reliability.
- **Meta-analysis of stated preference survey results** for the same good also may provide evidence of reliability. Meta-analysis evaluates multiple studies as though each was constructed to measure the same phenomenon. Meta-analysis attempts to sort out the effects of differences in the valuation approach used in different surveys, along with other factors influencing the elicited value. For example Boyle et al. (1994) use meta-analysis to evaluate eight studies conducted to measure values for groundwater protection. (Also see Section 7.4.)

Validity tests seek to assess whether WTP estimates from stated preference methods behave as a theoretically correct WTP should. Three types of validity discussed below are: content validity, criterion validity, and convergent validity.

- **Content validity.** Content validity refers to the extent to which the estimate captures the concept being evaluated. Content validity is largely a subjective evaluation of whether a study has been designed and executed in a way that incorporates the essential characteristics of the WTP concept. In a sense, it is akin to asking, "On the face of it, does the estimate capture the concept of WTP?" (This approach is sometimes referred to as "face validity.")

  To evaluate a survey instrument, analysts look for features that researchers should have incorporated into the survey scenario. First, the environmental change being valued should be clearly defined. 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 should be included. Next, the action or policy change should be described, including an illustration of how and when it would affect aspects of the environment that people might care about. Boyd and Banzahf (2007), and Boyd and Krupnick (2009) put a finer point on this concept and advocate developing the valuation scenario based on "ecological endpoints" rather than intermediate goods that are less clearly associated with outcomes of interest. For example, if respondents ultimately care about the survival of a certain species, it is more sensible to structure questions to ask about WTP for the species' survival than to ask about degradation of habitat, as respondents are unlikely to know the relationship between habitat attributes and species survival. Respondent attitudes about the provider and the implied property rights of the survey scenario can be used to evaluate the appropriateness of features related to the payment mechanism (Fischhoff and Furby 1988). Survey questions that probe for respondent comprehension and acceptance of the commodity scenario can offer important indications about the validity of the results (Bishop et al. 1997).

- **Criterion validity.** Criterion validity assesses whether stated preference results relate to other measures that are considered to be closer to the concept being assessed (WTP). Ideally, one would compare results from a stated preference study (the measure) with those from actual market data (the criterion). This is because market data can be used to estimate WTP more reliably than a stated preference survey. Another approach would be to estimate a sample of individuals' WTP for a commodity using a stated preference survey and then later give the same sample of individuals or a different random sample of individuals drawn from the same population a real opportunity to buy the good. (See Mitchell and Carson 1989, Carson et al. 1987a, Kealy et al. 1990, Brown et al. 1996, and Champ et al. 1997 for examples.)

  When unable to conduct such comparisons, sensitivity to scope and income has been used to assess criterion validity. "Scope tests" are concerned with how WTP responds to changes in the amount of the referenced good provided in the valuation scenario (Smith and Osborne 1996, Rollins and Lyke 1998, and Heberlien et al. 2005). If the referenced good is indeed a "normal good" utility theory implies that WTP should increase with the provision of the good. For the same reason one would expect WTP to exhibit positive income elasticity (McFadden 1994, and Schlapfer 2006). Neither test is necessary or sufficient to establish criterion validity (Heberlein et al. 2005) but can serve as useful proxies when an alternate measure of WTP for the same good is unavailable. Diamond (1996) suggests that stronger scope tests can be conducted by comparing departures from strict "adding up" of WTP for partial changes and relating them to the income elasticity of WTP. Other researchers, however, argue that the Diamond test may not be practicable or even necessarily correct (Carson et al. 2001).

- **Convergent validity.** Convergent validity examines the relationship between different measures of a concept.[40] This differs from criterion validity in that one of the measures is not taken as a criterion upon which to judge the other measure. The measure of interest and the other measure are judged together to assess consistency with one another. If they differ in a systematic way (e.g., one is usually larger than another for the same good), it is not clear which one is more correct. However, if stated preference results are found to be larger than revealed preference results for the same good, it is often presumed that the difference is the result of hypothetical bias because revealed preference results are based on actual behavior. There can be many other sources of bias and error in both stated preference and revealed preference results that cause them to differ from one another and from "true" WTP.

  Empirical convergent validity tests use comparisons of stated preference results with revealed preference or experimental results that are thought to be free of hypothetical bias.[41] In some circumstances, convergent validity tests may be incorporated as part of the study design. Such a test might compare results of an actual market exercise with the results of a hypothetical market exercise in which the exercises are otherwise identical. In this case there might be evidence of an upward or downward bias in the hypothetical results as compared to the simulated market results. See Section 7.3.3 for a discussion on combining revealed preference and stated preference data.

Hypothetical bias occurs when the responses to hypothetical stated preference questions are

40 Mitchell and Carson (1989) define convergent validity and theoretical validity as two types of construct validity. Construct validity examines the degree to which the measure is related to other measures as predicted by theory.

41 Some analysts include the comparisons of stated preference results to actual markets under convergent validity rather than criterion validity, as discussed in the previous section, because there is no actual observable measure of the theoretical construct WTP. Here, a distinction is made between simulated markets, as in a laboratory experiment in which values may be "induced" by giving subject cash at the end based on their choices, and actual markets in which subjects must pay with their own money.

systematically different than what individuals would pay if the transactions were to actually occur. Widely cited as one of the most common problems with the stated preference method (List and Gallet 2001, and Murphy and Allen 2005), and researchers have made advances in techniques to minimize such bias. These techniques include the use of "cheap talk" methods to directly tell respondents about the potential for hypothetical bias (Cummings and Taylor 1999, and List 2001); calibrating hypothetical values (List and Shogren 1998, and Blomquist et al. 2009); and allowing respondents to express uncertainty in their responses and restricting the set of positive responses to those about which the respondent was most certain (Vossler et al. 2003). Several studies have shown that attribute-based choice experiments reduce hypothetical bias in the bid amounts and the marginal value of attributes relative to other elicitation methods (Carlsson and Martinsson 2001, Murphy and Allen 2005, and List et al. 2006).

Tests for hypothetical bias often involve a comparison of actual payments and responses to hypothetical scenarios that use the same solicitation approach. The actual payments typically occur in one of three scenarios. Market transactions are the most common (Cummings et al. 1995, and List and Shogren 1998) but generally involve payments for private goods while most stated preference applications are concerned with public or quasi-public goods. Simulated markets can be used to solicit actual donations for public good provision (Champ et al. 1997). However, donation solicitations are subject to free riding, so while it may be possible to test for hypothetical bias using this approach, both the actual and hypothetical payment scenarios lack incentive compatibility and may not represent total WTP. In rare instances comparisons have been made between actual referenda for public good provision and hypothetical responses to the same scenario but the conditions for a valid comparison of this sort are exceedingly difficult to satisfy (Johnston 2006).

Non-response bias is introduced when non-respondents would have answered questions systematically differently than those who did answer. Non-response bias can take two forms: item non-response and survey non-response.

- **Item non-response bias** occurs when respondents who agreed to take the survey do not answer all of the choice questions in the survey. Information available about respondents from other questions they answered can support an assessment of potential item non-response bias for the WTP questions that were unanswered. The key issue is whether there were systematic differences in potential WTP-related characteristics of those who answered the WTP questions and those who did not. Characteristics of interest include income, gender, age, expressed attitudes and opinions about the good or service, and information reported on current use or familiarity with the good or service. Statistically significant differences may indicate the potential for item non-response bias, while finding no such differences suggests that the chance of significant non-response bias is lower. However, the results of this comparison are only suggestive because respondents and non-respondents may only differ in their preference for the good in question (McClelland et al. 1991).

- **Survey non-response bias** is created when those who refuse to take the survey have preferences that are systematically different from the preferences of those who do respond. Although it is generally thought that surveys with high response rates are less likely to suffer from survey non-response bias, it is not a guarantee.[42] For survey non-respondents, there may be no available data to determine how they might systematically differ from those who responded to the survey. The

42 Note that OMB's *Guidance on Agency Survey and Statistical Collections* (OMB 2006) has fairly strict requirements for response rates and their calculation for Agency-sponsored surveys, recommending that "ICRs for surveys with expected response rates of 80 percent or higher need complete descriptions of the basis of the estimated response rate...ICRs for surveys with expected response rates lower than 80 percent need complete descriptions of how the expected response rate was determined, a detailed description of steps that will be taken to maximize the response rate...and a description of plans to evaluate non-response bias" (pp. 60-70).

most common approach is to examine the relevant measurable characteristics of the respondent group, such as income, resource use, gender, age, etc., and to compare them to the characteristics of the study population. Similarity in mean characteristics across the two groups suggests that the respondents are representative of the study population and that non-response bias is expected to be minimal.

A second way to evaluate potential survey non-response bias is to conduct a short follow-up survey with non-respondents. This can sometimes be accomplished through interviews conducted during the recruiting phase. Such follow-ups typically ask a few questions about attitudes and opinions on the topic of the study as well as collecting basic socioeconomic information. Questions need to match those in the full survey closely enough to compare non-respondents to respondents. The follow-up must be very brief or response rates will be low (OMB 2006).

7.3.3 Combining Revealed and Stated Preference Data

Instead of looking at revealed preference and stated preference data as two separate methods for estimating environmental benefits, an increasing number of researchers are using them in combination. The practice has been in use much longer in the marketing and transportation literature and many of the lessons learned by those researchers are now being employed in environmental economics. In theory, the strengths of each data type should help overcome some of the weaknesses of the other. As described by Whitehead et al. (2008) in a recent assessment of the state of the science, the advantages of combining revealed preference and stated preference data include:

- Helping to ground the hypothetical stated preference data with real world behavior potentially decreasing any hypothetical bias;
- Providing the ability to test the validity of both data sources;[43]
- Increasing the range of historical stated preference data to include conditions not observed in the past and thereby reducing the need to make predictions outside of the sample;
- Increasing the sample size;
- Extending the size of the market or population to include larger segments than captured by either method alone; and
- Exploiting the flexibility of stated preference experimental design to overcome revealed preference data's potential multicollinearity and endogeneity problems (von Haefen and Phaneuf 2008).

The different strategies for combining revealed preference and stated preference data can be roughly grouped into three main methods. The first two methods rely on joint estimation. If the revealed preference and stated preference data have similar dependent and independent variables and the same assumed error structures, then they can simply be pooled together and treated as additional observations (Adamowicz et al. 1994; Boxall, Englin, and Adamowicz 2003; and Morgan, Massey, and Huth 2009). If the revealed preference and stated preference data sources cannot be pooled, it is sometimes possible to use them in a jointly estimated mixed model that relies on a utility theoretic specification of the underlying WTP function (Huang, Haab, and Whitehead 1997; Kling 1997; and Eom and Larson 2006). If the data cannot be combined in estimation, it can still be useful to estimate results separately and then use them to test for convergent validity between the two data sources (Carson et al. 1996, and Schlapfer et al. 2004).

7.4 Benefit Transfer

Benefit transfer refers to the use of estimated non-market values of environmental quality changes from one study in the evaluation of a different policy that is of interest to the analyst (Freeman 2003, p. 453). The case under consideration for a

43 Herriges, Kling, and Phaneuf (2004) point out that revealed preference may not always be valid for estimating WTP for quality changes when weak complementarity cannot be assured.

new policy is referred to as the "policy case." Cases from which estimates are obtained are referred to as "study cases." A benefit transfer study identifies stated preference or revealed preference study cases that sufficiently relate to the policy context and "transfers" their results to the policy case.

Benefit transfer is necessary when it is infeasible to conduct an original study focused directly on the policy case. Original studies are time consuming and expensive; benefit transfer can reduce both the time and financial resources required to develop estimates of a proposed policy's benefits. While benefit transfer should only be used as a last resort and a clear justification for using this approach over conducting original valuation studies should be provided (OMB 2003), the reality is that benefit transfer is one of the most common approaches for completing a BCA at EPA. However, the advantages of benefit transfer in terms of time and cost savings must be weighed against the disadvantages in terms of potential reduced reliability of the final benefit estimates. The transfer of benefits estimates from any single study case is unlikely to be as accurate as a primary study tailored specifically to the policy case, although it is difficult to characterize the uncertainty associated with transferred benefits estimates.

The number and quality of relevant studies available for application to the policy case can limit the use of benefit-transfer methods.[44] Even when a study case is qualitatively similar to the policy case, the environmental change associated with the policy case may be of a different scope or nature than the changes considered in the study cases. In addition, methodological advances and changes in demographic, economic, and environmental conditions over time may make otherwise suitable studies obsolete.[45]

44 One possible reason that a relatively limited number of value estimates exist in peer-reviewed literature is that researchers and editors of scholarly journals may be more interested in new theoretical or methodological advances than in studies that apply established valuation methods to confirm earlier findings.

45 A 2006 special issue of *Ecological Economics* (volume 60) focused exclusively on benefit transfer for environmental policy, covering diverse topics such as publication bias, theoretical motivation and emerging issues. Florax et al. (2002), and Navrud and Ready (2007) are two general references for benefit transfer studies.

Steps for conducting benefit transfer

While there is no universally accepted single approach for conducting benefit transfer there are some generalized steps involved in the process. These steps are described below.

1. Describe the policy case. The first step in a benefit-transfer study is to clearly describe the policy case so that its characteristics and consequences are well understood. Are human health risks reduced by the policy intervention? Are ecological benefits expected (e.g., increases in populations of species of concern)? It is also important to identify to the extent possible the beneficiaries of the proposed policy and to describe their demographic and socioeconomic characteristics (e.g., users of a particular set of recreation sites, children living in urban areas, or older adults across the United States). Information on the affected population is generally required to translate per person (or per household) values to an aggregate benefits estimate.

2. Select study cases. A benefit-transfer study is only as good as the study cases from which it is derived, and it is therefore crucial that studies be carefully selected. First, the analyst should identify potentially relevant studies by conducting a comprehensive literature search. Because peer-reviewed academic journals may be more likely to publish work using novel approaches compared to established techniques, some studies of interest may be found in government reports, working papers, dissertations, unpublished research, and other "gray literature."[46] Including studies from the gray literature may also help mitigate "publication bias" that results from researchers being more likely to present and/or editors being more likely to publish studies that demonstrate statistically significant results, or results that are of an expected sign or magnitude.[47] Online searchable databases

46 Peer review of benefit-transfer studies using gray literature is highly advisable.

47 There is some evidence of publication bias towards studies showing statistically significant results. For example, in a meta-analysis of studies in labor economics, Card and Krueger (1995) argue that just-significant results are reported more frequently than would be predicted by chance. Similar practices may prevail in other areas of economic research. Combining results from a group of studies that suffer from publication bias may lead to inaccurate conclusions. See Stanley (2005, 2008) for a discussion of methods to correct for and identify publication bias.

summarizing valuation research may be especially helpful at this stage.[48]

Next, the analyst should develop an explicit set of selection criteria to evaluate each of the potentially relevant studies for quality and applicability to the policy case. The quality of the value estimates in the study cases will in large part determine the quality of the benefit transfer. As a first step, the analyst should review studies according to the criteria listed for each methodology in the previous sections in this chapter. Results from study cases must be valid as well as relevant. Concerns about the quality of the studies, as opposed to their relevance, will generally hinge on the methods used. Valuation approaches commonly used in the past may now be regarded as unacceptable for use in benefits analysis. Studies based on inappropriate methods or reporting obsolete results should be removed from consideration.

It is unlikely that any single study will match perfectly with the policy case; however each potential study case should inform at least some aspect of the policy decision. Study cases potentially suitable for use in benefit transfer should be similar to the policy case in their: (1) definition of the environmental commodity being valued (include scale and presence of substitutes); (2) baseline and extent of environmental changes; and (3) characteristics of affected populations. Analysts should avoid using benefit transfer in cases where the policy or study case is focused on a "good" with unique attributes or where the magnitude of the change or improvement across the two cases differs substantially (OMB 2003).[49]

48 For example, the EVRI is maintained by Environment Canada and managed by a working group that includes the U.S. EPA and members of the European Union. EVRI contains over 1,100 studies that can be referenced according to medium, resource, stressor, method, and country. EVRI also provides a bibliography on benefit transfer. See www.evri.ca for more information. Envalue, developed by the New South Wales EPA in 1995, is similar: Studies can be identified according to medium, stressor, method, country, and author.

49 In some cases the transfer method itself may inform the choice of study cases to include. For example, meta-analysis approaches (discussed below) can facilitate some forms of statistical validity testing (Hunter and Schmidt 1990, and Stanley 2001), so some otherwise suitable studies may be rejected as "outliers."

The analyst should determine whether adjustments should and can be made for important differences between each study and policy case. For example, some case studies will report Marshallian demand while others may report Hicksian demand.[50] The ability of the analyst to make these adjustments will depend, in part, on both the number of value estimates for suitably similar study sites and the method used to combine these estimates. These methods are now discussed in turn.

3. Transfer values. There are several approaches for transferring values from study cases to the policy case. These include unit value transfers, value function transfers, and non-structural or structural meta-analysis. Each of these approaches is typically used to develop per person or per household value estimates that are then aggregated over the affected population to compute a total benefits estimate. As a general rule, the more related case study estimates involved in a benefit transfer, the more reliable the estimate.

Unit value transfers are the simplest of the benefit-transfer approaches. They take a point estimate of WTP for a unit change in the environmental resource from a study case or cases and apply it directly to the policy case. The point estimate is commonly a single estimated value from a single case study, but it can also be the (otherwise unadjusted) average of a small number of estimates from a few case studies. For example, a study may have found a WTP of $20 per household for a one-unit increase on some water quality scale. A unit value transfer would estimate total benefits for the policy case by multiplying $20 by the number of units by which the policy is expected to increase water quality and by the number of households who will benefit from the change. This approach can be useful for developing preliminary, order-of-magnitude estimates of benefits, but it should be possible to base final benefit estimates on more

50 See Desvousges et al. (1992), Brouwer (2000), Florax et al. (2002), Bergstrom and Taylor (2006), and Navrud and Ready (2007) for additional information on criteria used to determine quality and applicability. For more information on applicability as related to specific benefit categories, see Desvousges et al. (1998), the draft *Handbook for Non-Cancer Valuation* (U.S. EPA 2000c), and the *Children's Health Valuation Handbook* (U.S. EPA 2003b). It may also be useful for the analyst to discuss her interpretation and intended use of the study case with the original authors.

Text Box 7.6 - The Benefits and Costs of the Clean Air Act 1990 to 2010: Reduced Acidification in Freshwater Adirondack Lakes

One component of the total benefits of the Clean Air Act (CAA) was determined to be improved recreational fishing due to reduced acidification in freshwater Adirondack lakes. To value this benefit, EPA relied on the results of Montgomery and Needleman's (1997) New York State Adirondack region recreational fishing study. EPA first developed estimates of the percentage Adirondack of lakes affected by acidification pre and post CAA. Then, using a probit model, the likelihood that each individual lake would become acidified was estimated (the model relates acidity to lake characteristics such as elevation, surface area, watershed, and others) and the lakes were ranked from highest to lowest probability of being acidified. The acidification status of individual lakes in the choice set was then assigned, starting with the highest probability lake and proceeding down until the appropriate number of lakes affected under each scenario (i.e., the estimated percentage of lakes affected) was achieved. Using these lake designations and the Montgomery and Needleman model's estimated coefficients, welfare was calculated for the pre and post CAA levels of lake acidification. The difference between the two welfare estimates was assumed to be the value of improved Adirondack freshwater recreational fishing under the CAA.

information than a single point estimate from a single study. Point estimates reported in study cases are typically functions of several variables, and simply transferring a summary estimate without controlling for differences among these variables can yield inaccurate results. It is important to recognize that unit value transfer assumes that the original good, as well as the characteristics and tastes of the population of beneficiaries, are the same as the policy good. Unit values transfers should only be used if the case and policy studies are evaluating the same environmental good, the same change in environmental levels, and same affected populations.

Function transfers also rely on a single study, but they use information on other factors that influence WTP to adjust the unit value for quantifiable differences between the study case and the policy case. This is accomplished by transferring the estimated function upon which the value estimate in the study case is based to the policy case. This approach implicitly assumes that the population of beneficiaries to which the values are being transferred has potentially different characteristics, but similar tastes, as the original one and allows the analyst to adjust for these different characteristics. Generally, benefit function transfers are preferable to unit value transfers as they incorporate information relevant to the policy scenario (OMB 2003). For example, suppose that in the hypothetical example above the $20 unit value was the result of averaging the results of an estimated WTP function over all individuals in the study case sample, where the WTP function included income, the baseline water quality level, and the change in the water quality level for each household. A function transfer would estimate total benefits for the policy case by:

1. Applying the WTP function to a random sample of households affected in the policy case using each household's observed levels of income, baseline water quality, and water quality change;
2. Averaging the resulting WTP estimates; and
3. Multiplying this average WTP by the total number of households affected in the policy case.

See Text Boxes 7.6 and 7.7 for examples of value and function transfers.

If the WTP function is nonlinear and statistics on average income, baseline water quality, and water quality changes are used in the transfer instead of household level values, then bias would result. Feather and Hellerstein (1997) provide an example of a function transfer that attempts to correct for such bias. Although unit transfers can adjust and compensate for small differences between the case and policy study populations, they are subject to the same basic usage rules governing unit value transfers. Function transfers should only be used if the case and policy studies are evaluating very similar environmental goods, change in environmental levels, and affected populations.

Text Box 7.7 - Benefits Transfer: Water Quality Benefits in the Combined Animal Feeding Operations Rule

There are two prominent water quality benefit-transfer applications in the 2002 Combined Animal Feeding Operations (CAFO) rule. The first looks at the recreational value of water quality improvements in fresh water lakes and streams (see Section 4 of U.S. EPA 2002c). Field pollutant loadings were modeled by the National Water Pollution Control Assessment Model (NWPCAM) to produce pre and post regulation water quality estimates. Predicted changes in water quality were then valued using the results of Carson and Mitchell's (1993) national water quality contingent valuation survey. First, benefits were calculated based on estimates of willingness to pay (WTP) for water quality improvements resulting in discrete movements to higher "rungs" of the water quality ladder (boatable, fishable, swimmable, drinkable). Very simply described, Carson and Mitchell's "in-state" WTP estimates for discrete movements up the water quality ladder were multiplied by the number of affected residents in every state and "out-of-state," non-use values were multiplied times the remaining population. State totals were then summed up to a national total (see Appendix A-4 of U.S. EPA 2002c for more details). Benefits were also estimated a second way based on a continuous (1 to 100) water quality index constructed from six water quality parameters measured in the NWPCAM model. The minimum thresholds between rungs on the water quality ladder were then translated into points along the continuous water quality index (i.e., boatable = 25, fishable = 50, swimmable = 70). Carson and Mitchell's WTP function was then used to value changes in water quality as measured by the water quality index (see Appendix B-4 of U.S. EPA 2002c for more details). Benefits estimated by the water quality index method are larger by roughly a factor of two (Exhibits 4-12 and 4-13 of U.S. EPA 2002c).

The second major benefit-transfer application in the CAFO rule involves the valuation of reduced eutrophication in estuaries (Section 9 of U.S. EPA 2002c). EPA used a case study of Albemarle and Pamlico sounds to demonstrate the potential importance and value of reduced eutrophication on recreational fishing in affected estuaries. Again, NWPCAM was used to estimate pre and post regulation water quality levels. In this case, the benefit transfer made use of three studies (Kaoru 1995; Kaoru, Smith, and Liu 1995; and Smith and Palmquist 1988), all of which were based in part on the same dataset. All "reasonable" estimates of WTP for reduced phosphorus or nitrogen from the studies were retained and translated into their corresponding dollar per trip per ton reduction in pollutant per year value. A range of total benefits was then calculated by multiplying each $/trip/ton/year estimate by the number of trips taken and the change in loadings (in tons) for each pollutant (see Exhibit 9-3 of U.S. EPA 2002c).

Meta-analysis uses results from multiple valuation studies to estimate a new transfer function. Meta-analysis is an umbrella term for a suite of techniques that synthesize the summary results of empirical research. This could include a simple ranking of results to a complex regression. The advantage of these methods is that they are generally easier to estimate while controlling for a relatively large number of confounding variables. This approach has been widely used in environmental economics (Poe et al. 2001, Shrestha and Loomis 2003a and 2003b, Rosenberger and Loomis 2000, and Bateman and Jones 2003).

There are a number of guidelines for meta-analyses that outline protocols that should be followed in conducting or evaluating a study. See Begg et al. (1996), Moher et al. (1999), and U.S. EPA (2006e) for more information.[51] More recently Bergstrom and Taylor (2006) discuss the theory and practice underlying meta-analysis for benefit transfer, discussing three major necessary steps: theory, data collection, and analysis. In general, when reporting meta-analysis results, researchers should provide information on the background of the problem, the strategy for selecting studies, analytic methods, results, discussion, and conclusions. See U.S. EPA (2006e) for a detailed discussion of meta-analysis as applied to VSL estimates. U.S. EPA (2006e) specifically recommends carefully specifying the search process, selection criteria, and analytical methods.

Structural benefit transfer is a relatively new approach to benefit transfer. The advantages of

51 The last reference contains a detailed discussion of the protocols for conduction a meta-analsysis.

Text Box 7.8 - Structural Benefit Transfer with an Application to Visibility

U.S. EPA (2006b) employs a structural benefit transfer to derive values for visibility improvements associated with the Particulate Matter (PM) National Ambient Air Quality Standards (NAAQS). It specified a constant elasticity of substitution utility function for visibility in residential and Class I (national park and similar) areas. This function assumes that the value for Class I visibility differs in and out of region but that residential visibility is valued the same everywhere. EPA also assumed that in-region visibility was valued more highly than out-of-region visibility. The function further specified utility as a function of: (1) consumption of all goods; (2) visibility in a person's residential area; (3) recreational visibility in a person's residential region; and (4) recreational visibility outside of a person's residential region. Given the utility function and a budget constraint, it was then possible to define households' WTP for changes in visibility as a function of income and visibility measures. The regional preference parameters of the function were calibrated using existing WTP estimates for visibility in Class I areas (Chestnut and Rowe 1990, and Chestnut 1997) if estimates existed for a given region. If not, estimates were adjusted by visitation rate. The preference parameter for residential visibility was assumed to be the same in all counties and was solved for based on a WTP estimate presented in McClelland et al. (1991). With estimates of visibility (pre and post regulation), county-level income, and the required preferences parameters, nationwide estimates of the value of increased visibility were then computed for each of the six regions of the country.

structural transfer functions are that they can accommodate different types of economic value measures (e.g., WTP, WTA, or consumer surplus) and can be constructed in such a way that certain theoretical consistency conditions (e.g., WTP bounded by income) can be satisfied. This could be applied to value transfer, function transfer, or meta-analysis; although applications to function transfer are the most common. Structural transfer functions that have been estimated have specified a theoretically consistent preference model that is calibrated according to existing benefit estimates from the literature (see Smith and Pattanayak 2002; and Smith, Pattanayak, and van Houtven 2006 for descriptions on the method). See Text Box 7.8 for an application to of structural benefit transfer to visibility benefits.

4. Report the results. In addition to reporting the final benefit estimates from the transfer exercise, the analyst should clearly describe all key judgments and assumptions, including the criteria used to select study cases and the choice of the transfer approach. The uncertainty in the final benefit estimate should be quantified and reported when possible. (See Chapter 11 on Presentation of Analysis and Results.)

7.5 Accommodating Non-Monetized Benefits

It often will not be possible to quantify all of the significant physical impacts for all policy options. For example, animal studies may suggest that a contaminant causes severe illnesses in humans, but the available data may not be adequate to determine the number of expected cases associated with different human exposure levels. Likewise, it often is not possible to quantify the various ecosystem changes that may result from an environmental policy. While Chapter 11 discusses how to present these benefits so as to provide a fuller accounting of all effects, this section discusses what analysts can do to incorporate these endpoints more fully into the analysis.

7.5.1 Qualitative Discussions

When there are potentially important effects that cannot be quantified, the analyst should include a qualitative discussion of benefits results. The discussion should explain why a quantitative analysis was not possible and the reasons for believing that these non-quantified effects may be important for decision making. Chapter 11 discusses how to describe benefit categories that are quantified in physical terms but not monetized.

7.5.2 Alternative Analyses

Alternative analyses exist that can support benefits valuation when robust value estimates and/or risk estimates are lacking. These analyses, including break-even analysis and bounding analysis, can provide decision makers with some useful information. However analysts should remember that because these alternatives do not estimate the net benefits of a policy or regulation, they fall short of BCA in their ability to identify an economically efficient policy. This and other shortcomings should be discussed when presenting results from these analyses to decision makers.

7.5.2.1 Break-Even Analysis

Break-even analysis is one alternative that can be used when either risk data or valuation data are lacking.[52] Analysts who have per unit estimates of economic value but lack risk estimates cannot quantify net benefits. They can, however, estimate the number of cases (each valued at the per unit value estimate) at which overall net benefits become positive, or where the policy action will break even.[53] Consider a proposed policy that is expected to reduce the number of cases of endpoint X with an associated cost estimate of $1 million. Further, suppose that the analyst estimates that WTP to avoid a case of endpoint X is $200, but that because of limitations in risk data, it is not possible to generate an estimate of the number of cases of this endpoint reduced by the policy. In this case, the proposed policy would need to reduce the number of cases by 5,000 in order to "break even." This estimate then can be assessed for plausibility either quantitatively or qualitatively. Policy makers will need to determine if the break-even value is acceptable or reasonable.

The same sort of analysis can be performed when analysts lack valuation estimates, producing a break-even value that should again be assessed for credibility and plausibility. Continuing with the example above, suppose the analyst estimates that the proposed policy would reduce the number of cases of endpoint X by 5,000 but does not have an estimate of WTP to avoid a case of this endpoint. In this case, the policy can be considered to break even if WTP is at least $200.

One way to assess the credibility of economic break-even values is to compare them to risk values for effects that are more or less severe than the endpoint being evaluated. For the break-even value to be plausible, it should fall between the estimates for these more and less severe effects. For the example above, if the estimate of WTP to avoid a case of a more serious effect was only $100, the above break-even point may not be considered plausible.

Break-even analysis is most effective when there is only one missing value in the analysis. For example, if an analyst is missing risk estimates for two different endpoints (but has valuation estimates for both), then they will need to consider a "break-even frontier" that allows the number of both effects to vary. It is possible to construct such a frontier, but it is difficult to determine which points on the frontier are relevant for policy analysis.

7.5.2.2 Bounding Analysis

Bounding analysis can help when analysts lack value estimates for a particular endpoint. As suggested above, reducing the risk of health effects that are more severe and of longer duration should be valued more highly than those that are less severe and of shorter duration, all else equal. If robust valuation estimates are available for effects that are unambiguously "worse" and others that are unambiguously "not as bad," then one can use these estimates as the upper and lower bounds on the value of the effect of concern. Presenting alternative benefit estimates based on each of these bounds can provide valuable information to policy makers. If the sign of the net benefit estimate is positive across this range then analysts can have some confidence that the program is welfare enhancing. Analysts should carefully describe judgments or assumptions made in selecting appropriate bounding values.

52 Boardman et al. (1996) describes determining break-even points under the general subject of sensitivity analysis and includes empirical examples.

53 *Circular A-4* (OMB 2003) refers to these values as "switch points" in its discussion of sensitivity analysis.

Chapter 8
Analyzing Costs

The previous chapter discussed the process of estimating the benefits of environmental regulations and policies. This chapter discusses the estimation of costs, with a primary focus on estimating costs for use in benefit-cost analyses (BCA). While often portrayed as being relatively straightforward — particularly compared to the estimation of benefits — the estimation of costs presents a number of challenges in its own right.

The first challenge is to identify an appropriate measure of cost for a particular application. A number of concepts of cost exist, with some overlap of ideas. In conducting a BCA, the correct measure to use is the social cost. Social cost represents the total burden that a regulation will impose on the economy. It is defined as the sum of all opportunity costs incurred as a result of a regulation where an opportunity cost is the value lost to society of any goods and services that will not be produced and consumed as a result of a regulation.

A second challenge involves choosing an economic framework for the analysis. Depending on the scope of the regulation or policy, either a partial or general equilibrium framework is employed. Partial equilibrium analysis is usually appropriate when the scope of a regulation is limited to a single sector, or to a small number of sectors. General equilibrium analysis may be more appropriate if the analyst expects a large number of sectors to be impacted and that the effects will be spread more broadly throughout the economy.

The third challenge is choosing one or more models to use in an analysis. Factors to consider in selecting a model include the types of costs being investigated, the geographic and sectoral scope of the likely impacts, and the expected magnitude of the impacts. For some analyses, it may be necessary to use more than one model.

This chapter discusses social cost and its underlying economic theory as well as several alternative concepts of cost. In addition, the chapter discusses several additional issues in cost estimation and presents a number of the models that can be employed in the estimation and analysis of costs.

8.1 The Economics of Social Cost

The most comprehensive measure of the costs of a regulation — and thus the appropriate measure to use in a BCA — is "social cost." Social cost represents the total burden a regulation will impose on the economy; it can be defined as the sum of all opportunity costs incurred as a result of the regulation. These opportunity costs consist of the value lost to society of all the goods and services that will not be produced and consumed if firms comply with the regulation and reallocate resources away from production activities and towards pollution abatement. To be complete, an estimate of social cost should include both the opportunity costs of current consumption that will be foregone as a result of the regulation, and the losses that may result if the regulation reduces capital investment and thus future consumption.[1]

The purpose of estimating social cost is to have a reference point for comparing the costs of a regulation with the estimated benefits. Social cost is not a particularly meaningful concept unless it is used as part of a net social welfare calculation, or perhaps compared to other (less comprehensive) cost measures.[2] Conceptually, it should be noted that the social cost of a regulation is generally not the same as a change in gross domestic product (GDP), or another broad measure of economic activity, that may result from its imposition. Expenditures on inputs into pollution abatement, such as equipment, materials, and labor, are counted as part of social cost. All or part of their consumption will at the same time be included positively in the calculation of GDP. Thus, if a regulation has the effect of lowering GDP, this decline will in general be less than the social cost of the regulation.

Two broad analytical paradigms are used in the analysis of social cost: partial equilibrium and general equilibrium. A partial equilibrium approach is appropriate when it is assumed that the effects of a regulation will primarily be confined to a single or small number of closely related markets. If this is not the case, and the regulation is expected to cause significant impacts across the economy, it is more appropriate to use general equilibrium analysis to estimate social cost. The use of these two analytical paradigms is explored in the following sections.

1 This section discusses the prospective estimation of social cost for regulations that have not yet been implemented. However, the same principles apply to estimating costs retrospectively for regulations already in place. Likewise, while the text refers to the social cost of "a regulation" the same principles apply to the estimation of the social cost for each alternative in a set of regulatory alternatives. For a more rigorous and detailed treatment of the material in this section, see Pizer and Kopp (2005).

2 For example, comparing the social cost of different regulations may provide some sense of the relative burden they impose on the economy, but this exercise alone would not indicate which, if any, of the regulations may be worthwhile from a public policy standpoint. However, the accurate measurement of social cost would be an essential component in attempting to make such a determination.

8.1.1 Partial Equilibrium Analysis

When the analyst expects that the effects of a regulation will be confined primarily to a single market or a small number of markets, partial equilibrium analysis is the preferred approach for estimation of social cost. The use of partial equilibrium analysis assumes that the effects of the regulation on all other markets will be minimal and can either be ignored or estimated without employing a model of the entire economy. This section presents some simple diagrams to show how social cost can be defined in a partial equilibrium framework.

Figure 8.1 shows a competitive market before the imposition of an environmental regulation. The intersection of the supply (S_0) and demand (D) curves determines the equilibrium price (P_0) and quantity (Q_0). The shaded area below the demand curve and above the equilibrium price line is the consumer surplus. The area above the supply curve and below the price line is producer surplus. The sum of these two areas defines the total welfare generated in this market: the net benefits to society from producing and consuming the good or service represented in this market.[3]

In this market, the imposition of a new environmental regulation raises firms' production costs. Each unit of output is now more costly to produce because of expenditures incurred to comply with the regulation. As a result, firms will respond by reducing their level of output. For the industry, this will appear as an upward shift in the supply curve. This is shown in Figure 8.2 as a movement from S_0 to S_1. The effect on the market of the shift in the supply curve is to increase the equilibrium price to P_1 and to decrease the equilibrium output to Q_1, holding all else constant.

3 It should be noted that total welfare as depicted ignores the negative pollution externality arising in this market, which the environmental regulation is designed to correct. Appendix A presents a graphical description of how to account for this externality. Reduction of this negative externality would be quantified in the benefits portion of an analysis. The supply curve in Figure 8.1 corresponds to the marginal private cost (MPC) curve described in Figure A.5 of Appendix A.

Figure 8.1 - Competitive Market Before Regulation

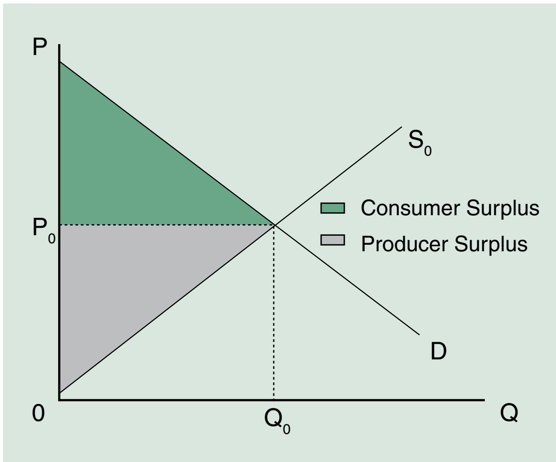
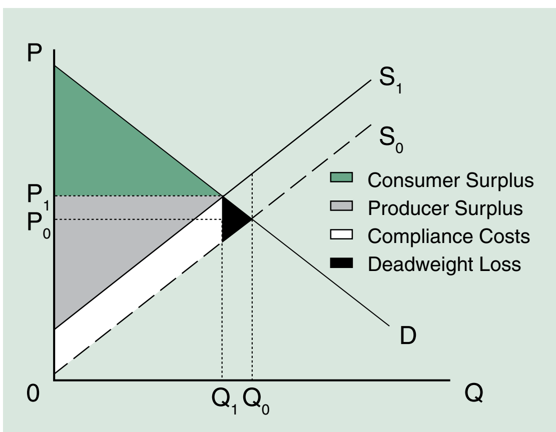


Figure 8.2 - Competitive Market After Regulation



As seen by comparing Figures 8.1 and 8.2, the overall effect on welfare is a decline in both producer and consumer surplus.[4]

Compliance costs in this market are equal to the area between the old and new supply curves, bounded by the new equilibrium output, Q_1.[5] Noting this, a number of useful insights about the total costs of the regulation can be derived from Figures 8.1 and 8.2. First, when consumers are price sensitive — as reflected in the downward sloping demand curve — a higher price causes them to reduce consumption of the good. If costs are estimated ex ante and this price sensitive behavior is not taken into account (i.e., the estimate is based on the original level of output (Q_0) compliance costs will be overstated. Extending the vertical dotted line in Figure 8.2 from the original equilibrium to the new supply curve (S_1) illustrates this point.[6]

A second insight derived from Figures 8.1 and 8.2 is that compliance costs are usually only part of the total costs of a regulation. The "deadweight loss" (DWL) shown in Figure 8.2 is an additional, real cost arising from the regulation. It reflects the foregone net benefit due to the reduction in output.[7] Moreover, unlike many one-time compliance costs, DWL will be a component of social cost in future periods.

Under the assumption that impacts outside this market are not significant, then the social cost of the regulation is equal to the sum of the compliance costs and the deadweight loss (shown in Figure 8.2). This is exactly equal to the reduction in producer and consumer surplus from the pre-regulation equilibrium (shown in Figure 8.1). This estimate of social cost would be the appropriate measure to use in a BCA of the regulation. As noted above, if some of the compliance costs are spent on other goods and services or on hiring additional labor, any fall in GDP attributable to the imposition of the regulation will be less than the social cost.

The preceding discussion describes the use of partial equilibrium analysis when the regulated

4 The figure depicts an equal distribution of welfare between consumers and producers, in both the old and new equilibria. Depending on the elasticities of supply and demand, this may not be the case. The elasticities will determine the magnitude of the price and quantity changes induced by the cost increase, as well as the distribution of costs.

5 Here distinctions between the fixed and variable costs of abatement are abstracted and it is assumed that all of the costs are represented in the movement of the supply curve. See Tietenberg (2002).

6 In the extreme, if the regulation raised production costs so much that firms decided to halt production altogether, or if an outright ban on the product was issued, a strict compliance cost analysis would yield zero cost as no direct expenditures on abatement would be made. Clearly this would constitute an underestimate of the loss in consumer welfare.

7 Typically, in a market already distorted with pollution externalities, the DWL triangle shown in Figure 8.2 will serve to offset (at least in part) the existing DWL in the market that results when the real costs of production (including the pollution damages) are not considered in the production decision. Of course, if the regulatory action is too stringent and "over controls" the pollution problem, the optimal outcome will not be achieved and additional DWL will be created. Figure 8.2 is silent on where the optimal solution is achieved. See Appendix A for more detail.

market is perfectly competitive. In many cases, however, some form of imperfect competition, such as monopolistic competition, oligopoly, or monopoly, may better characterize the regulated market. Firms in imperfectly competitive markets will adjust differently to the imposition of a new regulation and this can alter the estimate of social cost.[8] If the regulated market is imperfectly competitive, the market structure can and should be reflected in the analysis.

In certain situations, when the effects of a regulation are expected to impact a limited number of markets beyond the regulated sector, it still may be possible to use a partial equilibrium framework to estimate social cost. Multi-market analysis extends a single-market, partial equilibrium analysis of the directly regulated sector to include closely related markets. These may include the upstream suppliers of major inputs to the regulated sector, downstream producers who use the regulated sector's output as an input, and producers of substitute or complimentary products. Vertically or horizontally related markets will be affected by changes in the equilibrium price and quantity in the regulated sector. As a consequence, they will experience equilibrium adjustments of their own that can be analyzed in a similar fashion.[9]

8.1.2 General Equilibrium Analysis

In some cases, the imposition of an environmental regulation will have significant effects in markets beyond those that are directly subject to the regulation. As the number of affected markets grows, it becomes less and less likely that partial equilibrium analysis can provide an accurate estimate of social cost. Similarly, it may not be possible to accurately model a large change in a single regulated market using partial equilibrium analysis. In such cases, a general equilibrium framework, which captures linkages between markets across the entire economy, may be a more appropriate choice for the analysis.

For example, the imposition of an environmental regulation on emissions from the electric utility sector may cause the price of electricity to rise. As electricity is an important intermediate input in the production of most goods, the prices of these products will most likely also rise. Households will be affected as both consumers of these goods and as consumers of electricity. The increase in prices may cause them to alter their relative consumption of a variety of goods and services. The increase in the price of electricity may also cause feedback effects that result in a reduction in the total consumption of electricity.

General equilibrium analysis is built around the assumption that for some discrete period of time, an economy can be characterized by a set of equilibrium conditions in which supply equals demand in all markets. When the imposition of a regulation alters conditions in one market, a general equilibrium model will determine a new set of prices for all markets that will return the economy to equilibrium. These prices in turn determine the outputs and consumption of goods and services in the new equilibrium. In addition, the model will determine a new set of prices and demands for the factors of production (labor, capital, and land), the returns to which compose the income of businesses and households. Changes in aggregate economic activity, such as GDP, household consumption, and other variables, also can be calculated in the model.

The previous section shows how the social cost of a regulation can be estimated in a single market using partial equilibrium analysis. The example demonstrates how a regulation causes

8 The opportunity costs of lost production from the regulation will be less for a monopoly than a perfectly competitive industry, even if they face the same market demand curve. This result may seem counterintuitive, but the monopolist operates on a more elastic, or price sensitive, portion of the demand curve. As a result, it will have lower profits if it tries to increase price (and lower output) by as much as the competitive industry.

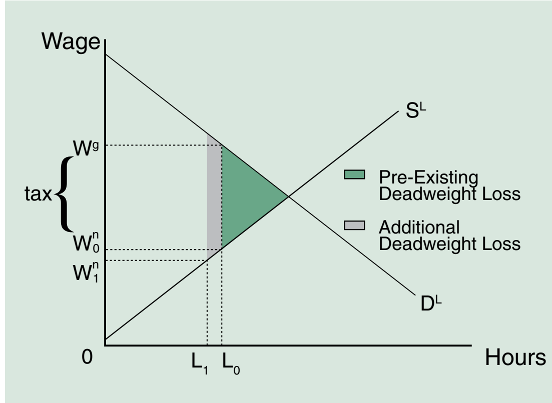
9 In theory, impacts in undistorted related markets are "pecuniary" and do not need to be included if the social costs have been correctly measured in the primary market, but pecuniary effects are important in inefficient related markets and should be considered (Boardman et al. 2006). Just et al. (2005) provide a detailed treatment of multi-market analysis. Kokoski and Smith (1987) demonstrate, however, that one must use caution when using these methods.

a DWL in that market, reflecting a decline in economic welfare as measured by consumer and producer surplus. In reality, DWL already exists in many, if not most, markets as a result of taxes, regulations, and other distortions. When the imposition of a regulation causes a new distortion in one market, it may interact with pre-existing distortions in other markets and this may cause additional impacts on welfare.

An important example of how a regulation can interact with pre-existing distortions can be found in the labor market, depicted in Figure 8.3. Here, a pre-existing tax on wages causes the net, after-tax wage (W_0^n) to be lower than the gross, pre-tax wage (W^g) by the amount of the tax. With this tax distortion, the quantity of labor supplied is L_0 and there is a DWL. When a new regulation is imposed in another market, raising production costs, one of the indirect effects may be an increase in the price level as those costs are passed through the economy. This increase in the price level will reduce the real wage and, given an upward sloping labor supply curve, the amount of labor supplied.[10] This is shown in Figure 8.3 as a decrease in the net wage to W_1^n and a decrease in the amount of labor supplied to L_1.

The interaction between new and pre-existing distortions is especially pronounced in the labor market because pre-existing distortions there are large. As shown in Figure 8.3, even a small reduction in the amount of labor supplied will result in a large increase in DWL.[11] Similar interactions are likely to occur in other markets with pre-existing distortions. In cases where they are likely to have a significant impact, analysts

Figure 8.3 - Labor Market with Pre-Existing Distortions



should incorporate these distortions into models used to estimate social cost.[12]

In a general equilibrium analysis, the social cost of a regulation is estimated using a computable general equilibrium (CGE) model. CGE models simulate the workings of a market economy and can include representations of the distortions caused by taxes and regulations. As described above, they are used to calculate a set of price and quantity variables that will return the simulated economy to equilibrium after the imposition of a regulation. The social cost of the regulation can then be estimated by comparing the value of variables in the pre-regulation, "baseline" equilibrium with those in the post-regulation, simulated equilibrium.[13]

10 In general equilibrium analysis, all prices and wages are real, i.e., they are measured relative to a *numéraire*, a specific single price or weighted average of prices, such as the GDP deflator. Here, the consumer price level rises relative to the *numéraire*. The result is a fall in the real wage — the nominal wage divided by the consumer price level.

11 The labor tax distortion affects individual labor supply decisions at the margin. Thus, a full-time worker may not change (or be able to change) her hours worked in response to a fall in the real wage. However, part-time workers, workers in households with more than one full-time worker, or potential retirees, may be more likely to adjust the number of hours they work or whether they work at all. A discussion of the theoretical and empirical basis for this depiction of the labor market can be found in Parry (2003).

12 Economists have long recognized these interaction effects (Ballard and Fullerton 1992). A more recent body of work has focused on them in the context of environmental regulation. In this literature, these interactions are known as the "tax-interaction effect." If an environmental regulation raises revenue through a tax on pollution or other revenue raising provision, and the revenue is used to reduce pre-existing distortions such as taxes on wages, the tax-interaction effect may be offset. This is known as the "revenue recycling effect." The offset may be partial, complete, or in some cases, the overall efficiency of the tax system may actually be improved. The net result is an empirical matter, depending on the nature of the full set of interactions across the economy and how the revenue is raised. Some of the early papers in this literature include Bovenberg and de Moojii (1994), Parry (1995), and Bovenberg and Goulder (1996). Goulder (2000) provides an accessible summary of the early literature. More recent papers include Parry and Bento (2000); Murray, Keeler, and Thurman (2005); and Bento and Jacobsen (2007).

13 CGE models are discussed in more detail in the modeling section of this chapter. Applications of CGE models to the estimation of the social cost of environmental regulation include Hazilla and Kopp (1990) and Jorgenson and Wilcoxen (1990). A version of the Jorgenson and Wilcoxen model was used as part of EPA's retrospective study of the benefits and costs of the Clean Air Act for the period 1970 to 1990 (U.S. EPA 1997a).

Even in a general equilibrium analysis, analysts must take care in selecting an appropriate measure of social cost. Calculating social cost by adding together estimates of the costs in individual sectors can lead to double counting. For example, counting both the increased costs of production to firms resulting from a regulation and the attendant increases in prices paid by consumers for affected goods would mean counting the same costs twice, leading to an overestimate of social cost. Instead, focusing on measures of changes in final demand, so that intermediate goods are not counted, can avoid the double-counting problem.[14]

While it is theoretically possible to estimate social cost by adding up the net change in consumer and producer surplus in all affected markets, the measures most commonly used in practice are consumer's equivalent variation (EV) and compensating variation (CV). Both are monetary measures of the change in utility brought about by changes in prices and incomes resulting from the imposition of a regulation. As households are the ultimate beneficiaries of government and investment expenditures, the EV and CV measures focus on changes in consumer welfare, rather than on changes in total final demand.

8.1.3 Dynamics

In most cases, a regulation will continue to have economic impacts for a number of years after its initial implementation. If these intertemporal impacts are likely to be significant, they should be included in the estimation of social cost. For example, if a regulation requires firms in the electric utility sector to invest in pollution control equipment, they may not invest as much in electric generation capacity as they would have in the absence of the regulation. This may result in slower growth in electricity output and reduce the overall growth rate of the economy. In some cases, the effect of a regulation on long-term growth may be much more significant than the effect on the regulated sector alone.

14 Final demand consists of household purchases, investment, government spending, and net exports (exports minus imports).

When conducting a BCA in which the analyst expects intertemporal effects of a regulation to be confined to the regulated sector, it may be appropriate to simply apply partial equilibrium analysis to multiple periods. Relevant conditions, like expected changes in market demand and supply over time, should be taken into account in the analysis. The costs in individual years can then be discounted back to the initial year for consistency.

If the intertemporal effects of a regulation on non-regulated sectors are expected to be significant, analysts can estimate social cost using a dynamic CGE model. Dynamic CGE models can capture the effects of a regulation on affected sectors throughout the economy. They can also address the long-term impacts of changes in labor supply, savings, factor accumulation, and factor productivity on the process of economic growth.[15] In a dynamic CGE model social cost is estimated by comparing values in the simulated baseline (i.e., in the simulated trajectory of the economy without the regulation) with values from a simulation with the regulation in place.

8.1.4 Social Cost and Employment Effects

At times of recession, questions arise about whether jobs lost as a result of a regulation should be counted as an additional cost of the regulation. However, counting the number of jobs lost (or gained) as a result of a regulation generally has no meaning in the context of BCA as these are typically categorized as transitional job losses.[16] BCA requires monetized values of both the social benefits and costs associated with the regulation. The social cost of a regulation already includes the value

15 In addition to affecting the growth of the capital stock, an environmental regulation may also negatively affect the supply of labor through the interaction effects discussed above, thus increasing social cost. However, there may also be a positive effect on labor supply if improved environmental quality confers health benefits that make the work force more productive.

16 In very rare cases in which a regulation contributes additional job losses to a sector exhibiting structural unemployment, analysts should consider including job losses as a separate cost category. See Appendix C for more detail.

of lost output associated with the reallocation of resources (including labor) away from production of output and towards pollution abatement. This does not mean, of course, that specific individual workers are not harmed by a policy if they lose their jobs. EPA estimates the magnitude of such losses as part of an Economic Impact Analysis (EIA). See Chapter 9 for more details on this topic.

8.2 A Typology of Costs

The previous section defined social cost as the sum of the opportunity costs incurred as the result of the imposition of a regulation, and introduced the basic economic theory used in its estimation. Conceptually, social cost is the most comprehensive measure of cost, and is thus the appropriate measure to use in BCA. In addition to social cost, a number of other concepts of cost exist and are often used to describe the effects of a regulation. This section discusses these alternative concepts and introduces a number of additional terms. This section also provides a discussion of measures that define temporary costs or define how costs are distributed across different entities.

8.2.1 Alternative Concepts of Cost

Three alternative concepts of cost, each of which is composed of two components, are: explicit and implicit costs, direct and indirect costs, and private sector and public sector costs. Like social cost, all of these concepts are comprehensive in nature. An important distinction is that while social cost is a measure derived from economic theory, these three alternative concepts are in general only descriptive.[17]

Consideration of these alternative concepts can provide insights into the full range of the costs of a regulation. They may also be useful in determining the appropriate framework and modeling methodology for an analysis. Several executive and legislative mandates require that a number of different types of costs be included in a regulatory impact analysis (RIA).[18]

8.2.1.1 Explicit and Implicit Costs

The total costs of a regulation can include both explicit and implicit costs.[19] Explicit costs are those costs for which an explicit monetary payment is made, or for which it is straightforward to infer a value. For firms, the explicit costs of environmental regulation normally include the costs of purchase and operation of pollution control equipment. This includes payments for inputs (such as electricity) and wages for time spent on pollution control activities. For households, explicit costs may include the costs of periodic inspections of pollution control equipment on vehicles. For government regulatory agencies, wages paid to employees for developing a regulation and then for administration, monitoring, and enforcement are included in explicit costs. Implicit costs are costs for which monetary values do not readily exist and are thus likely more difficult to quantify. Implicit costs may include the value of current output lost because inputs are shifted to pollution control activities from other uses, as well as lost future output due to shifts in the composition of capital investment. Implicit costs may also include the lost value of product variety as a result of bans on certain goods, time costs of searching for substitutes, and reduced flexibility of response to changes in market conditions.

8.2.1.2 Direct and Indirect Costs

Direct costs are those costs that fall directly on regulated entities as the result of the imposition of a regulation. These entities may include firms, households, and government agencies. Indirect costs are the costs incurred in related markets or experienced by consumers or government agencies

17 In certain cases, a single component, such as direct cost, may provide a reasonable estimate of social cost.

18 EO 12866 specifies that an assessment of the costs of a regulation should include "any adverse effects on the efficient functioning of the economy and private sector (including productivity, employment, and competitiveness)" in addition to compliance costs. The UMRA of 1995 requires that cost estimates take into account both indirect and implicit costs on state and local governments.

19 The term "total cost" is used here when discussing alternative concepts of cost in order to reinforce the distinction between these concepts and social cost.

not under the direct scope of the regulation. These indirect costs are usually transmitted through changes in the prices of the goods or services produced in the regulated sector. Changes in these prices then ripple through the rest of the economy, causing prices in other sectors to rise or fall and ultimately affecting the incomes of consumers. Government entities can also incur indirect costs. For example, if the tax base changes due to the exit of firms from an industry, revenues from taxes or fees may decline. In some cases, the indirect costs of a regulation may be considerably greater than the direct costs.

8.2.1.3 Private Sector and Public Sector Costs

The total costs of a regulation can also be divided between private sector and public sector costs. Private sector costs include all of the costs of a regulation borne by households and firms. Public sector costs consist of the costs borne by various government entities.

8.2.2 Additional Cost Terminology

In addition to the conceptual categories and their components discussed above, a variety of other terms are often used in describing the costs of environmental regulation. A number of these terms are defined here. It should be noted that there are numerous overlaps between these concepts, and analysts must take care to avoid double counting.[20]

8.2.2.1 Incremental Costs

Incremental costs are the additional costs associated with a new environmental regulation or policy. Incremental costs are determined by subtracting the total costs of environmental regulations and policies already in place from the total costs after a new regulation or policy has been imposed.

20 References that provide definitions of cost terminology include U.S. CBO (1988), and Callan and Thomas (1999).

8.2.2.2 Compliance Costs

Compliance costs (also known as *abatement costs*) are the costs firms incur to reduce or prevent pollution to comply with a regulation. They are usually composed of two main components: capital costs and operating costs. Compliance costs can be further defined to include any or all of the following:

- Treatment/Capture — The cost of any method, technique, or process designed to remove pollutants, after their generation in the production process, from air emissions, water discharges, or solid waste.
- Recycling — The cost of postproduction on-site or off-site processing of waste for an alternative use.
- Disposal — The cost involving the final placement, destruction, or disposition of waste after pollution treatment/capture and/or recycling has occurred.
- Prevention — The cost of any method, technique, or process that reduces the amount of pollution generated during the production process.

8.2.2.3 Capital Costs

Capital costs include expenditures on installation or retrofit of structures or equipment with the primary purpose of treating, capturing, recycling, disposing, and/or preventing pollutants. These expenditures are sometimes referred to as "one-time costs" and include expenditures for equipment installation and startup. Once equipment is installed, capital costs generally do not change with the level of abatement and are thus functionally equivalent to "fixed costs." In BCA, capital costs are usually "annualized" over the period of the useful life of the equipment.

8.2.2.4 Operating and Maintenance Costs

Operating and maintenance costs are annual expenditures on salaries and wages, energy inputs, materials and supplies, purchased services, and maintenance of equipment associated with pollution abatement. In general, they are directly related to the level of abatement. Operating costs are functionally equivalent to "variable costs."

8.2.2.5 Industry Costs

Industry costs are the costs of a regulation to an industry, including the effects of actual or expected market reactions. They often differ from compliance costs because compliance costs do not normally account for market reactions. Market reactions may include plant closures, reduced industry output, or the passing on of some costs directly to consumers.

8.2.2.6 Transactions Costs

Transactions costs are those costs that are incurred in making an economic exchange beyond the cost of production of a good or service. They may include the costs of searching out a buyer or seller, bargaining, and enforcing contracts. Transactions costs may be important when setting up a new market, such as those markets designed to be used for market-based regulations.

8.2.2.7 Government Regulatory Costs

Government regulatory costs are those borne by various government entities in the course of researching, enacting, and enforcing a policy or regulation.[21]

8.2.3 Transitional and Distributional Costs

In addition to the concepts and terms defined above, several other types of cost exist. Two qualitatively different types of cost from those above are transitional and distributional costs.

8.2.3.1 Transitional Costs

At some point in time after the imposition of a new environmental regulation, the economy can be expected to adjust to a new equilibrium. While many costs are likely to be permanent additions to the costs of production, others will be short term in nature, being incurred only during the adjustment to the new equilibrium. These are known as transitional costs. Transitional costs may include the costs of training workers in the use of new pollution control equipment. After workers receive their initial training, the time they spend on pollution control activities would be counted as operating costs.

8.2.3.2 Distributional Costs

Distributional costs are those costs that relate to how certain entities or societal groups are impacted by the imposition of a policy or regulation. While BCA is by definition concerned only with the net benefits, it is likely that most policies or regulations will result in winners and losers. In some cases, the models described later in this chapter can be used for distributional analysis as well as BCA. Distributional costs are covered in detail in Chapter 10.

8.3 Measurement Issues in Estimating Social Cost

A number of issues may arise when estimating the expected social cost of a proposed regulation, or when measuring costs incurred as a result of an existing regulation. These issues can be divided into two broad categories: (1) those that arise when estimating costs over time; and (2) those associated with difficulties in developing numeric values for estimating social cost. This section discusses both these issues in turn. It concludes with a short analysis of how estimates of Title IV of the Clean Air Act's costs evolved over time, illustrating the importance of accurately accounting for these issues when estimating the costs of a regulation.

8.3.1 Evaluating Costs Over Time

Most regulations cause permanent changes in production and consumption activities, leading to permanent (ongoing) social costs. As a result, regulations are often phased in gradually over time in an effort to limit any disruptions created

21 Government entities may themselves be polluters and therefore subject to regulation. Compliance costs under this scenario would be captured as such.

by their imposition. When measuring costs over time, assumptions related to the time horizon of the analysis, the use of a static versus a dynamic framework, discounting, and technical change are extremely important. These assumptions are each discussed in more detail in the paragraphs that follow.

8.3.1.1 Time Horizon

Irrespective of the method used for the estimation of social cost, the time horizon for calculating producer and consumer adjustments to a new regulation should be considered carefully. Ideally, the analyst estimates the value of all future costs of a regulation discounted to its present value. If the analyst is only able to estimate a regulation's costs for one or a few representative future years, she must take great care to ensure that the year(s) selected are truly representative, that no important transitional costs are effectively dismissed by assumption, and that no one-time costs are assumed to be on-going.

In the short term, at least some factors of production are fixed. If costs are evaluated over a short period of time, then contractual or technological constraints prevent firms from responding quickly to increased compliance costs by adjusting their input mix or output decisions. In the long term, by contrast, all factors of production are variable. Firms can adjust any of their factors of production in response to changes in costs due to a new regulation. A longer time horizon affords greater opportunities for affected entities to change their production processes (for instance, to innovate). It is important to select a time horizon that captures any flexibility the regulation provides firms in the way they choose to comply.

8.3.1.2 Choosing Between a Static and Dynamic Framework

In many cases, costs are evaluated in a static framework. That is, costs are estimated at a given point in time or for a selection of distinct points in time. Such estimates provide snapshots of costs faced by firms, government, and households but do not allow for behavioral changes from one time period to affect responses in another time period.

In addition to the capital-induced growth effects discussed in Section 8.2.3, the evaluation of costs in a dynamic framework may be important when a proposed regulation is expected to affect product quality, productivity, innovation, and changes in markets indirectly affected by the environmental policy.[22] These may have impacts on net levels of measured consumer and producer surplus over time.

8.3.1.3 Discounting

Social discounting procedures for economic analyses are reviewed in considerable detail in Chapter 6. Benefits and costs that occur over time must be properly and consistently discounted if any comparisons between them are to be legitimate.[23]

There is one application of discounting that is unique to cost analysis. When calculating firms' private costs (e.g., the internal cost of capital used for pollution abatement), the analyst should use a discount rate that reflects the industry's cost of capital, just as a firm would. The social cost of the regulation, on the other hand, would be calculated using the social discount rate, the same discount rate used for the benefits of the regulation.

8.3.1.4 Technical Change and Learning

Estimating the costs of a given environmental regulation frequently entails estimating future technical change. Despite its importance as a determinant of economic welfare, the process of technical change is not well understood. Different approaches to environmental regulation present widely differing incentives for technological innovation. As a result, the same environmental end may be achieved at significantly different costs, depending on the pace and direction of technical change. Recent empirical work supports this hypothesis. Most notably, the realized costs of Title IV of the 1990 Clean Air Act Amendment's SO_2 Allowance Trading program are considerably lower than initial predictions, in part due to unanticipated technical change (see Text Box 8.1).

22 See Section 8.1.3 for a discussion of dynamics.

23 In a CEA, it is equally important to properly discount cost estimates of different regulatory approaches to facilitate valid comparisons.

Text Box 8.1 - The Sulfur Dioxide Cap-and-Trade Program — A Case Study[24]

Under Title IV of the 1990 Clean Air Act Amendments (CAAA), coal fired power plants are required to hold one sulfur dioxide (SO_2) allowance for each ton of SO_2 they emit during the year. Utilities are allowed to buy, sell and bank unused allowances to cover future SO_2 emissions (see Chapter 4 for additional detail). Title IV was subject to intensive ex ante and ex post analysis. The evolution of these analyses illustrates the importance of complete and thorough estimation of social costs and highlights the difference some of the issues discussed above (e.g., discounting or uncertainties) can make to actual cost estimates.

Estimates of Title IV's compliance costs have declined over time, particularly so once the program was launched and researchers were able to observe the behavior of electric utilities. Title IV proved less costly than originally estimated due to behavior responses, indirect effects, technological improvements, market structure, and prices that changed over time. Table 8.1 provides a comparison of some of the program's cost estimates over time. Rows that report ex ante estimates are shaded gray.

Table 8.1 - Estimates of Compliance Costs for the SO_2 Program*

Study	*Annual Costs (Billions)*	*Marginal Costs per ton SO_2*	*Average costs per ton of SO_2*
Carlson et al. (2000)	$1.1	$291	$174
Ellerman et al. (2000)	1.4	350	137
Burtraw et al. (1998)	0.9	n/a	239
Goulder et al. (1997)	1.09	n/a	n/a
White (1997)	n/a	436	n/a
ICF (1995)	2.3	532	252
White et al. (1995)	1.4-2.9	543	286-334
GAO (1994)	2.2-3.3	n/a	230-374
Van Horn Consulting et al. (1993)	2.4-3.3	520	314-405
ICF (1990)	2.3-5.9	579-760	348-499

*Based on Table 2-1, Burtraw and Palmer (2004); n/a — not reported.

Most of the early estimates of Title IV's compliance costs were based on engineering models, which do not fully capture the concepts of consumer and producer surplus. In addition, many of these studies relied on the data and methodologies used to evaluate traditional command-and-control environmental policies, adjusted to estimate the efficiency gains of a permit trading system. Later studies that included more extensive examinations of both the regulatory impacts as well as outside economic pressures on the industry came up with significantly smaller compliance cost estimates for the regulation.

Several developments occurred around the time of Title IV that helped reduce the program's ex post cost estimates. For example, reductions in the price of low-sulfur coal, along with technological improvements that lowered the cost of fuel switching, allowed utilities in the East to reduce compliance costs by using low-sulfur coal from the Powder River Basin in Wyoming (Carlson et al. 2000, and Burtraw and Palmer 2004). Furthermore Popp (2003) concluded that Title IV-induced R&D led to technological innovations that improved the efficiency of scrubbers, thereby leading to lower operating costs.

The varying cost estimates also show the importance of accounting for changing implementation costs and uncertainty over time. The ability of facilities to "bank" SO_2 allowances allowed flexibility in implementation and thus reduced compliance costs. Cost estimates by Carlson et al. (2000) and Ellerman et al. (2000) factor in the discounted savings from banking. According to the latter study, costs savings are a relatively minor source of overall savings, but are important in developing a picture of the program's total effectiveness. This is because firms were able to "avoid the much larger losses associated with meeting fixed targets in an uncertain world" (Ellerman et al. 2000, p. 285).

24 This example is taken from Burtraw and Palmer (2004).

Organizations are able to learn with experience, which permits them to produce a given good or service at lower cost as their cumulative experience increases. While there are many different explanations for this phenomenon (e.g., labor forces learn from mistakes and learn shortcuts; ad hoc processes become standardized) its existence has been borne out by experiences in many sectors. Indeed, OMB now requires cost analyses to consider possible learning effects among the cost-saving innovations.[25] Recent EPA Advisory Council guidance recommends that default learning effects be applied even when sector- or process-specific empirical data are not available (U.S. EPA 2007b).

The decrease in unit cost as the number of units produced increases is referred to as an experience or learning curve. A useful description of the calculations used to identify a learning curve can be found in van der Zwaan and Rabl (2004). Learning rates for 26 energy technologies are described in McDonald and Schrattenholzer (2001). Dutton and Thomas (1984) summarize more than 100 studies, including some dealing with the energy and manufacturing sectors. Note that the empirical estimates in the literature represent a biased sample, since they only represent technology that has been successfully deployed (Sagar and van der Zwaan 2006).[26]

8.3.2 Other Issues in Estimating Social Cost

Difficulties in measuring social cost generally fall into two categories: (1) difficulties in developing a numeric value for some social cost categories; and (2) for social cost categories where numeric values have been successfully developed, accounting for uncertainty in these values.

8.3.2.1 Difficulties in Developing Numeric Values

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 populations, and poorly-understood effects on large-scale ecosystems are difficult to capture in a quantitative BCA. Some alternative techniques for measuring and presenting these effects to policy makers are reviewed in Section 7.6.3. The relative significance of social cost categories that are not quantified — or are quantified but not valued — should be described in the social cost analysis.

8.3.2.2 Uncertainty

The values of various costs in the social cost analysis can be estimated, but cannot be known with certainty. In fact, some data and models will likely introduce substantial uncertainties into these estimates. Numerous assumptions are made regarding the baseline, predictions of responses to policy, and the number of affected markets. The conclusions drawn in the social cost analysis are sensitive to the degree of uncertainty regarding these assumptions. The uncertainty associated with the data and methods, the assumptions made, and how the uncertainty and assumptions affect the results are all-important components of the presentation of social cost, and should be carefully reported.

8.3.2.3 Estimating Costs Under Different Statutory Criteria

Some statutes require EPA to choose a regulatory option that is demonstrably affordable. One way for a decision maker to ensure that a regulatory option is affordable is to estimate an upper bound of the compliance cost associated with the chosen option and then to show that it is affordable. However, this approach is inconsistent with the practice of producing the best central estimate of the cost of a regulation for the RIA and will cause the net benefits of the regulation to be biased

25 OMB's *Circular A-4* asserts that a cost analysis should incorporate credible changes in technology over time, stating that "...retrospective studies may provide evidence that 'learning' will likely reduce the cost of regulation in future years" (OMB 2003). Other cost-saving innovations to consider include those resulting from a shift to regulatory performance standards and incentive-based policies.

26 Note that cost decreases associated with technological change and learning may not always be free but may have additional costs associated with them such as training costs. See Section 8.2.3.1 for a discussion of transitional costs.

Table 8.2 - Major Attributes of Models Used in the Estimation of Costs

	Compliance Cost	Partial Equilibrium	Linear Programming	Input-Output	Input-Output Econometric	CGE
Can be used to measure direct compliance costs	•		•			
Can be used to measure transitional costs	•	•	•	•	•	
Can be used to measure distributional impacts	•	•		•	•	•
Can capture indirect effects				•	•	• Ê
Can capture feedback and interaction effects						•

downward. Furthermore, using solely an upper bound estimate of the cost of a regulation could result in artificially low levels of regulation in situations where EPA must determine whether or not the benefits of the regulation justify the costs. It is thus very important that analysts rely on the best central estimate of the cost of a regulation for the RIA.

8.3.3 Use of Externally-Produced Cost Estimates

At various times EPA depends on externally (e.g., contractor, industry association, or advocacy group) generated cost estimates for use in its internal analyses. Any cost estimate produced by an external source and used by EPA in its internal analysis should be vetted by EPA to ensure that: (1) the information is relevant for its intended use; (2) the scientific and technical procedures, measures, methods and/or models employed to generate the information are reasonable for, and consistent with, the intended application; and (3) the data, assumptions, methods, quality assurance, sponsoring organizations, and analyses employed to generate the information are well documented.

8.4 Models Used in Estimating the Costs of Environmental Regulation

A number of different types of models have been used in the estimation the costs of environmental regulation. They range from models that estimate costs in a single industry (or part of an industry), to models that estimate costs for the entire economy. In practice, implementation of some of the models can be simple enough to be calculated in a spreadsheet. Others may be complex systems of thousands of equations that require highly specialized software.[27]

Table 8.2 summarizes some of the major attributes of the models discussed in this section. Each has strengths and weaknesses in analyzing different types of economic costs. When estimating social cost, there will be some cases where a single model is enough to provide a reasonable approximation. In other cases the use of more than one model is required. For example, a compliance cost model can be used to estimate the direct costs of a regulation in the affected sector. These direct cost estimates could then be used in a partial equilibrium model to estimate social cost. While most of the models discussed in this section can be used in some form in the estimation of social cost, many of them also have particular strengths in the estimation of transitional and/or distributional costs, as may be required as part of an RIA.

Selecting the most appropriate model (or models) to use in an analysis can be difficult. Below are a number of factors that may be helpful in making a choice.[28]

27 Data requirements for these models vary. Refer to Chapter 9 for a discussion of the process of conducting an industry profile and details on a range of public and private data sources that can be used for cost estimation.

28 This list of factors is derived from Industrial Economics, Inc. (2005). Proprietary models discussed in this section are examples only and no endorsement by EPA is given or implied.

- **Types of impacts being investigated.** Model selection should take into account the types of impacts that are important in the analysis being performed because models differ in their abilities to estimate different types of costs.
- **Geographic scope of expected impacts.** While some models may be well suited for the analysis of impacts on a national scale, it may not be possible to narrow their resolution to focus on regional or local impacts. Similarly, models that are well suited for examining regional or local impacts may not capture the full range of impacts at the national level.
- **Sectoral scope of expected impacts.** Some models are highly aggregated, and while proficient at capturing major impacts and interactions between sectors, are not well suited for focusing on a single or small number of specialized sectors. Likewise, models that are highly specialized for capturing impacts in a particular sector will usually be inappropriate for examining impacts on a broader set of sectors.
- **Expected magnitude of impacts.** A model that is well suited for capturing the impacts of a regulation that is expected to have large effects may have difficulty estimating the impacts of a regulation with relatively smaller expected effects, and vice versa.
- **Expected importance of indirect effects.** For a regulation that is expected to have substantial indirect effects beyond the regulated sector it is important to choose a model that can capture those effects.

Usually, some combination of the above factors will determine the most appropriate model for a particular application. Finally, it should be noted that advances in computing power, data availability, and more user-friendly software packages continually reduce the barriers to sophisticated model-based analysis.

8.4.1 Compliance Cost Models

Compliance cost models are used to estimate an industry's direct costs of compliance with a regulation. Estimates by engineers and other experts are used to produce algorithms that characterize the changes in costs resulting from the adoption of various compliance options. The particular parameters are usually determined for a number of individual plants with varying baseline characteristics. To estimate the control costs of a regulation for an entire industry, disaggregated data that reflects the industry's heterogeneity is input into the model. The disaggregated cost estimates are then aggregated to the industry level.

Compliance cost models may include capital costs, operating and maintenance expenditures, and costs of administration. Some compliance cost models are designed to allow the integrated estimation of control costs for multiple pollutants and multiple regulations. Some models are able to account for cost changes over time, including technical change and learning. Compliance cost models often are implemented in a spreadsheet; in general, they are relatively easy to modify and interpret.

While precise estimates of compliance costs are an important component of any analysis, it is only in cases where the regulation is not expected to significantly impact the behavior of producers and consumers that compliance costs can be considered a reasonable approximation of social cost. As discussed in Section 8.2.1, estimating social cost often requires knowledge of both supply and demand conditions. Compliance cost models focus on the supply side, and in circumstances where producer and consumer behavior is appreciably affected, these models are not able to provide estimates of changes in industry prices and output resulting from the imposition of a regulation. However, in these cases, estimates from compliance cost models can be used as inputs to other models that estimate social cost.

One example of a compliance cost model or tool is AirControlNET (ACN). ACN is a database tool for conducting pollutant emissions control strategy and costing analysis. It overlays a detailed control measure database of EPA emissions inventories to compute source- and pollutant-specific emission reductions and associated costs at various geographic levels (national, regional, local) and for many industries.

ACN contains a database of control measures and cost information that can be used to assess the impact of strategies to reduce criteria pollutants [e.g., NO_X, SO_2, volatile organic compounds (VOCs), PM_{10}, $PM_{2.5}$, or Ammonia (NH_3)] as well as carbon monoxide (CO) and mercury (Hg) from point (utility and non-utility), area, nonroad, and mobile sources as provided in EPA's National Emission Inventory (NEI). ACN is strictly a compliance cost model, because it does not account for changes in the behavior of consumers and producers.

Advantages:

- Compliance cost models often contain significant industry detail and provide relatively precise estimates of the direct costs of a regulation. This is particularly true for regulations with minor cost impacts.
- Once constructed, compliance cost models require a minimum of resources to implement and are relatively straightforward to use and easy to interpret.

Limitations:

- As they are focused exclusively on the supply side, compliance cost models can only provide estimates of social cost in certain limited cases.
- Compliance cost models are usually limited to estimating costs for a single industry.

8.4.2 Partial Equilibrium Models

While compliance cost models may provide reasonable estimates of the compliance costs of a regulation, they do not incorporate the likely behavioral responses of producers and consumers. As shown in Section 8.2.1, if these responses are not taken into account, estimates of social cost are likely to be inaccurate. In cases where the effects of a regulation are confined to a single market, partial equilibrium models, which incorporate the behavioral responses of producers and consumers, can be used to estimate social cost.

Inputs into an analysis employing a partial equilibrium model may include regulatory costs estimated using a compliance cost model and the supply and demand elasticities for the affected market. The model then can be used to estimate the change in market price and output. Changes in producer and consumer surplus reflect the social cost of the regulation. The relative changes between producer and consumer surplus provide an estimate of the distribution of regulatory costs between producers and consumers.

In a partial equilibrium model, the magnitude of the impacts of a regulation on the price and quantity in the affected market depends on the shapes of the supply and demand curves. The shapes of these curves reflect the underlying elasticities of supply and demand. These elasticities can be either estimated from industry and consumer data or taken from previous studies.[29]

If the elasticities used in an analysis are drawn from previous studies, they should be consistent with the following conditions:

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

In some cases, if the effects of a regulation are expected to spill over into adjoining markets (e.g., suppliers of major inputs or consumers of major outputs), partial equilibrium analysis can be extended into these additional markets as well. These "multi-market models" have been used in the analysis of a number of EPA regulations.[30]

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29 Because of the widespread use of elasticity estimates, the Air Benefit and Cost (ABC) Group in EPA's Office of Air and Radiation maintains an elasticity database. This Elasticity Databank serves as a searchable database of elasticity parameters across economic sectors/product markets and a variety of types including demand and supply elasticities, substitution elasticities, income elasticities, and trade elasticities. An online submittal form allows users to provide elasticity estimates for consideration as part of this databank. The Elasticity Databank is available online at http://www.epa.gov/ttn/ecas/Elasticity.htm (U.S. EPA 2007d).

30 See, for example, U.S. EPA (1989) *Regulatory Impact Analysis of Controls on Asbestos and Asbestos Products: Final Report.*

Advantages:

- Because they usually simulate only a single market, partial equilibrium models generally have fairly limited data requirements and are relatively simple to construct.
- Partial equilibrium models are comparatively easy to use and interpret.

Limitations:

- Partial equilibrium models are limited to cost estimation in a single or small number of markets and do not capture indirect or feedback effects.
- Because partial equilibrium models are generally data driven and specific to a particular application, they are usually not available "off-the-shelf" for use in a variety of analyses.

8.4.3 Linear Programming Models

Although linear programming models can be employed in a variety of applications, their use in the analysis of EPA regulations occurs most frequently in the estimation of compliance costs.[31] Linear programming models minimize (or maximize) an objective function by choosing a set of decision variables, subject to a set of constraints. In EPA's regulatory context, the objective function is usually direct compliance costs, which are minimized. The decision variables represent the choices available to the regulated entities. The constraints may include available technologies, productive capacities, fuel supplies, and regulations on emissions.

Although linear programming models can be constructed to examine multiple sectors or economy-wide effects, they are more commonly focused on a single sector. For the regulated sector, a linear programming model can incorporate a large number of technologies and compliance options, such as end-of-pipe controls, fuel switching, and changes in plant operations. Similarly, the model's constraints can include multiple regulations that require simultaneous compliance. The objective function usually includes the fixed and variable costs of each compliance option. The program then chooses a set of decision variables that minimize the total costs of compliance. In addition to compliance costs, the outputs from the model may include other related variables, such as projected fuel use, output and input prices, emissions, and demand for new capacity in the regulated industry.

An example of a linear programming model used by EPA is the Integrated Planning Model (IPM). The IPM is a model of the electric power sector in the 48 contiguous states and the District of Columbia. It can provide long-term (10-20 year) estimates of the control costs of complying with proposed regulations, while meeting the projected demand for electricity. In the model, nearly 13,000 existing and planned electrical generating units are mapped to approximately 1,700 representative plants. Results are differentiated into 40 distinct demand and supply regions. IPM can be used to estimate the impacts on costs for policies to limit emissions of SO_2, NO_x, CO_2, and Hg.

Advantages:

- Compared to compliance cost models, linear programming models are better able to incorporate and systematically analyze a wide range of technologies and multiple compliance options.
- Linear programming models allow for a considerable amount of flexibility in the specification of constraints. This permits an existing model to be used in a range of applications.

Limitations:

- Linear programming models normally do not estimate costs beyond a single sector and are thus unable to estimate indirect or distributional costs.

31 An introduction to linear programming is provided in Chiang (1984). The "linear" in the name refers to the linear specification of the objective function and constraint equations. Similar, eponymous model types include non-linear, integer, and mixed integer programming models.

Table 8.3 - Input-Output Table for the United States, 1999 (bil. $)

	1 Agriculture	2 Manufacturing	3 Services	**Total Intermediate Outputs**	Final Demand	**Total Outputs**
1 Agriculture	70	150	30	**250**	30	**280**
2 Manufacturing	50	1,930	840	**2,820**	2,470	**5,290**
3 Services	60	1,070	2,810	**3,940**	6,780	**10,720**
Total Intermediate Inputs	**180**	**3,150**	**3,680**	**7,010**	**9,280**	**16,290**
Value Added	100	2,140	7,040	**9,280**		
Total Inputs	**280**	**5,290**	**10,720**	**16,290**		

Source: Adapted from Bureau of Economic Analysis (BEA) 10-sector table.

- A linear programming model designed for estimating sectoral compliance costs will likely be quite complex and have heavy input requirements. If an existing model is not available, the time and effort to construct one may be prohibitive.
- Linear programming models minimize aggregate control costs for the entire industry simultaneously, whereas the regulated entities actually do so individually. This may result in an underestimation of total compliance costs.

8.4.4 Input-Output Models

While input-output models have been used in many environmental applications, their primary use in a regulatory context is for estimating the distributional and short-term transitional impacts that may result from the implementation of a policy. For example, an input-output model could be used to estimate the regional economic effects of a regulation that would ban a particular pesticide. In this case, an input-output model could provide estimates of the effects on output and employment in the affected region. A key feature of input-output models is their ability to capture both the effects on sectors directly affected by a regulation and the indirect effects that occur through spillovers onto other sectors.[32]

An input-output model is based on an input-output table. The input-output table assembles data in a tabular format that describes the interrelated flows of goods and factors of production over the course of a year. An input-output table may consist of hundreds of sectors or may be aggregated into as few as two or three sectors. Table 8.3 is an example of a highly aggregated input-output table for the United States for the year 1999. The columns for the individual sectors denote how much of each commodity is used in the production of that sector's output. These intermediate inputs are combined with factors of production — labor, capital, and land — whose payments as wages, profits, and rents, compose sectoral value added. For the agricultural sector, total inputs consist of $70 billion of agricultural inputs, $50 billion of manufactured inputs, $60 billion of service inputs, and $100 billion of value added, for a total of $280 billion in inputs. The row for each sector shows how that sector's output is consumed. In the case of the agricultural sector, $250 billion is consumed as intermediate inputs, while the remainder, $30 billion, is consumed as final demand, which is composed of household consumption, government purchases, and investment.

An input-output table can be turned into a simple linear model through a series of matrix operations. The model relates changes in final demand to changes in the total amount of goods and services, including intermediate inputs, required to meet that demand. The model can also relate the change in final demand to changes in employment of factors of production, such as the demand for labor. In the case of the banned pesticide, if a separate analysis determines that there will be a

32 Miller and Blair (1985) is a standard reference on input-output analysis.

decline in the output of cotton, the input-output model could be used to determine the effect on those sectors that supply inputs to the cotton sector, as well as on industries that are users of cotton, such as the producers of textiles and clothing. Declines in the output of these industries will have further effects on the demand for other intermediate inputs, like electricity, which are also estimated by the model.

Input-output models are relatively simple to use and interpret and are often the most accessible tool for analyzing the short-term impacts of a regulation on regional output and income.[33] However, they embody a number of assumptions that make them inappropriate for long-term analysis or the analysis of social cost. Although their specifications can sometimes be partially relaxed, input-output models embody the assumptions of fixed prices and technology, which do not allow for the substitution that normally occurs when goods become more or less scarce. Similarly, input-output models are demand driven and not constrained by limits on supply, which would normally be transmitted through increases in prices. While the rigidities in the models may be reasonable assumptions in the short run or for regional analysis, they limit the applicability of input-output models for long-run or national issues. Because input-output models do not include flexible supply-demand relationships or the ability to estimate changes in producer and consumer surpluses, they are not appropriate for estimating social cost.

Advantages:

- Particularly in a regional context, input-output models are often well suited for estimating distributional and short-term transitional impacts.
- Input-output models are relatively transparent and easy to interpret.
- Some input-output models have a great deal of sectoral and regional disaggregation and can be readily applied to issues that require a high degree of resolution.

Limitations:

- Input-output models are not appropriate for estimating social cost.
- Because of their lack of endogenous substitution possibilities in production, input-output models are not appropriate for dealing with long-run issues.
- Because of their fixed prices and lack of realistic behavioral reactions by producers and consumers, input-output models are not well suited for dealing with issues that are likely to have large effects on prices.

8.4.5 Input-Output Econometric Models

Input-output econometric models are economy-wide models that integrate the structural detail of conventional input-output models with the forecasting properties of econometrically estimated macroeconomic models. Input-output econometric models are often constructed with a considerable amount of regional detail, including the disaggregation of regional economies at the state and county level. At EPA, input-output econometric models, like conventional input-output models, are often used to examine the regional impacts of policies and regulations. However, unlike conventional input-output models, input-output econometric models are also able to estimate long-run impacts.

When used for policy simulations, a major limitation of conventional input-output models is that the policy under consideration must be translated into changes in final demand. Furthermore, because they do not include resource constraints, the resulting solution may not be consistent with the actual supply-demand conditions in the economy. Input-output econometric models, in contrast, are driven by econometrically estimated macroeconomic

33 An off-the-shelf input-output model often used in the analysis of the impacts of environmental regulation is Impact Analysis for Planning (IMPLAN). IMPLAN is based on data for the United States that covers more than 500 sectors and can be disaggregated down to the county level.

relationships that more accurately account for these conditions. However, unlike standard macro-econometric models, input-output econometric models integrate input-output data and structure into the specification of production. This allows them to estimate changes in the demand for and the production of intermediate goods. The macroeconomic component enables the models to be used for long-run forecasting, including accounting for business cycles and involuntary unemployment. This makes input-output econometric models particularly useful for estimating transitional costs arising from the implementation of a regulation.

An example of an input-output econometric model that has been used for policy analysis at EPA is the Regional Economic Models, Inc. (REMI) Policy Insight. The standard REMI model includes 70 production sectors and 25 final demand sectors and can provide output on changes in income and consumption for more than 800 separate demographic groups. The model is both national in scope and can be specially tailored to individual regions. The REMI model has been applied to a wide range of regional environmental policy issues, including extensive analysis of air quality regulation in the greater Los Angeles area.

Advantages:

- Input-output econometric models can be used to estimate both long- and short-run transitional costs.
- Input-output econometric models can be used to estimate distributional costs.

Limitations:

- Because input-output econometric models combine elements of both macro and micro theory, it may not be easy to disentangle the mechanisms actually driving model results.
- Compared to standard input-output models, input-output econometric models may not have the sectoral resolution necessary to analyze the impact of a policy expected to have limited impacts.

8.4.6 Computable General Equilibrium Models

CGE models have been used in a number of applications in the analysis of environmental regulation. Examples include estimation of the costs of the Clean Air Act (CAA), the impacts of domestic and international policies for GHG abatement, and the potential for market-based mechanisms to reduce the costs of regulation.

CGE models simulate the workings of the price system in a market economy. Markets exist for commodities and can also be specified for the factors of production: labor, capital, and land. In each market, a price adjusts to equilibrate supply and demand. A CGE model may contain several hundred sectors or only a few, and may include a single "representative" consumer or multiple household types. It may focus on a single economy with a simple representation of foreign trade, or contain multiple countries and regions linked through an elaborate specification of global trade and investment. The behavioral equations that govern the model allow producers to substitute among inputs and consumers to substitute among final goods as the prices of commodities and factors shift. The behavioral parameters can be econometrically estimated, calibrated, or drawn from the literature. In some models, agents may be able to make intertemporal trade-offs in their consumption and investment choices.

Simulating the effects of a policy change involves "shocking" the model, by, for example, introducing a regulation, such as a tax on emissions. Prices in affected markets will then move up or down until a new equilibrium is established. Prices and quantities in this new equilibrium can be compared to those in the initial equilibrium. A static CGE model will be able to describe changes in economic welfare measures due to a reallocation of resources across economic sectors following a policy shock. In a policy simulation using a dynamic CGE model, a time path of new prices and quantities is generated. This time path can be compared to a baseline path of prices and quantities that is estimated by running the model without the policy shock. As some policies can be expected to have impacts over a

Text Box 8.2 - The Pollution Abatement Costs and Expenditures Survey

The Pollution Abatement Costs and Expenditures (PACE) survey is the primary source of information on pollution abatement-related operating costs and capital expenditures for the U.S. manufacturing sector (U.S. Bureau of the Census, various years). The PACE survey collects data on costs of pollution treatment (i.e., end-of-pipe controls), pollution prevention (i.e., production process enhancements to prevent pollution from being produced), disposal, and recycling. The survey is sent to approximately 20,000 establishments (who are required by law to respond to it) and was conducted annually by the U.S. Census Bureau from 1973 to 1994 (except in 1987) and then again in 1999.

EPA funded the 1999 PACE survey. However, this survey was substantially different from its predecessors, making direct longitudinal analysis difficult (see Becker and Shadbegian 2005 for a comprehensive description of the conceptual differences between the 1994 and 1999 PACE surveys). More recently, with the guidance and financial support of EPA, a completely revised version of the PACE survey was administered by the Census Bureau to collect 2005 data. The 2005 PACE survey was the result of a multi-year effort to evaluate the quality of the survey instrument and the accuracy and reliability of the responses to the survey. The 2005 PACE data, which was released in April 2008, is longitudinally consistent with previous PACE surveys, with the exception of the 1999 iteration. EPA has no current plan to collect PACE data beyond 2005, but hopes to reinstate the survey in the future to once again collect data on an annual basis. The annual collection of pollution abatement costs would provide EPA with information required for its RIAs, and would better enable researchers to answer questions of interest, particularly those that require longitudinal data.

The PACE survey contains operating costs and capital expenditures disaggregated by media: air, water, and solid waste; and by abatement activity: pollution treatment, recycling, disposal, and pollution prevention. Total operating costs are further disaggregated into: salary and wages, energy costs, materials and supplies, contract work, and depreciation.

The PACE survey data, both aggregate and establishment-level, have been used to analyze a wide range of policy questions. These include assessing the impact of pollution abatement expenditures on productivity growth, investment, labor demand, environmental performance, plant location decisions, and international competitiveness.

longer time horizon, dynamic models are used to capture, in addition to static impacts, the welfare consequences of reallocating resources over time, such as the impact that changes in savings may have on capital accumulation. Forward-looking models can also capture the effects that future policies may have on current decisions.

An example of the use of a CGE model at EPA is the retrospective BCA of the CAA, which used a dynamic CGE model to compute the costs of CAA compliance over the period 1970 to 1990 (U.S. EPA 1997a). Estimates of pollution abatement expenditures for the U.S. manufacturing sector were first calculated using Pollution Abatement Costs and Expenditures (PACE) survey data (see Text Box 8.2). As the analysis was retrospective, the relevant policy simulations involved *removing* the long-term capital and operating costs from the industries that incurred them. The retrospective BCA compared the simulated path of the economy without these abatement expenditures and the actual path of the economy, which included them. EPA computed changes in both long-run GDP and equivalent variation, as well as impacts on investment, household consumption, and sectoral prices, output, and employment.

CGE models have also been used extensively in estimating the costs of GHG mitigation. Here, the analyses have been prospective, such as efforts to estimate the costs of complying with the Kyoto Protocol and more recently, proposed climate change legislation. Some studies have focused on the control of CO_2 emissions by introducing carbon taxes or emissions trading. Other studies have expanded the analysis by examining other GHGs and incorporating the effects of changes in land use patterns and carbon sinks. Of particular concern has been the problem of "leakage," in which a fall in emissions in participating countries is offset by an increase in emissions in non-participating countries, induced by the fall in demand, and thus the world price, of energy inputs.

CGE models can be useful tools for examining the medium- to long-term impacts of policies that are expected to have relatively large, economy-wide effects. A growing use of these models has been to quantify previously unrecognized welfare costs that can occur when environmental policies interact with pre-existing distortions in the economy. An expanding body of work has begun to include non-market goods into CGE models (Smith et al. 2004, and Carbone and Smith 2008).

Given the large number of parameters in a typical CGE model, analysts should take great care in ensuring the accuracy of a model's data and specifications. Sensitivity analysis should be performed on critical parameters. One strategy, currently used in EPA's analyses of climate legislation, is to use two CGE models concurrently to analyze the same policy scenarios.

Advantages:

- CGE models are best suited for estimating the cost of policies that will have large economy-wide impacts, especially when indirect and interaction effects are expected to be significant.
- CGE models are generally most appropriate for analyzing the medium- or long-term effects of policies or regulations.
- With the appropriate specifications incorporated, CGE models can be used to estimate the distributional impacts of policy shocks on household groups or industrial sectors.

Limitations:

- Because of their equilibrium assumptions, CGE models are generally not appropriate for analyzing short-run transitional costs. However, when appropriate specifications are included in a model, they may be used in this type of analysis.
- CGE models are generally not well suited for estimating the effects of policies that will affect only small sectors or will impact a limited geographic area. Although the costs have been reduced in recent years, the effort and data required to construct a new CGE model or revise an existing one may be prohibitive for some analyses.

Chapter 9

Employment Impacts Update

EPA is currently revising its guidance for assessing the employment impacts of environmental regulation. Section 9.2.3.3 "Impacts on employment" will be replaced with a discussion based on more recent literature and feedback from the Economy Wide Modeling Science Advisory Board Panel.[i] The new section will summarize the theory and methods for assessing employment impacts. Please note that subsequent to publication of the current Section 9.2.3.3, researchers attempted to replicate and extend the empirical estimates in Morgenstern, et al. (2002).[ii] However, as Belova, et al. (2013) note, "the original datasets and data management code used by MPS [Morgenstern, et al. (2002)] in the Census Research Data Center were not available to us because of the failure of the backup drive at the Census on which they had been archived." In light of this loss, replication attempts were not successful (Belova et al. 2013, 2015).[iii] In preparing economic analyses, analysts should not rely on the empirical estimates from Morgenstern, et al. (2002). Likewise, analysts should not rely on the estimates from Belova et al. (2013, 2015) as the authors "recommend that EPA refrain from using these results until the underlying cause(s) for the implausibly large estimates in the employment effects found in Belova et al. (2013a) are uncovered and resolved."[iv]

While EPA is awaiting the Science Advisory Board Panel report and continuing to explore recent areas of the literature, analysts are encouraged to look at recent EPA Regulatory Impact Analyses (RIAs) for best available methods and approaches for conducting employment impact analyses. Recent RIAs include those for the final Clean Power Plan published in August 2015,[v] the Residential Wood Heater New Source Performance Standard in February 2015,[vi] and the final Tier 3 Vehicle Emission and Fuel Standards Program in March 2014.[vii] These employment impact analyses contain an updated description of theoretic models and empirical methods that are more reflective of what will be incorporated into the employment impacts update to the *Guidelines*. Please contact EPA's National Center for Environmental Economics with any questions.

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[i] For more information please see http://yosemite.epa.gov/sab/sabproduct.nsf//LookupWebProjectsCurrentBOARD/07E67CF77B54734285257BB0004F87ED?OpenDocument

[ii] Morgenstern, R.D., W.A. Pizer, and J. Shih. 2002. Jobs Versus the Environment: An Industry Level Perspective. *Journal of Environmental Economics and Management* 43: 412-436.

[iii] Belova, A., W.B. Gray, J. Linn, and R.D. Morgenstern. 2013. Environmental Regulation and Industry Employment: A Reassessment. Discussion Papers, U.S. Census Bureau, Center for Economic Studies 2K132B, 4600 Silver Hill Road, Washington, DC 20233.

Belova, A., W.B. Gray, J. Linn, R.D. Morgenstern, and W. Pizer. 2015. Estimating the Job Impacts of Environmental Regulation. *Journal of Benefit-Cost Analysis*, 6(2), pp 325 – 340.

[iv] Quote is from Belova et al. (2015). Note that Belova et al. (2013a) in the quote is identical with Belova et al. (2013) cited above.

[v] See Chapter 6 of the RIA (EPA–HQ–OAR–2013–0602 at https://www.epa.gov/cleanpowerplan/clean-power-plan-final-rule-regulatory-impact-analysis).

[vi] See Chapter 5, Section 5.7 of the RIA (EPA-452/R-15-001 at https://www.epa.gov/sites/production/files/2015-02/documents/20150204-residential-wood-heaters-ria.pdf).

[vii] See Chapter 9 of the RIA (EPA-420-R-14-005 at https://www3.epa.gov/otaq/documents/tier3/420r14005.pdf).

Chapter 9
Economic Impact Analysis

The detailed study of regulatory consequences allows policy makers to fully understand a regulation's impacts, and to make an informed decision on its appropriateness. Economic information is necessary for the evaluation of at least two types of consequences of a regulatory policy: the regulation's efficiency, and its distributional effects. In principle, both could be estimated simultaneously using a general equilibrium model. In practice however, they are usually estimated separately.

The distributional effects of environmental regulations can be examined through an economic impact analysis (EIA). A related analysis, called an equity assessment, addresses the distribution of impacts across individuals and households, with particular attention to economically or historically disadvantaged or vulnerable groups (e.g., low-income households, racial or ethnic minorities, and young children). Equity assessments are sometimes referred to as environmental justice (EJ) analyses and are the subject of Chapter 10.

An EIA identifies the specific entities that benefit from or are harmed by a policy, and then estimates the magnitude of their gains and losses including changes in profitability, employment, prices, government revenues or expenditures, and trade balances. These estimates are derived from a study of the economic changes that occur across broadly-defined economic sectors of society, including industry, government, and not-for-profit organizations, but may also include more narrowly defined sectors within these broad categories, such as the solid waste industry or even an individual solid waste company. EIAs can measure a broad variety of impacts, such as direct impacts on individual plants, whole firms, and industrial sectors, as well as indirect impacts on consumers and suppliers.

9.1 Statutes and Policies

The following major statutes and EOs, all described in Chapter 2, directly address impact analyses:[1]

- Regulatory Flexibility Act of 1980 (RFA), as amended by the Small Business Regulatory Enforcement Fairness Act of 1996 (SBREFA);
- Unfunded Mandates Reform Act of 1995 (UMRA);
- EO 13132, "Federalism";
- EO 13175, "Consultation and Coordination with Indian Tribal Governments;" and
- EO 13211, "Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution, or Use."

Together with OMB's *Circular A-4*, they raise important dimensions relevant for economic impact analyses as summarized in Table 9.1.

1 EPA's Regulatory Management Division's Action Development Process (ADP) Library (http://intranet.epa.gov/adplibrary) is a resource for those who wish to access relevant statutes, EOs, or Agency policy and guidance documents in their entirety.

Table 9.1 - Potentially Relevant Dimensions to Economic Impact Analyses[2]

Dimension	Statute, Order, or Directive	Entity	Subpopulation
Sector	UMRA; EO 13132; OMB *Circular A-4*	Industry or government	Industries or state, local, or tribal governments
Entity size	RFA; UMRA; OMB *Circular A-4*	Businesses, governments, or not-for-profit organizations	Small businesses, small governmental jurisdictions, or small not-for-profit organizations
Time	OMB *Circular A-4*	Individuals or households	Current or future generations
Geography	OMB *Circular A-4*; UMRA	Region	Regions, states, counties, or non-attainment areas
Energy	EO 13211	Entities that use, distribute, or generate energy	Energy sector

The term "affected" is used throughout this chapter as a general term. Analysts should be aware that the authorizing statute for the rule, as well as other applicable statutes and administrative orders noted in this chapter, may 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, meaning that the analysis considers only those entities that are directly regulated by the rule. On the other hand, provisions in the UMRA and EO 12866 address both direct and indirect impacts, and therefore define the affected population more broadly. Care should be taken to avoid double counting when estimating direct and indirect impacts.

9.2 Conducting an Economic Impact Analysis

There are three important distinctions between BCA and EIA to keep in mind when conducting an EIA.[3] First, total social benefits and total social costs are not of primary importance in an EIA, as they are in a BCA. Rather, the main focus is on the components and distribution of the total social benefits and costs.

Second, transfers of economic welfare from one group to another are no longer assumed to cancel each other out, as they do in a BCA. Taxpayers, consumers, producers, governments, and the many sub-categories of these groups are all considered separately. While a BCA relies on estimates of the social benefits and costs of a regulation, an EIA focuses on the private benefits and costs associated with compliance responses. The EIA should use the same "starting point" as the BCA (i.e., same engineering or direct compliance costs, same benefit categories, etc.) for developing private benefit and cost estimates. In addition, some adjustments to these costs may be needed, as discussed below. For example, the tax status of a required piece of equipment is considered in private costs, but not in social costs.

Finally, there is a greater need for disaggregation in EIAs than in BCAs. Results may be presented for specific counties or other geographic units or types of entities, as appropriate, placing heavy demands on the modeling framework.

For any regulation, it is essential to ensure consistency between the EIA and the benefit-cost analysis (BCA). If a BCA is conducted, the corresponding EIA must be conducted within the same set of analytical assumptions. To the extent possible, adjustments to these assumptions or to the overall modeling framework used for the BCA should only be made when absolutely necessary, and then should be noted clearly in the text of the analysis.

2 Some environmental statutes may also identify subpopulations that merit additional consideration. This document is limited to those statutes with broad coverage.

3 Traditionally, EIAs focus on the costs of a particular rule or regulation. However, it is also possible to focus on the distribution of benefits or to calculate the net benefits for particular entities.

9.2.1 Screening for Potentially Significant Impacts

A comprehensive analysis of all aspects of all economic impacts associated with a rule can require significant time and resources, and its accuracy and thoroughness depend on the quality and quantity of available data. Thus, screening analyses are often employed to determine data availability, the severity of a rule's anticipated impacts, and the potential consequences of further analysis if undertaking it would require a delay in the regulatory schedule. A screening analysis can be thought of as a "mini-EIA" consisting of a rough examination of the data to identify sectors that may warrant further analysis.[4] Screening is effective for identifying the magnitude of the overall level of impacts on the regulated industry, but may fail to identify potentially large impacts on a single sector, region, or facility.

There are no established definitions for what constitutes a large or a small impact. However, a screening analysis is a tiered approach that initially captures most of the possible impacts (i.e., allows for many false positives) followed by a more detailed analysis that can help eliminate unfounded impacts. In this way, the screening analysis will eventually balance the risk of identifying "false positives" and "false negatives."

9.2.2 Profile of Affected Entities

Analysts should consider changes imposed by the rule in the regulated industry, as well as how related industries may be affected. Some industries may benefit from the regulation, while others may be subject to significant costs. If the regulation causes a firm to use different inputs or new technologies, then the producers of the new inputs will gain, while the producers of the old inputs will suffer. Developing a detailed industry profile will identify those industries that may be affected positively and negatively by the regulation.

4 The screening analysis discussed in this section is distinct from the screening analysis required to comply with the Regulatory Flexibility Act (as referred to in Section 9.3).

9.2.2.1 Compiling an Industry Profile and Projected Baseline

To determine the impacts of a particular regulation the analyst must understand the underlying structure of the affected industry and its various linkages throughout the economy.[5] This includes an understanding of the condition of the industry in terms of its finances and structure in the absence of the rule —the baseline of the EIA. A rule may impose different requirements and costs on new versus existing entities. Such rules may affect industry competition, growth, and innovation by raising barriers to new entry or encouraging continued use of outdated technology. Thus, a substantial portion of an EIA involves characterizing the state of the affected firms and industries in the absence of the rule as a basis for evaluating economic impacts.

The following are important inputs to defining an industry profile:

- **North American Industrial Classification System (NAICS) industry codes.** NAICS has replaced the U.S. Standard Industrial Classification (SIC) system in the U.S. Department of Commerce (DOC) Economic Census and other official U.S. Government statistics. NAICS was developed to provide comparable statistics about business activity across North America. It identifies hundreds of new, emerging, and advanced technology industries and reorganizes existing industries into more meaningful sectors, particularly in the service sector.[6]
- **Industry summary statistics.** Summary statistics of total employment, revenue, number of establishments, number of firms, and size of firms are available from U.S. DOC Economic Census or the Small Business Administration.[7]

5 Generally, analysts should initially assume a perfectly competitive market structure. One of the primary purposes of developing an industry profile is to confirm this assumption or discover evidence to the contrary.

6 For more information see www.census.gov/epcd/www/naics.html, which includes a NAICS/SIC correspondence (accessed on January 21, 2011).

7 See www.sba.gov/advocacy/849 for more information (accessed on January 21, 2011).

- **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. The number and size distribution of firms/facilities and the degree of vertical integration within the industry are important aspects of industry structure that affect the economic impact of regulations.
- **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 or take advantage of potential benefits without experiencing financial distress. Industries that are enjoying strong growth may be better able to recover increased costs through price increases than they would if there were no demand growth. Section 9.3.3.3 provides suggestions for using financial ratios to assess the significance of economic impacts on a firm's financial condition.
- **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. See Appendix A for a discussion of elasticities.

9.2.2.2 Profile of Government Entities and Not-for-Profit Organizations

Analysts should carefully consider whether a particular rule will directly affect government entities, not-for-profit organizations, or households.[8] For example, air pollution regulations that apply to power plants may affect government entities such as municipally-owned electric companies. Air regulations that apply to vehicles may affect municipal buses, police cars, and public works vehicles. Effluent guidelines for machinery repair activities may affect municipal garages. The profile of these affected entities should include a brief description of relevant factors or characteristics.

Relevant factors for *government entities* may include:

- Number of people living in the community;
- Property values;
- Household income levels (e.g, median, income range);
- Number of children;
- Number of elderly residents;
- Unemployment rate;
- Revenue amounts by source; and
- Credit or bond rating of the community.

If property taxes are 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 a government entity serves multiple communities, such as a regional water or sewer authority, then relevant information should be collected for all the communities covered by the government entity. Socioeconomic factors influence demands on state or local government resources; for example a high proportion of children means more educational resources.

Data on community size, income, number of children and elderly, and unemployment levels are available from the U.S. Census Bureau. Data on property values, amount of revenue collected from each revenue source, and credit rating may be available from the community or state finance agencies. Most county websites provide information on property values. Private companies, such as Standard and Poor's (S&P), or Fitch's, provide community credit ratings.

8 Government entities that may be affected include states, cities, counties, townships, water authorities, villages, Indian Tribes, special districts, and military bases. Not-for-profit entities that may be affected include not-for-profit hospitals, colleges, universities, and research institutions.

Depending on the number of communities affected and the level of detail warranted, the analysis may rely on generally available aggregate data only. In other cases, a survey of affected communities may be necessary.[9]

Relevant characteristics of *not-for-profit entities* include:

- Entity size and size of community served;
- Goods or services provided;
- Operating costs; and
- Amount and sources of revenue.

If the entity is raising its revenues through user fees or charging a price for its goods or 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 and beneficiaries. If it relies on government funding (such as Medicaid) then possible future changes in these programs should be identified.

9.2.2.3 Profile of Small Entities

Small entities include small businesses, small governments and small not-for-profit institutions. While these entities may require special considerations, as detailed below, the profiling of them should follow the same steps as discussed above.

9.2.2.4 Data Sources for Profiles

Profiles generally rely on information from the following sources: websites for affected communities, industry trade publications, and the U.S. Census Bureau.[10] Relevant literature can be useful in characterizing industry activities and markets as well as regulations that already affect the industry. Relevant literature can usually be efficiently identified through a computerized search using on-line services such as Dialog, BRS/Search Services, Dow Jones News/Retrieval, or EconLit. These on-line services contain more than 800 databases covering business, economic, and scientific topic areas. Table 9.2 describes some commonly used data sources for retrieving quantitative data.[11]

The industry profile may also identify situations where insufficient data are available from standard sources. This situation could potentially arise when the affected industry has many product lines or activities affected by the rule. In addition, for some rules it may be difficult to identify the appropriate NAICS industry for all the firms or facilities affected by the rule if the industry can be categorized in multiple ways. In these cases, and particularly if facility-level data are required to estimate economic impacts, a survey of affected facilities may be required to provide sufficient data for analysis.

9.2.3 Detailing Impacts on Industry

This section explains how to determine the impact on individual plants or businesses so as to identify whether a particular plant or industry is likely to bear a disproportionate portion of the costs or benefits of a regulation.

9.2.3.1 Impacts on Prices

Predicted impacts on prices form the basis for determining how compliance costs are distributed between the directly-affected firms, their customers, and other related parties in a typical market. At one extreme, regulated firms may not be able to raise prices at all, and would consequently 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.

9 In cases where a survey is needed, care should be taken to comply with the requirements of the Paperwork Reduction Act (PRA) (44 U.S.C. 3501).

10 Academic literature may or may not contain quantitative data.

11 The Thomas Registry (www.thomasnet.com) is a source of qualitative information on manufacturing companies in the United States (accessed on January 21, 2011). In addition, Lavin (1992) provides sources of business information.

Table 9.2 - Commonly Used Profile Sources for Quantitative Data

Source	Data
Trade Publications and Associations	Market and technological trends, sales, location, regulatory events, ownership changes
U.S. Department of Commerce, Economic Census (www.census.gov)	Sales, receipts, value of shipments, payroll, number of employees, number of establishments, value added, cost of materials, capital expenditures by sector, household and community characteristics
U.S. Department of Commerce, *U.S. Industry & Trade Outlook* (http://www.ita.doc.gov/td/industry/OTEA/outlook/ or http://outlook.gov/)	Description of industry, trends, international competitiveness, regulatory events
U.S. Department of Commerce, *Pollution Abatement Costs and Expenditures Survey* (www.census.gov/mcd)	Pollution abatement costs for manufacturing facilities by industry, state, and region
U.S. Department of Commerce, *Census of Governments* (www.census.gov/govs/index.html)	Revenue, expenditures debt, employment, payroll, assets for counties, cities, townships, school districts
United Nations, International Trade Statistics Yearbook	Foreign trade volumes for selected commodities, major trading partners
Risk Management Association, *Annual Statement Studies* (www.rmahg.org/ann_studies/asstudies.html)	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 (www.dnb.com/us/)	Type of establishment, NAICS code, address, facility and parent firm revenues and employment
Standard & Poors (www.standardandpoors.com)	Publicly-held firms, prices, dividends, and earnings, line-of-business and geographic segment information, S&P ratings, quarterly history (10 years), income statement, ratio, cash flow and balance sheet analyses and trends
Securities and Exchange Commission Filings and Forms (EDGAR System Database) (www.sec.gov/edgar.shtml)	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

Suppliers to the directly-affected firms might bear part of the burden in lost earnings if the regulation results in a decline in demand for particular products.[12] At the other extreme, firms may be able to raise prices enough to recover costs fully. In this case, there is no impact on the profitability of the directly-affected firms but their customers bear the burden of increased prices. Assuming perfect competition, the amount of price pass-through depends on the relative elasticity of supply and demand. Another economic impact to consider is the potential backward shifting of regulatory costs (e.g., lowering wages of workers).

In general, the likelihood that price increases will occur can be evaluated by considering whether competitive conditions allow the affected facilities to pass their costs on to consumers.

12 For example, regulations limiting SO_2 emissions may result in reduced demand for high-sulfur coal, which results in a fall in the price of such coal and lost profits for its producers. While there is no clear rule for how far down the chain of effects one needs to consider, it is important to address effects that are likely to be substantial.

The methods used to conduct the analysis of the directly-affected markets depend on the availability of appropriate estimates of supply and demand elasticities.[13] As noted above, in cases where reliable estimates of elasticities are not available, the analyst must rely on a more basic investigation of the characteristics of supply and demand in the affected market to reach a conclusion about the likelihood of full or partial pass-through of costs via price increases. An examination of the number of firms, quantity of a product produced, and industry size will provide basic information about supply and demand. If an industry is highly concentrated with few producers then firms may be able to easily pass costs on to households and a 100 percent pass-through assumption may be justifiable. Of course, an industry with many producers would mean the opposite assumption.

9.2.3.2 Impacts on Production

Abatement costs tend to be only a small fraction of total manufacturing revenues. As such, even small changes in wage rates, materials costs, or capital costs are likely to have a much larger effect on manufacturing industries than any changes in environmental regulation. The U.S. Census Bureau collects data on pollution abatement capital expenditures and operating costs incurred to comply with local, state, and federal regulations and on voluntary or market-driven pollution abatement activities.[14] According to the 2005 PACE Survey, the U.S. manufacturing sector spent approximately $20.7 billion dollars on pollution abatement operating costs. This figure represents less than 1 percent of the sector's total revenue, which is similar to the historical average. Moreover, every manufacturing industry, including the most highly regulated ones, spend less than 1.2 percent of their revenues on pollution abatement. Figure 9.1 presents data for the five industries with the highest pollution abatement operating costs (PAOC) as a percent of total revenues.

Figure 9.1 - Pollution Abatement Costs as a Percentage of Total Revenues for Industries with Highest Pollution Abatement Costs in 2005

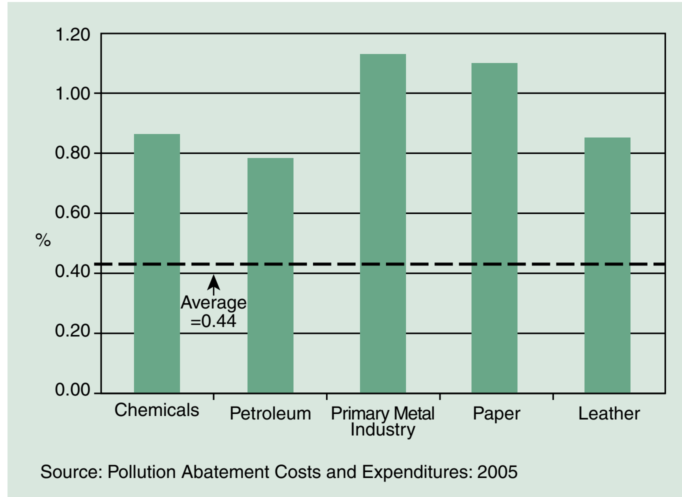
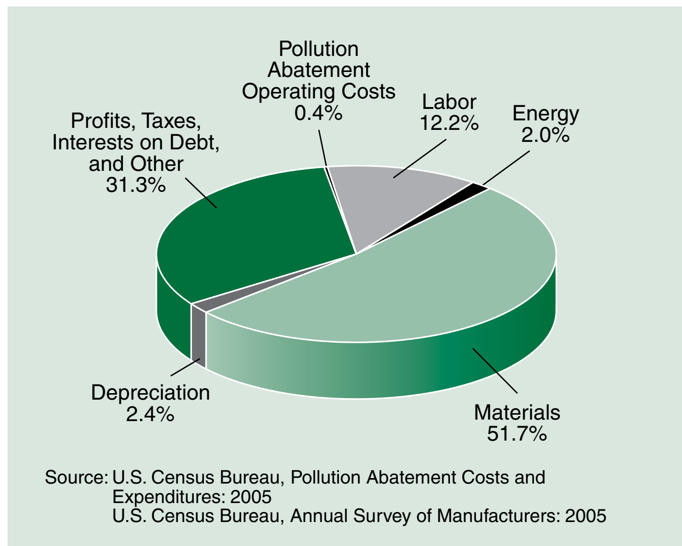


Figure 9.2 - Pollution Abatement Costs are a very Small Percentage of Total Manufacturing Costs



Considering the historical data, it is unlikely that the typical pollution control regulation will sufficiently increase the cost of doing business so as to make a meaningful part of production unprofitable, or will significantly reduce the quantity of output demanded as producers raise their prices to maintain profitability. Figure 9.2 shows the relative magnitude of each cost category for the manufacturing sector. Based on these relative magnitudes, reducing abatement costs by 10 percent will only reduce the total costs faced by industry by less than 1 tenth of 1 percent. Conversely, lowering material costs by 10 percent will reduce total costs by just over 5 percent as

13 See Appendix A for a more complete discussion of elasticity.

14 More detail on the PACE Survey is available at http://yosemite.epa.gov/ee/epa/eed.nsf/pages/pace2005.html (accessed March 13, 2011).

material costs were roughly 50 percent of revenues in 2005. Exceptions may be regulations banning the sale or manufacture of a specific product (e.g., a chemical ban) or when a production process is made obsolete. In these situations, the analyst should assess whether the existing plants have other profitable uses.

9.2.3.3 Impacts on employment

The chapters on benefits (Chapter 7) and costs (Chapter 8) point out that regulatory-induced employment impacts are not, in general, relevant for a BCA. For most situations, employment impacts should not be included in the formal BCA.[15] However, if desired the analyst can assess the employment impacts of a regulation as part of an EIA. If this task is undertaken, the analyst needs to quantify all of the employment impacts, positive and negative, to present a complete picture of the effects. This section identifies pitfalls often encountered when performing an EIA and discusses the preferred approaches for conducting one.

Many analyses only present the employment effect on the regulated industry as a result of higher regulatory compliance costs. In doing so, these analyses make simplifying assumptions that employment in a given industry is proportional to output, i.e., if production goes down by 1 percent, employment goes down by 1 percent. These limited assessments on employment impacts from regulation examine how higher manufacturing costs lead to fewer sales and therefore lower employment in that sector. However, empirical and theoretical modeling suggests that these simplified relationships are faulty and should not be used.

In fact, it is not even clear that employment in the regulated industry goes down as a result of environmental regulation. Morgenstern et al. (2002) decompose the labor consequences in an industry facing increased abatement costs. They identify three separate components:

- **Demand effect:** Higher production costs raise market prices. Higher prices reduce consumption (and production) reducing demand for labor within the regulated industry;
- **Cost effect:** As production costs increase, plants use more of all inputs including labor to produce the same level of output. For example, pollution abatement activities require additional labor services to produce the same level of output; and
- **Factor-shift effect:** Post-regulation production technologies may be more or less labor intensive (i.e., more/less labor is required per dollar of output).

Morgenstern et al. empirically estimate this model for four highly polluting/regulated industries to examine the effect of higher abatement costs from regulation on employment. They conclude that increased abatement expenditures generally do not cause a significant change in employment. Specifically, their results show that, on average across the industries they consider, each additional $1 million of spending on pollution abatement results in a (*statistically insignificant*) net increase of 1.5 jobs. However, they find that for two of their four industries (pulp and paper, and steel) additional abatement spending leads to a *statistically significant*, yet quite small, net increase in jobs due to the substitution of labor for other inputs and relatively inelastic estimated demand for their output.[16]

Finally, one effect that Morgenstern et al. do not consider is the effect regulation has on employment in industries that make substitute products, often cleaner products. Demand for these products increases as consumers respond to changes in costs. For example, more expensive virgin paper will cause a shift to more recycled paper. The recycled paper industry will employ more workers as sales increase. Similarly, employment in industries that are complements

15 Appendix C discusses long-term, structural employment changes brought on by land clean up and reuse or other policies that may have a benefit component to them.

16 These results are similar to Berman and Bui (2001) who find that while sharply increased air quality regulation in Los Angeles to reduce NOx emissions resulted in large abatement costs they did not result in substantially reduced employment.

may decrease. The analyst should also take these effects into consideration when analyzing the effect of regulations on employment.

In addition to the changes in the regulated industry as modeled by Morgenstern et al., the analyst should assess the increased employment in the environmental protection industry. The engineering analysis may provide some data on the labor required to design, build, install (and in some cases operate) the pollution control equipment. For example, a recent study by Industrial Economics Inc. shows that a $19 million order for a new scrubber will immediately fund 77 to 91 new jobs for a year constructing and installing the new equipment. It will also create 16 permanent jobs to operate the new equipment (Price et al. 2010).

9.2.3.4 Impacts on Profitability and Plant Closures

In other cases, analysts may assess the impacts of rules on the profitability of specific firms or industry segments and identify potential plant closures based on a financial analysis. If partial or full plant closures are projected, then 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 simply vanish. If excess industry capacity exists in the baseline and facilities are able to operate profitably while complying with the rule, then these facilities may expand production to meet the demand created by the loss of plants that are no longer able to operate profitably. Some surviving plants could experience increases in production, capacity utilization, and profits even though they are subjected to regulatory requirements, if their competitors face even greater cost increases.

9.2.3.5 Impacts on Related Industries

The economic and financial impacts of regulatory actions spread to industries and communities that are linked to the regulated industries and to the pollution abatement industries, resulting in indirect business impacts. To build scrubbers, the environmental protection industry will order more steel. If a plant produces less, it will order fewer raw materials. These indirect impacts may include employment and income gains and losses.

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 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 important to consider less obvious but potentially significant links, such as basic suppliers like electricity generators.

For these reasons, 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 how employment, competitiveness, and income are likely to change for related entities and households given a certain amount of employment, competitiveness, and income in a regulated market.

9.2.3.6 Impacts on Economic Growth and Technical Inefficiency

While regulatory interventions can theoretically lead to macroeconomic impacts, such as growth and technical efficiency, such impacts may be impossible to observe or predict. In some cases, however, it may be feasible to use macroeconomic models to evaluate the regulatory impact on GDP, factor payments, inflation, and aggregate employment. For regulations that are expected to have significant impacts in a particular region, use of regional models, either general equilibrium or other regionally-based models, may be valuable.[17]

Typically in regulatory impact analyses some macroeconomic regulatory effects go unquantified due to analytic constraints. For example, price changes induced by a regulation can lead to technical inefficiency because firms are not choosing the production techniques that minimize

17 Chapter 8 discusses the use of regional modeling.

the use of labor and other resources in the long run. However, measuring these effects can be difficult due to data or other analytical limitations.

9.2.3.7 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.[18] An analysis of impacts on competitiveness begins by examining barriers to entry and market concentration, and by answering the following two key questions:

- **Does 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.[19] 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 may not be sufficient to be a barrier to entry.
- **Does the regulation tend to create or enhance market power and reduce the economic efficiency of the market?** Important measures of competitiveness of an industry are degrees of horizontal and vertical integration (i.e., concentration) between both buyers and sellers in the baseline compared to post-compliance. If an industry becomes more concentrated as a result of the regulation then there are fewer firms within the industry. In this case, market power will be concentrated in the hands of a few entities, which may result in a less efficient market than before the regulation. Closely related to concentration, product differentiation may occasionally either increase or decrease due to a regulatory action. A regulation may result in less product differentiation due to restrictions on production. This could mean that market power is more concentrated among the firms that manufacture the product.

18 See Jaffe et al. (1995) for an overview.

19 Sunk costs are sometimes referred to as exit barriers.

9.2.3.8 Impacts on Energy Supply, Distribution, or Use

EO 13211 requires agencies to prepare a Statement of Energy for "significant energy actions," which are defined as significant regulatory actions (under EO 12866) that also are "likely to have a significant adverse effect on the supply, distribution, or use of energy."[20] These significant adverse effects are defined as:

- Reductions in crude oil supply in excess of 10,000 barrels per day;
- Reductions in fuel production in excess of 4,000 barrels per day;
- Reductions in coal production in excess of 5 million tons per year;
- Reductions in natural gas production in excess of 25 million mcf per year;
- Reductions in electricity production in excess of 1 billion kilowatt-hours per year or in excess of 500 megawatts of installed capacity;
- Increases in energy use required by the regulatory action that exceed any of the thresholds above;
- Increases in the cost of energy production in excess of 1 percent;
- Increases in the cost of energy distribution in excess of 1 percent; or
- Other similarly adverse outcomes.

For actions that may be significant under EO 12866, particularly for those that impose requirements on the energy sector, analysts must be prepared to examine the energy effects listed above.

20 See Section 2.1.6 for EPA and OMB's guidance on EO 13211.

9.2.4 Detailing Impacts on Governments and Not-for-Profit Organizations

Section 9.3.5 discusses how to measure the impact of regulations and requirements on private entities, such as firms and manufacturing facilities. When dealing with private entities, an important focus is on measures that assess changes in profits (or proxy measures of profit). This section describes impact measures for situations where profits and profitability are not the focus of the analysis. Rather, the ultimate measure of impacts is the ability of the organization or its residents to pay for the requirements. Many of the same questions apply:

- Which entities are affected and what are their characteristics?
- To what extent does the regulation increase operating costs?
- To what extent does the regulation impact operating procedures?
- Does the regulation change the amount and/or quality of the goods and services provided?
- Can the entity raise the necessary capital to comply with the regulation?
- Does the regulation change the entity's ability to raise capital for other projects?

EPA regulations can affect governments and not-for-profit organizations in at least three significant ways. First, a regulation may directly impose requirements on the entity, such as imposing water pollution requirements for publicly-owned wastewater treatment works, or initiating air pollution restrictions that affect municipal bus systems or power plants. Second, a regulation may impose implementation and enforcement costs on government agencies. Finally, a regulation may impose indirect costs. For example increased unemployment due to reduced production (or even plant closure) could result in less tax revenues in a community.

9.2.4.1 Direct Impacts on Government and Not-for-Profit Entities

Direct impact measures can fall into two categories:

- Those that measure the impact itself in terms of the relative size of the costs and the burden it places on residents; and
- 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 aspects of the direct impacts.

Measuring the relative cost and burden of the regulations

There are three commonly used approaches to measuring the direct burden of a 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 affected service.** This measure defines the impact as narrowly as possible and measures impacts according to the increase in costs to the entity. In practice, EPA has often defined compliance costs that are less than 1 percent of the current annual costs of the activity as 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 the RFA Guidance (U.S. EPA 2006b).
- **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. The ability of residents to pay for the costs affects government entities because fees and taxes on residents fund these entities. To the extent that residents can (or cannot) pay for the cost increases, government entities will

be impacted. Commonly referred to as the "Income Test," this measure is described in the RFA Guidance (U.S. EPA 2006b) and the EPA Office of Water *Interim Economic Guidance for Water Quality Standards: Workbook* (U.S. EPA 1995a).[21] Costs can be compared to either median household or median per capita income. In calculating the per household or per capita costs, the actual allocation of costs needs to be considered. If the costs are paid entirely through property taxes, and the community is predominately residential, then an average per household cost is probably appropriate. If 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 a more narrow measure may be appropriate. If some of the costs are borne by local firms, then that portion of the costs should be analyzed separately.

There are two commonly used impact measures for *not-for-profit entities:* (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 are paid out of the institution's assets.

Measuring the economic and financial health of the community or government entity

The second category of direct impact measures examines the economic and financial health of the community involved, since this affects its ability to finance or pay for 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 debt indicators have been developed and used, this section describes two common indicators. One measure is the government entity's bond rating. Awarded by companies such as Moody's and Standard & Poor's, bond ratings evaluate a community's credit capacity and thus reflect the current financial conditions of the government body.[22] A second frequently used measure is the ratio of overall net debt to the full market value of taxable property in the community, i.e., debt to be repaid by property taxes. Overall net debt should include the debt of overlapping districts. For example, a household may be part of a town, regional school district, and county sewer and water district, all of which have debt that the household is helping to pay.[23] See Table 9.3 for interpretations of the values for these measures. **Debt measures are not always 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. 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

21 For example, in the water guidance and other EPA Office of Water analyses compliance costs are considered to have little impact if they are less than 1 percent of household income. Compliance costs greater than 2 percent are categorized as a large impact, and a range from 1 to 2 percent fall into a gray area and are considered to have an indeterminate impact.

22 The indicators and benchmark values in Table 9.3 are drawn from *Combined Sewer Overflows — Guidance for Financial Capability Assessment and Schedule Development*, which discusses how to assess the feasibility of systems being able to comply with rules (U.S. EPA 1997b). These are general benchmarks that may prove useful in assessing financial stability in an EIA.

23 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 indicates a strong financial condition, between $1,000 and $3,000 indicates a mid-range or gray area, and greater than $3,000 indicates a weak financial condition.

Table 9.3 - 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% - 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 revenue as percent of full market value of taxable property	Above 4%	2% - 4%	Below 2%
Property tax collection rate	Less than 94%	94% - 98%	More than 98%

Source: U.S. EPA 1997b

to full market value of taxable property is not relevant. Information on debt and assessed property values are available from the financial statement of each community. The state auditor's office is likely to maintain this information for all communities within a state.

- **Indicators of the economic/financial condition of the households in the community.** There are a wide variety of household economic and financial indicators. Commonly used measures are the unemployment rate, median household income, and foreclosure rates. Unemployment rates are available from the Bureau of Labor Statistics. Median household income is available from the U.S. Census Bureau. Benchmark values for these and other measures are presented in Table 9.3.
- **Financial management indicators.** This category consists of indicators that gauge the general financial health of the community, as opposed to the general financial health of the residents. Because most local communities rely on property taxes as their major source of revenues, there are two ratios that provide an indicator of financial strength. First, property tax revenue as a percentage of the full market value of taxable property indicates the burden that property taxes place on the community.[24] Second, the property tax collection rate gauges the efficiency with which the community's finances are managed, and indirectly whether the tax burden may already be excessive. As the property tax burden on taxpayers increases, they are more likely to avoid paying their taxes or to pay them late.

Measuring the financial strength of *not-for-profit* entities includes assessing:

- The size of the entity's reserves;
- How much debt the entity already has and how its annual debt service compares to its annual revenues; and
- How the entity's fees or user charges compare with the fees and user charges of similar institutions.

As with government entities, this analysis is meant to judge whether the entity is in a strong or weak financial position to absorb additional costs.

9.2.4.2 Administrative, Enforcement, and Monitoring Burdens on Governments

Many EPA programs require effort on the part of

24 If the state caps local property taxes (e.g., Proposition 13 in California or Proposition 2½ in Massachusetts) then it may be relevant to examine the ratio of property tax to the allowed level of the taxes.

different levels of government for administration, enforcement, and monitoring. These costs must be included when estimating impacts of a regulation to comply with UMRA and to calculate the full social costs of a program or rule. See Chapter 8 for more information on government regulatory costs.

9.2.4.3 Induced Impacts on Government Entities

The induced impacts on government entities should also be considered. For example, a manufacturing facility may reduce or suspend production in response to a regulation, thus reducing the income levels of its employees. In turn, these reductions will spread through the economy by means of changes in household expenditures. These induced impacts include the 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.

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. As some linkages may be more localized than others, it is important to clearly identify where the gains and losses occur.

9.2.5 Detailing Impacts on Small Entities

The Regulatory Flexibility Act, as amended by the Small Business Regulatory Fairness Act of 1996 (RFA), and Section 203 of the Unfunded Mandates Reform Act of 1995 (UMRA) require agencies to consider a proposed regulation's economic effects on small entities, specifically, small businesses, small governmental jurisdictions, or small not-for-profit organizations. The definition of "small" for each of these entities is described below. For guidance on when it is necessary to examine the economic effects of a regulation under the RFA or UMRA, analysts should consult EPA guidelines on these administrative laws (U.S. EPA 2006b and U.S. EPA 1995b, respectively). In general, the Agency must fulfill certain procedural and/or analytical obligations when a rule has a "significant impact on a substantial number of small entities" (abbreviated as SISNOSE) under the RFA or when a rule might "significantly" or "uniquely" affect small governments under Section 203 of UMRA.

9.2.5.1 Small Businesses

The RFA requires agencies to begin with the definition of small business that is contained in the Small Business Administration's (SBA) small business size standard regulations.[25] 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 also publish any alternative definition in the *Federal Register* (U.S. EPA 2006b).

The analytical tasks associated with complying with the RFA include a screening analysis for SISNOSE. If the screening analysis reveals that a rule *cannot* be certified as having no SISNOSE, then the RFA requires a regulatory flexibility analysis be conducted for the rule, which includes a description of the economic impacts on small entities. Impacts on small businesses are generally assessed by estimating the direct compliance costs and comparing them to sales or revenues. Because an estimate of direct compliance costs tends to be a conservatively low estimate of a regulation's impact, further analysis examining the impacts

25 The current version of SBA's size standards can be found at http://www.sba.gov/size (accessed March 13, 2011).

discussed in Section 9.3.3 (specifically in relation to small businesses) may provide additional information for decision makers.[26]

9.2.5.2 Small Governmental Jurisdictions

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 and publication in the *Federal Register*. 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" (U.S. EPA 2006b). Under the RFA, economic impacts on small governments are included in the SISNOSE screening analysis, and any required regulatory flexibility analysis for a rule.

UMRA uses the same definition of small government as the RFA with the addition of tribal governments. Section 203 of UMRA requires the Agency to develop a "Small Government Agency Plan" for any regulatory requirement that might "significantly" or "uniquely" affect small governments. In general, "impacts that may significantly affect small governments include — but are not limited to — those that may result in the expenditure by them of $100 million [adjusted annually for inflation] or more in any one year." Other indicators that small governments are uniquely affected may include whether they would incur the higher per-capita costs due to economies of scale, a need to hire professional staff or consultants for implementation, or requirements to purchase and operate expensive or sophisticated equipment.[27] See Section 9.3.4 for information on measures of impacts to governments in general.

26 See Agency guidance (U.S. EPA 2006c) for details on complying with the RFA.

27 Guidance on complying with Section 203 of UMRA, "Interim Small Government Agency Plan," is available on EPA's intranet site, ADP Library at http://intranet.epa.gov/adplibrary/statutes/umra.htm (accessed March 21, 2011, internal EPA document)

9.2.5.3 Small Not-for-Profit Organizations

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 may 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 and publication in the *Federal Register.* Under the RFA, economic impacts on small not-for-profit organizations are included in the SISNOSE screening analysis, and if required, the regulatory flexibility analysis for a rule. See Section 9.3.4 for more information on measuring impacts on not-for-profit organizations in general.

9.3 Approaches to Modeling in an Economic Impact Analysis

This section returns to the methods for estimating social costs covered in Chapter 8, adding more insight on their application to EIA. The reader should refer to Chapter 8 for a more in-depth discussion. As noted above, the analytic assumptions used for the EIA of a particular regulation should be consistent with those used for the corresponding BCA.

9.3.1 Direct Compliance Costs

The simplest approach to measuring the economic impacts is to estimate and verify the private costs of compliance. This is necessary regardless of whether the entities affected are for-profit, governmental, communities, or not-for-profit. Direct compliance costs are considered the most conservative estimate of private costs and include annual costs (e.g., operation and maintenance of pollution control equipment), as well as any capital costs. Direct compliance costs do not include implicit costs.

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 and any differences in response across sub-sectors and/or geographic regions. Second, the costs for each response need to be examined to determine if all elements are included and if the costs are consistent within a given base year. To ensure consistency across years, either a general inflation factor, such as the GDP implicit price deflator, or various cost indices specific to the type of project should be used.[28] 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 on programs involving recycling of household wastes. The cost of acquiring a permit includes the permit fee as well as 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 a registrant's time.

In addition, it is important to recognize that these expenditures may have other benefits and costs. For example, they may confer tax breaks (complying with regulations may be a tax deductible expense) and the new capital may be more productive than the old capital. These "offsets" should be considered, particularly when they may be substantial.

There are several issues analysts should consider when estimating the direct compliance costs of environmental polices for an EIA. These include:

- **Before- versus after-tax costs.** For businesses, the cost of complying with regulations is generally deductible as an expense for income tax purposes. Therefore, the effective burden 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. Operating costs are generally fully deductible as expenses in the year incurred. Capital investments associated with compliance must generally be depreciated.[29] 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, not-for-profits, and households.
- **Transfers.** 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. However, individual private cost estimates used in the EIA include such transfers.[30]
- **Discounting.** Compliance costs often vary over time, perhaps requiring initial capital investments and then continued operating costs. To estimate impacts, the stream of costs is generally discounted to provide a present value of costs that reflects the time value of money.[31] 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.[32] 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. For firms, the cost of capital may also be determined by their ability to deduct debt from their tax liability.

28 The GDP implicit price deflator is reported by the U.S. DOC, BEA in its *Survey of Current Business* (http://www.bea.gov/scb/index.htm). The annual *Economic Report of the President*, Executive Office of the President, is another convenient source for the GDP deflator, available at www.gpoaccess.gov/eop/ (accessed March 13, 2011).

29 Current federal and state income tax rates can be obtained from the Federation of Tax Administrators, *State Tax Rates & Structure*, available at http://www.taxadmin.org/fta/rate/default.html (accessed January 31, 2011).

30 These transfers cancel out in a BCA. In an EIA the distribution of results is important, therefore the transfers are included.

31 The present value of costs can then be annualized to provide an annual equivalent of the uneven compliance cost stream. Annualized costs are also discussed in Chapter 6.

32 While the discount rate differs, the formula used to discount private costs is the same as used for social costs. See Chapter 6 for details.

- **Annualized costs.** Annualizing costs involves calculating the annualized equivalent of the stream of cash flows associated with compliance over the period of analysis. This provides a single annual cost number that reflects the various components of compliance costs incurred over this period. 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, such as quantity of production. Other components of cost may be fixed with respect to production or other size measures, such as the costs involved in reading and understanding regulatory requirements. 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 are reflected in the compliance costs used to analyze economic impacts.[33] Using the same average annualized cost per unit of production for all firms may mask the importance of such fixed costs and understate impacts on small entities.

9.3.2 Partial Equilibrium Models

A partial equilibrium framework is an alternative way to examine distributional effects when impacts are limited to a few directly and indirectly affected output markets only. For example, a regulation may increase the costs of producing a particular chemical. Partial equilibrium models can be used to examine the distribution of these changes across directly affected industries, and a small number of indirectly affected entities (e.g., upstream and downstream). Partial equilibrium models can range in size from an analysis that estimates compliance costs for the affected industry only (i.e., direct compliance costs) to multi-market models encompassing several directly and indirectly affected sectors.

If a single-market partial equilibrium model is the only information source available for an analysis of impacts, then it may be possible to adopt further assumptions and acquire additional data to approximate impacts on other areas of concern. This may include deriving ratios to aggregate changes in order to assign these changes to specific regions or sectors. These new assumptions should be consistent with those used for the corresponding BCA.

Multi-market models consider 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). These models are best used when potential impacts on related markets might be considerable, but more complete modeling using a CGE framework may not be available or practical. Partial equilibrium models may also be more appropriate for regionally based or resource specific regulations that are too specific for more aggregated CGE models.[34] Care should be taken, however, to avoid double counting, particularly when both upstream and downstream entities are affected and included in the partial equilibrium analysis. If cost increases due to a regulation are passed on from the upstream to the downstream businesses then analysts should take care not to include impacts on both sets of entities to avoid double counting results.

33 Economies of scale characterize costs that decline on a per unit basis as the scale of the operation increases.

34 See the discussion of multi-market modeling in Chapter 8 and Just et al. (1982).

9.3.3 Computable General Equilibrium Models

CGE models are particularly effective in assessing resource allocation and welfare effects. These effects include the allocation of resources across sectors (e.g., employment by sector), the distribution of output by sector, the distribution of income among factors, and the distribution of welfare across different consumer groups, regions, and countries. As noted in Chapter 8, for example, regulations in the electric utility sector are likely to cause electricity prices to increase. The price increase will affect all industries that use electricity as an input to production (i.e., most industries), as well as households. A CGE model can assess the distribution of the changes in production and consumption that result. By design, the basic capacity to describe and evaluate these sorts of impacts exists to some extent within every CGE model. More detailed impacts (e.g., affects on a particular facility) or impacts of a particular kind (e.g., affects on drinking water) 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. Additional complexities can be specified for the model in a variety of ways. Consumers may be divided into different groups by income, occupation, or other socioeconomic criteria. Producers can 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 provide a public good or run a deficit. CGE models can 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 can be solved dynamically over a long time horizon, incorporating intertemporal decision making on the part of consumers or firms. These choices have implications for the treatment of savings, investment, and the long-term profile of consumption and capital accumulation.

As effective as CGE models can be for looking at long-term resource allocation issues, they have limitations for the kinds of impact analyses described above. CGE models assume that markets clear in every period and often do not consider short-term adjustment costs, such as lingering unemployment. The analyst should be careful to select a model that does not assume away the underlying issue addressed by the distribution analysis. Moreover, a CGE model may not be feasible or practical to use when data and resources are limited or when the scope of expected significant market interactions is limited to a subset of economic sectors. In such instances a partial equilibrium model can be adopted as a more appropriate alternative to a CGE model.[35] Finally, it is worth noting that while CGE modeling is complex, the effort may be worthwhile when data are available and the distributional impacts are likely to be widespread.

35 For a discussion of CGE analysis see Chapter 8 and Dixon et al. (1992).

Chapter 10

Environmental Justice, Children's Environmental Health and Other Distributional Considerations

Evaluating a regulation's distributional effects is an important complement to benefit-cost analysis. Rather than focusing on quantifying and monetizing total benefits and costs, economic impact and distributional analyses examine how a regulation allocates benefits, costs and other outcomes across populations or groups of interest. See Chapter 9 of these *Guidelines* for more information on analyzing economic impacts. This chapter considers the distribution of environmental quality and human health risks across several populations: those that have traditionally been the focus of environmental justice (EJ) (i.e., minority, low-income, or indigenous populations); children; and the elderly. Consideration of costs or other potential impacts may also be addressed in a distributional analysis using approaches discussed in this chapter. The chapter also briefly discusses inter-generational impacts.

This chapter suggests approaches that EPA program offices can use for characterizing distributional effects of policy choices associated with rulemaking activities. Based on academic literature and EPA documents and policies, the chapter provides a variety of methodological approaches that may be suitable across various regulatory scenarios. A clear consensus does not exist, however, regarding the most appropriate methods. Instead, this chapter provides a broad overview of options for analyzing distributional effects in regulatory analysis. Information in the chapter is intended to provide flexibility to programs that face dissimilar data, resources and other constraints while introducing greater consistency in the way EJ is addressed in rulemaking activities.[1]

The purpose of analyzing distributional effects in regulatory analysis is to examine how benefits (e.g., risk reductions or environmental quality) and, when relevant and feasible, costs are distributed across population groups and lifestages of interest.[2] While the chapter is focused on EJ, children, and the elderly, the methods discussed could be applied to any population of concern.

The chapter begins with an overview of Executive Orders (EOs) and policies related to distributional analyses. It then discusses the analysis of distributional impacts in the context of EJ and children's health. The chapter concludes with a brief discussion of other distributional considerations, including the elderly and inter-generational impacts that may arise in select rules.

10.1 Executive Orders, Directives, and Policies

Consideration of distributional effects arises from a variety of executive orders, directives, and other

1 The guidance in this chapter complements, and does not supersede, any subsequent EJ-related guidance released by EPA. In addition, the Office of Environmental Justice website (http://www.epa.gov/environmentaljustice/resources/policy/index.html) provides resources on Plan EJ2014 and other implementation guidelines related to EJ (accessed on January 24, 2012).

2 This chapter recommends examining the distribution of benefits prior to monetization for reasons discussed in Section 10.1.

documents with broad coverage, including:[3]

- EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" (1994);
- EO 13045, "Protection of Children From Environmental Health Risks and Safety Risks" (1997);
- EO 13166, "Improving Access to Services for Persons With Limited English Proficiency" (2000); and the subsequent EPA Order No.1000.32, "Compliance with Executive Order 13166: Improving Access to Services for Persons with Limited English Proficiency" (2011);
- EO 13175, "Consultation and Coordination with Indian Tribal Governments" (2000);
- EO 12866, "Regulatory Planning and Review" (1993);
- *Circular A-4*, Regulatory Analysis (OMB 2003);
- National Environmental Policy Act (NEPA) Guidance (U.S. EPA 1998a);
- EPA's *Interim Guidance on Considering Environmental Justice During the Development of an Action* (U.S. EPA 2010a); and
- EPA's FY2011-2015 Strategic Plan (U.S. EPA 2010b).

Each of these is described below. Some environmental statutes may also identify population groups that merit additional consideration.[4]

EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations"[5] (1994), calls on each Federal agency to make achieving EJ part of its mission. It directs Federal agencies, "[t]o the greatest extent practicable and permitted by law," to "identify[...] and address[...], as appropriate, disproportionately high and adverse human health or environmental effects" of agency programs, policies, and actions on minority populations and low-income populations. Issued by President Clinton in 1994, it requires that EJ be considered in all Agency activities, including rulemaking activities.

The President issued a memorandum to accompany EO 12898 directing Federal agencies to analyze environmental effects, including human health, economic, and social effects, of Federal actions when such analysis is required under the National Environmental Policy Act (NEPA). The Presidential memorandum also states that existing civil rights statutes provide opportunities to address environmental hazards in minority communities and low-income communities.[6]

EO 13045, "Protection of Children From Environmental Health Risks and Safety Risks" (1997), states that each Federal agency: (1) shall make it a high priority to identify and assess environmental health risks and safety risks that may disproportionately affect children; and (2) shall ensure that its policies, programs, activities, and standards address disproportionate risks to children that result from environmental health risks or safety risks. The EO also states that each "covered regulatory action" submitted to the Office of Management and Budget (OMB), unless prohibited by law, should be accompanied by ". . . an evaluation of the environmental health or safety effects of the planned regulation on children."[7]

3 EPA's Regulatory Management Division's Action Development Process Library (http://intranet.epa.gov/adplibrary) is a resource for accessing relevant statutes, executive orders, and EPA policy and guidance documents in their entirety (accessed on December 1, 2011).

4 See *Plan EJ 2014 Legal Tools* (U.S. EPA 2011a) for a review of legal authorities under the environmental and administrative statutes administered by EPA that may contribute to the effort to advance environmental justice.

5 This chapter addresses analytical components of EO 12898, and does not cover other components such as ensuring proper outreach and meaningful involvement.

6 "In accordance with Title VI of the Civil Rights Act of 1964, each Federal agency shall ensure that all programs or activities receiving Federal financial assistance that affect human health or the environment do not directly, or through contractual or other arrangements, use criteria, methods, or practices that discriminate on the basis of race, color, or national origin." See *Memorandum for the Heads of All Departments and Agencies: Executive Order on Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations* (White House 1994).

7 A "covered regulatory action" is any substantive action in a rulemaking that may be economically significant (i.e., 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, or the environment) and concern an environmental health risk that an agency has reason to believe may disproportionately affect children.

EO 13166, "Improving Access to Services for Persons With Limited English Proficiency" (2000), requires Federal agencies to examine the services they provide, identify any need for services to those with limited English proficiency (LEP), and develop and implement a system to provide those services so LEP persons can have meaningful access to them. The EO also requires Federal agencies work to ensure that recipients of Federal financial assistance provide meaningful access to their LEP applicants and beneficiaries. EPA's Order 1000.32 "Compliance with Executive Order 13166: Improving Access to Services for Persons with Limited English Proficiency"[8] requires that EPA ensure its programs and activities are meaningfully accessible to LEP persons.

EO 13175, "Consultation and Coordination with Indian Tribal Governments" (2000), calls on Federal agencies to have "an accountable process to ensure meaningful and timely input by tribal officials in the development of regulatory policies that have tribal implications." To the extent practicable and permitted by law, if a regulatory action with tribal implications is proposed and imposes substantial direct compliance costs on Indian tribal governments, and is not required by statute, then the agency must either provide funds necessary to pay direct compliance costs of tribal governments or consult with tribal officials early in the process of regulatory development and provide OMB a tribal summary impact statement.

EO 12866, "Regulatory Planning and Review" (1993), allows agencies to consider "distributive impacts" and "equity" when choosing among alternative regulatory approaches, unless prohibited by statute. EO 13563, issued in January 2011, supplements and reaffirms the provisions of EO 12866.

OMB's *Circular A-4* states that regulatory analyses "should provide a separate description of distributional effects (i.e., how both benefits and costs are distributed among populations of particular concern) so that decision makers can properly consider them along with the effects of economic efficiency." It specifically calls for a description of "the magnitude, likelihood, and severity of impacts on particular groups" if the distributional effects are expected to be important (OMB 2003).

The President's memorandum to heads of departments and agencies that accompanied EO 12898 specifically raised the importance of procedures under NEPA for identifying and addressing environmental justice concerns (White House 1994). The memorandum states that "each Federal agency shall analyze the environmental effects, including human health, economic and social effects, of Federal actions, including effects on minority communities and low-income communities when such analysis is required by [NEPA]." The Council on Environmental Quality (CEQ) issued EJ guidance for NEPA in 1997 (CEQ 1997). EPA issued guidance in 1998 for incorporating EJ goals into EPA's preparation of environmental impact statements and environmental assessments under NEPA (U.S. EPA 1998a).

In July 2010, EPA published its *Interim Guidance on Considering Environmental Justice During the Development of an Action* (U.S. EPA 2010a). This guide is designed to help EPA staff incorporate EJ into the rulemaking process, from inception through promulgation and implementation. The guide also provides information on how to screen for EJ effects and directs rulewriters to respond to three basic questions throughout the rulemaking process:

1. How did your public participation process provide transparency and meaningful participation for minority, low-income, indigenous populations, and tribes?

2. How did you identify and address existing and new disproportionate environmental and public health impacts on minority, low-income, and indigenous populations during the rulemaking process?

3. How did actions taken under #1 and #2 impact the outcome or final decision?

8 EPA Order 1000.32 is available at http://www.epa.gov/civilrights/docs/lep_order_1000_32.pdf (accessed on May 28, 2013).

Finally, in September 2010 EPA released its FY2011-2015 Strategic Plan outlining how EPA would achieve its mission to protect human health and the environment over the next five years (U.S. EPA 2010b). Included in the plan is a cross-cutting fundamental strategy to focus on "working for environmental justice and children's health." To implement this strategy, EPA released Plan EJ 2014 in September 2011 that provides a roadmap for the Agency to incorporate environmental justice into policies, programs and activities. One of five cross-agency focus areas identified in Plan EJ 2014 is "Incorporating Environmental Justice into Rulemaking."[9]

Together these documents provide a solid foundation for considering distributional effects for population groups of concern in the rulemaking process.

10.2 Environmental Justice

EPA defines environmental justice as "the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies" (EPA 2010a). EO 12898 specifically states that Federal agencies should "…identify and address…disproportionately high and adverse human health or other environmental effects… on minority populations and low-income populations…" (EPA 2010a).

For policies that strengthen an environmental standard, EPA regulatory analyses have often relied on a default assumption that these policies have no EJ concerns because they reduce overall environmental burdens. However, it is incorrect to conclude that tighter standards necessarily improve environmental quality for everyone. The nuances of a rule could result in negative effects, such as higher emissions in some areas, even though net environmental quality improves. It is also possible that older, more polluting facilities close as a result of a rule and new facilities open in different locations, changing the distribution of emissions across communities.[10] Hence, when data are available, a basic analysis can support conclusions regarding potential distributional effects. In addition, while there may be no adverse environmental impacts, other economic impacts, like costs, could affect population groups of concern disproportionately and may warrant examination.[11]

Distributional analysis also improves transparency of rulemaking and provides decision makers and the public with more complete information about a given policy's potential effects. Such documentation helps EPA and the public track and measure progress in addressing EJ concerns. Analysts play a role in ensuring meaningful involvement by explaining distributional analysis in plain language, including key assumptions, methods, and results, and by asking for information from the public (e.g., asking for comment in the proposed rulemaking) on exposure pathways, end points of concern, and data sources that may improve the distributional analysis.[12] Further guidance on ensuring meaningful engagement of environmental justice stakeholders in the rulemaking process can be found in U.S. EPA (2010a).

10.2.1 Background Literature

The study of economic efficiency (the focus of benefit-cost analysis) of regulatory approaches has a long history in the economics literature, including an established theoretical foundation and generally accepted empirical methodology. But an assessment of distributional consequences

9 Plan EJ 2014 is available at http://www.epa.gov/compliance/ej/resources/policy/plan-ej-2014/plan-ej-2011-09.pdf (accessed on May 9, 2012).

10 U.S. EPA (2010a) provides additional information on how an EJ concern may arise in the context of a rule.

11 See U.S. EPA (2008a) for an example where changes in costs are addressed in an analysis of distributional impacts in the context of EJ.

12 Meaningful involvement is defined by EPA to mean that "1) potentially affected community members have an appropriate opportunity to participate in decisions about a proposed activity that will affect their environment and/or health; 2) the public's contribution can influence the regulatory agency's decision; 3) the concerns of all participants involved will be considered in the decision-making process; and 4) the decision makers seek out and facilitate the involvement of those potentially affected" (U. S. EPA 2010a, U.S. EPA 2012a).

has received relatively less attention.[13] Media and government interest in potential environmental inequity arising from landfill siting decisions in the mid-1980s led to an increased focus in the economics literature on distributional issues in the context of race, poverty, and income.[14] This section provides a brief overview of key studies from the economics and health literature. For a more comprehensive discussion see Ringquist (2005), Banzhaf (2012a), and Banzhaf (2012b).

Studies of EJ can vary by specific pollutant, the proxy used for risk or exposure, geographic area, and time period, making it difficult to directly apply general findings to a particular rulemaking. The literature illustrates, however, that EJ is a potential concern with regard to plant emission decisions and is therefore worthy of analysis in a regulatory context (see, for example, Wolverton 2009). It is important to note that the economics literature typically focuses on addressing the question of whether certain population groups are exposed to greater amounts of pollution. There is also the possibility that some populations are more susceptible to pollution for a given level of exposure and that socioeconomic factors may play a role. While literature addressing this issue is not discussed here, Section 10.2.8.5 of this chapter discusses various risk considerations including susceptibility. In addition, both the EJ literature and this chapter tend to focus on the distribution of physical aspects of environmental outcomes.[15]

Evidence exists of potential disproportionate impacts from environmental stressors on various population groups using a wide variety of proxies for exposure. Many studies are proximity-based: distance to a polluting facility is a surrogate for exposure. These studies often find evidence that locally-unwanted land-uses such as landfills or facilities that treat, store, or dispose of hazardous waste are more likely to be concentrated in predominantly minority or low-income neighborhoods (for example, Bullard 1983; GAO 1983; UCC 1987; Boer et al. 1997; and Mohai et al. 2009).[16]

Other studies attempt to better approximate exposure by examining whether existing emission patterns are related to socio-economic characteristics. These studies often focus on a particular type of pollution and geographic area. They also often differ in how they define the relevant neighborhood and comparison group. As such, results with regard to race and income vary across studies. For example, after controlling for other factors, Hamilton (1993, 1995) finds that expansion decisions for waste sites are unrelated to race and finds mixed evidence for income, while Aurora and Cason (1998) find both race and poverty are positively related to toxicity-weighted Toxic Release Inventory (TRI) emissions, although the significance of these relationships varies by region. Gray and Shadbegian (2004) find poor communities are exposed to more air and water pollution from pulp and paper mills, but find the opposite for minority communities.

Finally, other studies attempt to account for health risks. For example, Rosenbaum et al. (2011) combine information on ambient concentrations of diesel particulate matter in marine harbor areas throughout the United States with exposure and carcinogenic risk factors broken out by race, ethnicity, and income. They find that the most important factor in predicting higher particulate

13 For a discussion of the possible distributional effects of environmental policies with regard to income, see Fullerton (2009).

14 The rise in concern over environmental justice is often traced to demonstrations in Warren County, North Carolina in 1982 over the siting of a polychlorinated biphenyl (PCB) landfill in a poor and minority community.

15 Differences in exposures or health effects alone may not be representative of differences in total benefits and costs. As discussed in Serret and Johnstone (2006) and Fullerton (2011), for example, the full distribution of environmental policy could include differences in product prices, wage rates, employment effects, economic rents, etc. It is likely, however, that the methods used to analyze the full distributional effects (e.g., computable general equilibrium models) are beyond the scope of a typical regulatory analysis and the policy tools to address any resultant distributional concerns (e.g., tax policy and redistribution programs) are beyond the scope of environmental policy.

16 Others note the strength of this contemporaneous relationship but find that the direction and magnitude of the relationship between location and race or income at time of siting is less clear (see Been 1994; Been and Gupta 1997; and Wolverton 2009). See Shadbegian and Wolverton (2010) for a summary of the literature on firm location and environmental justice, including a discussion of whether plant location precedes changes in socioeconomic composition that result in higher percentages of non-white and poor households nearby or vice versa. Most of these studies examine partial correlations between pollution and household characteristics, using statistical techniques that control for other factors.

matter intake fractions (i.e., mass of a pollutant inhaled or ingested divided by mass emitted) is population density and that low-income and minority individuals are over-represented in marine harbor areas that exceed risk thresholds. Likewise, Morello-Frosch et al. (2001) combine estimates of hazardous air pollutant concentrations in southern California with information on lifetime cancer risks by socioeconomic status and race and find that even though lifetime cancer risks are high for all individuals in the study, race and ethnicity are positively related to lifetime cancer risk after controlling for economic and land use variables.

Ringquist (2005) conducts a meta-analysis of both facility location and emissions across 49 studies published prior to 2002 and finds evidence that plant location and higher emissions are more likely to occur in communities with a higher percent non-white population. He finds little evidence, however, that this is the case in communities with lower income or higher poverty rates. The finding for race holds across a wide variety of environmental risks (e.g., hazardous waste sites and air pollution concentrations), levels of aggregation (e.g., zip codes, census tracts, and concentric circles around a facility), and controls (e.g., land value, population density, and percent employed in manufacturing). The finding for race appears sensitive, however, to comparison groups (e.g., all communities versus a subset of communities).

A potential unintended consequence of improving environmental quality in some communities more than others is that rents may increase in the improved neighborhoods, making them potentially unaffordable for poorer households. For example, Grainger (2012) shows that about half of the increases in home prices due to the Clean Air Act Amendments are passed through to renters. Thus, the net health effect of improvements in environmental quality for renters depends on whether or not they move. Those who do not move experience higher rents, but also improved neighborhoods. For those who do move the net effect depends on the quality of the neighborhood to which they relocate. If these households receive far less of the health benefit predicted from a static model and also face transaction costs from moving, they could be worse off. The literature refers to this phenomenon as "environmental gentrification" (see also Banzhaf and McCormick 2012).

Sieg et al. (2004) find that even with no moving costs, local households could be worse off because other households move into the clean neighborhood and bid up the rents.[17] Earlier work by Banzhaf and Walsh (2008) shows that neighborhood income increases following cleanup, but more recent analysis (Banzhaf et al. 2012) shows racial characteristics in the neighborhood may not change. The authors postulate that richer minorities may move back into neighborhoods following cleanup.

10.2.2 Analyzing Distributional Impacts in the Context of Regulatory Analysis

In the context of regulatory analysis, examining distributional effects of health and environmental outcomes or costs can be accomplished, when data are available, by comparing effects in the baseline to post-regulatory scenarios for minority, low-income, or indigenous populations.[18]

When evaluating health and environmental outcomes, the following fundamental questions can guide the process of considering potential analytical methods for assessing EJ.[19]

- What is the baseline distribution of health and environmental outcomes across population groups of concern for pollutants affected by the rulemaking?[20]

17 The market dynamics associated with the relationship between household location decisions and pollution was first examined in a rigorous context in Been and Gupta (2007), and further explored by Banzhaf and Walsh (2008).

18 OMB (2003) defines the baseline as "the best assessment of the way the world would look absent the proposed action." Section 10.2.6 describes the concept of baseline briefly. For a more detailed discussion on properly defining a baseline to measure the incremental effects of regulation, see Chapter 5 of these *Guidelines*.

19 See Maguire and Sheriff (2011) for more detail.

20 The term "outcome" is used to indicate that these questions should be interpreted more broadly than just applying to health effects. EPA Program Offices have the flexibility to adapt the wording of these questions to reflect the realities of the particular endpoints under consideration for a rulemaking.

- What is the distribution of health and environmental outcomes for the options under consideration for the rulemaking effort?

- Under the options being considered, how do the health and environmental outcomes change for population groups of concern?[21]

Note that these analytic questions recommend the analyst provide information on the distribution of outcomes, but do not ask for a determination of whether differences across population groups constitute disproportionate impacts.[22] The term disproportionate is neither defined in EO 12898, nor does the academic literature provide clear guidance on what constitutes a disproportionate impact. The determination of whether an impact is disproportionate is ultimately a policy judgment.

This chapter presents a suite of methods for analyzing distributional effects across a variety of regulatory contexts. Because the data, time, and resource constraints will differ across programs and rules, these guidelines are intended to provide flexibility to the analyst while introducing greater rigor and transparency in how EJ is considered in a regulatory context.

10.2.2.1 Evaluating Changes in the Distribution of Health and Environmental Outcomes

The analysis of EJ should ideally consider how a policy affects the distribution of relevant health and environmental outcomes (e.g., mortality risk from a regulated pollutant). If the outcome data are unavailable, distribution of ambient environmental quality indicators (e.g., pollutant concentrations) can be a useful proxy. Such indicators are less informative than the outcomes themselves if population groups of concern vary in vulnerability to the pollutant, for example.[23] If projecting ambient environmental quality is not feasible, then the analysis may examine the distribution of pollutants from regulated sources. Distribution of pollutants is less desirable than distributions in ambient environmental quality or health and environmental outcomes due to uncertainty regarding how a reduction in emissions from a given source translates into environmental quality and how that, in turn, translates into the human impacts that are the ultimate objective of the analysis.

It is important to consider changes in distributions of health and environmental outcomes between baseline and various policy options, rather than just the distribution of changes since an unequal distribution of environmental improvements may actually help alleviate existing disparities (Maguire and Sheriff 2011). For example, suppose a policy is expected to reduce a pollutant, causing a greater reduction in particular adverse health outcomes for non-minorities than for minorities. One might conclude that this change in the distribution of outcomes could pose an EJ concern. If, however, the non-minority population suffered greater ill effects from the pollutant at baseline than the minority population, such a change in the distribution of outcomes may reduce, rather than increase, a pre-existing disparity in outcomes.

The difference between these two measures — the distribution of change in health and environmental outcomes and the change in the distribution of health and environmental outcomes — has implications for the suitability of data for analysis. In particular, analyzing the distribution of monetized benefits from a benefit-cost analysis can be problematic. Benefit-cost analyses do not estimate each affected individual's monetized welfare at baseline and policy levels of environmental quality. Instead, they

21 It would be useful to quantify the degree to which disparities change from baseline, so that one could rank in order of preference the relative merits of various options. Any ranking metric, however, would require adoption of an implicit social welfare function. Such approaches are analytically meaningful, but still under development and recommendation of a specific social welfare function is beyond the scope of this chapter. Text Box 10.1 provides additional discussion on this topic.

22 The EJ guidance for NEPA (CEQ 1997) provides some guidance on the use of the term. A population group may be disproportionately affected if health effects are significant or "above generally accepted norms," the risk or rate of exposure is significant or "appreciably exceeds or is likely to appreciably exceed the risk or rate to the general population or other appropriate comparison group," or is subject to "cumulative or multiple adverse exposures from environmental hazards."

23 A large epidemiological literature explores differences in health effects across various demographic groups. See, for example, Schwartz et al. (2011b).

estimate society's willingness to pay for a *change* in environmental quality. Thus, although the distribution of this change in welfare across groups may be of interest in its own right, in isolation it does not inform the question of whether the policy increases or reduces pre-existing disparities.

To address the question of how a policy affects disparities it is necessary to evaluate the distribution of environmental and health outcomes in the baseline and for each policy option. As an alternative to the change in willingness to pay one could examine the distribution of physical indicators. Such an evaluation is fairly straightforward if there is only one outcome to consider. Analysis of multiple outcomes (e.g., asthma risk and fatal heart attack risk) raises the problem of whether and how to aggregate these outcomes into a single measure. Combining several outcomes into a single aggregate measure may be desirable, but entails normative value judgments regarding the weight to be given to each component. For example, how much asthma risk is equivalent to a given risk of a fatal heart attack? One possible weighting scheme would be to use quality-adjusted life years (QALYs) or similar measures, but these are generally not consistent with willingness-to-pay measures and benefit-cost analysis (IoM 2006). Another alternative is to use the willingness-to-pay values from the benefit-cost analysis as weights (see Chapter 7 of these *Guidelines* for a discussion of willingness to pay).

A standard benefit-cost analysis aggregates multiple outcomes by multiplying the number of cases of each outcome by its respective *marginal* willingness-to-pay. In principle one could use this weighting scheme in a distributional analysis. There is a theoretical issue, however. The empirical techniques used to monetize health and environmental benefits estimate an individual's marginal willingness to pay for a change in the outcome. That is, they reflect the amount of money an individual would give up for a very small improvement in the outcome variable, evaluated at a particular level. The problem is that economic theory suggests that even if all individuals had identical preferences, the marginal willingness to pay to avoid a bad outcome should increase with the level of the outcome (e.g., an individual would be willing to pay more to reduce her probability of death from a particular disease from 99 percent to 98 percent, than she would to reduce it from 2 percent to 1 percent). As a practical matter, however, marginal willingness-to-pay measures typically used in benefit-cost analysis are constant values. The approximation implicit in this approach is defensible when the changes considered are not too large. However, it is not necessarily reasonable to multiply, say, the baseline mortality risk by the value of a statistical life in order to get the dollar value of eliminating the entire baseline risk. Yet this type of calculation would be necessary in order to evaluate how policy options would change the distribution of monetized environmental outcomes across population groups of concern. Consequently, if analysts use monetized values to aggregate across outcomes, the exposition should include appropriate caveats and be presented alongside outcome-by-outcome levels for the baseline and each policy option.

10.2.2.2 Evaluating the Distribution of Costs

Activities to address environmental justice often focus on reducing disproportionate environmental and health outcomes in communities. However, certain directives (e.g., EO 13175 and OMB *Circular A-4*) specifically identify distribution of economic costs as an important consideration. The economic literature also typically considers both costs and benefits when evaluating distributional consequences of an environmental policy in order to understand their net effects on welfare. For instance, Fullerton (2011) discusses six possible types of distributional effects that may result from an environmental policy: higher product prices, changes in the relative returns to factors of production, how scarcity rents are distributed, the distribution of environmental benefits, transitional effects of the policy, and the capitalization of environmental improvements into asset prices (e.g., land or housing values). Policy decisions involve trade-offs, and these may differ across affected groups. While health or environmental improvements may accrue to certain population groups of concern, costs may be borne by others. As a result, some groups may experience net costs even if everyone is expected to receive gross environmental benefits.

This chapter frames the discussion in terms of environmental and health outcomes (referred to as benefits, when monetized), but many of the methods can be applied to costs and other impacts as well. Whether or not costs are included in an evaluation of EJ issues associated with a regulation should be evaluated on a case-by-case basis. If regulatory costs are spread fairly evenly across many households (e.g., in the form of higher prices) and expected to be small on a per-household basis, further analysis is likely not warranted or feasible. However, there may be cases where the analysis of the distribution of costs is warranted.[24] Such cases may include situations where costs to consumers may be concentrated among particular types of households (e.g., renters); identifiable plant closures or facility relocations that could adversely affect certain communities; or when households may change their behavior in response to the imposition of costs.

In many cases, detailed analyses of costs may be challenging due to data or modeling constraints. For example, EPA may expect air pollution control costs to be passed on to electricity consumers. The Agency might not have information, however, on how costs are passed through as rate increases, how these increases may be broken down between residential and commercial customers, what assistance is available for low-income consumers, and how consumption patterns differ by race and income. Likewise, if air quality improvements associated with a regulation are unevenly distributed, demand for housing in particular neighborhoods may affect rental prices. While hedonic approaches (discussed in Chapter 7) may be useful for demonstrating how changes in environmental quality factor into housing prices, predicting the effect of such price changes on household migration by race or income may be infeasible.[25] Absent such data, it might not be possible to predict the total impact of the rule on different populations. In these instances, those issues that cannot be quantified can be qualitatively discussed.

24 EPA's Lead Renovation, Remodeling, and Painting Final Rule (U.S. EPA 2008c) provides the best example to date of consideration of costs in the context of a rulemaking.

10.2.3 Relevant Populations

EO 12898 identifies a number of relevant population groups of concern: minority populations, low-income populations, Native American populations and tribes, and "populations who principally rely on fish and/or wildlife for subsistence."[26] It may be useful to analyze these categories in combination — for example, low-income minority populations — or to include additional population groups of concern, but such analysis is not a substitute for examining populations explicitly mentioned in the Executive Order. In this section, we discuss existing Federal definitions for population groups of concern in the context of EJ. We also discuss credible options for defining these populations in the absence of a Federal definition.

10.2.3.1 Minority and Native American Populations

OMB (1997) specifies minimum standards for "maintaining, collecting, and presenting data on race and ethnicity for all Federal reporting purposes.... The standards have been developed to provide a common language for uniformity and comparability in the collection and use of data on race and ethnicity by Federal agencies." In particular, it defines the following minimum race and ethnic categories:

- American Indian or Alaska Native
- Asian
- Black or African American

25 See Section 8.2.5.1 of the *Handbook on the Benefits, Costs and Impacts of Land Cleanup and Reuse* (U.S. EPA 2011c) for a more detailed discussion of EJ in the context of the potential effects of environmental policy on land values and household location decisions.

26 EO 12898 clarifies in Section 6 that the EO applies to Native Americans and also Indian Tribes, as specified in 6-606, as well as populations who principally rely on fish and/or wildlife for subsistence as specified in 4-401.

- Native Hawaiian or Other Pacific Islander
- White
- Hispanic or Latino

Statistical data collected by the Federal government, such as the U.S. Census Bureau, use this classification system.[27] Beginning with the 2000 Census, individuals were given the option of selecting more than one race, resulting in 63 different categories. OMB (2000) provides guidance on how to aggregate these data in a way that retains the original minimum race categories (i.e., the first five categories listed above) and four double race categories that are most frequently reported by respondents.[28] In addition, the U.S. Census Bureau collects data useful for identifying minority populations not completely captured by either the race or ethnicity categories, such as households that speak a language other than English at home or foreign-born populations.

CEQ's NEPA Guidance for EJ (CEQ 1997) provides useful direction for defining minority and minority population based on these Federal classifications. Minority is defined as "individual(s) who are members of the following population groups: American Indian or Alaskan Native; Asian or Pacific Islander; Black, not of Hispanic origin; or Hispanic." A population is identified as minority if "either (a) the minority population of the affected area exceeds 50 percent or (b) the 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." The term meaningfully greater is not defined, although the guidance notes that a minority population exists "if there is more than one minority group present and the minority percentage, as calculated by aggregating all minority persons, meets one of the above-stated thresholds." Finally, the CEQ Guidance states that analysts "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 Native Americans), where either type of group experiences common conditions of environmental exposure or effect."

10.2.3.2 Low-Income Populations

OMB has designated the U.S. Census Bureau's annual poverty measure, produced since 1964, as the official metric for program planning and analytic work by all Executive branch agencies in *Statistical Policy Directive No. 14* (Federal Register 1978), although it does not preclude the use of other measures. Many Federal programs use variants of this poverty measure for analytic or policy purposes, and the U.S. Census Bureau publishes data tables with several options.

The U.S. Census Bureau measures poverty by using a set of money income thresholds that vary by family size and composition to determine which households live in poverty. If a family's total income is less than the threshold, then that family and every individual in it is considered in poverty. The official poverty thresholds do not vary geographically, but they are updated for inflation using the national Consumer Price Index for All Urban Consumers (CPI-U). The official poverty definition uses money income before taxes and does not include capital gains or noncash benefits (such as public housing, Medicaid, and food stamps).[29] This measure of poverty has remained essentially unchanged — apart from relatively minor alterations in 1969 and 1981— since its inception.[30]

There is considerable debate regarding this poverty measure's ability to capture differences in

27 Analysts should refer to the OMB Federal Register notice for the specific definitions: http://www.whitehouse.gov/omb/fedreg_1997standards/ (accessed on December 20, 2012).

28 See OMB (2000) for specific guidance on how to conduct this aggregation.

29 See "How the Census Bureau Measures Poverty" available at http://www.census.gov/hhes/www/poverty/about/overview/measure.html (accessed on November 30, 2011).

30 The U.S. Census Bureau produces single-year estimates of median household income and poverty by state and county, and poverty by school district as part of its *Small Area Income and Poverty Estimates*. It also provides estimates of health insurance coverage by state and county as part of its *Small Area Health Insurance Estimates*. These data are broken down by race at the state level and by income categories at the county level.

economic well-being. In particular, the National Research Council (NRC) recommended that the official measure be revised because "it no longer provides an accurate picture of the differences in the extent of economic poverty among population groups or geographic areas of the country, nor an accurate picture of trends over time" (Citro and Michael 1995). OMB convened an interagency group in 2009 to define a supplemental poverty measure based on NRC recommendations. The U.S. Census Bureau released the Supplemental Poverty Measure (SPM) in November 2011 (Short 2011). This measure uses different measurement units to account for "co-resident unrelated children (such as foster children) and any co-habitors and their children," a different poverty threshold, and modified resource measures (to account for in-kind benefits and medical expenses, for example). It also adjusts for differences in housing prices by metropolitan statistical area, as well as family size and composition.

The NRC recognized that annual income is not necessarily the most reliable measure of relative poverty as it does not account for differences in accumulated assets across households. Neither the SPM nor the official U.S. poverty thresholds take into account differences in wealth across families. However, the SPM examines whether a household is likely to fall below a particular poverty threshold as a function of inflows of income and outflows of expenses. The U.S. Census Bureau asserts that this measure is therefore more likely to capture short-term poverty since many assets are not as easily convertible to cash in the short run (Short 2012).

The U.S. Census Bureau also includes several additional measures that may prove useful in characterizing low-income families. Unlike poverty, there is no official or standard definition of what constitutes "low-income," though it is expected to vary similarly by region due to differences in cost-of-living as well as with family composition. It is therefore appropriate to examine several different low-income categories, including families that make some fixed amount above the poverty threshold (e.g., two times the poverty threshold) but still below the average household income for the United States or for a region.

Educational attainment or health insurance coverage may also be useful for characterizing low-income families relative to other populations, although we caution analysts that some measures may be hard to interpret and use in a regulatory context. It is also possible to examine the percent of people who are chronically poor versus those that experience poverty on a more episodic basis using the *Survey of Income and Program Participation* which provides information on labor force participation, income, and health insurance for a representative panel of households on a monthly basis over several years (see Iceland 2003). Finally, cross-tabulations often are available between many of these poverty measures and other socioeconomic characteristics of interest such as race, ethnicity, age, sex, education, and work experience.

10.2.3.3 Populations that Principally Subsist on Fish and Wildlife

EO 12898 directs agencies to analyze populations that principally subsist on fish and wildlife. CEQ's NEPA Guidance for EJ (CEQ 1997) defines subsistence on fish and wildlife as "dependence by a minority population, low-income population, Indian tribe or subgroup of such populations on indigenous fish, vegetation and/or wildlife, as the principal portion of their diet." It also states that differential patterns of subsistence consumption are defined as "differences in rates and/or patterns of subsistence consumption by minority populations, low-income populations, and Indian tribes as compared to rates and patterns of consumption of the general population."

Neither the U.S. Census Bureau nor other Federal statistical agencies collect nationally representative information on household consumption of fish and/or wildlife. However, EPA has conducted consumption surveys in specific geographic areas. If fish and wildlife consumption is a substantial concern for a particular rulemaking, EPA's guidance can provide useful information for collecting these data (see U.S. EPA 1998b). There may also be surveys conducted by state or local governments. It is important to verify that any survey used in an analysis of distributional impacts in the context of EJ adheres to the parameters and methodology set out in U.S. EPA (1998b).

10.2.4 Data Sources

Many data sources can be used for conducting analyses of EJ issues. The U.S. Census Bureau's "Quick Facts" website contains frequently requested Census data for all states, counties, and urban areas with more than 25,000 people.[31] Data include population, percent of population by race and ethnicity, and income (median household income, per-capita income, and percent below poverty line).

In 2010 the U.S. Census Bureau began to administer the decennial Census using a short form to collect basic socioeconomic information. More detailed socioeconomic information is now collected annually by the American Community Survey (ACS), which is sent to a smaller percentage of households than the decennial Census.[32] The ACS provides annual estimates of socioeconomic information for geographic areas with more than 65,000 people, three-year estimates for areas with 20,000 or more people, and five-year estimates for all areas.[33] The five-year estimates, which are based on the largest sample, are the most reliable and are available at the census tract and block group levels. Some of the Quick Facts data include estimates from the ACS.

The U.S. Census Bureau's American Housing Survey (AHS), is a housing unit survey that provides data on a wide range of housing and demographic characteristics, including information on renters.[34] Unlike the ACS, which selects a random sample every year, the AHS returns to the same 50,000 to 60,000 housing units every two years.

31 Quick Facts is available at: http://quickfacts.census.gov/qfd/index.html. The year associated with data from Quick Facts is important to note. Data are updated as new information becomes available. Therefore, not all data elements represent the same year.

32 The ACS is available at: http://www.census.gov/acs/www/index.html. (accessed December 1, 2011.)

33 Because ACS variables change over time, caution should be used when comparing ACS estimates across samples and years. Guidance for comparing ACS data can be found at: http://www.census.gov/acs/www/guidance_for_data_users/comparing_data/ (accessed on April 27, 2011).

10.2.5 Scope and Geographic Considerations

Most EPA rules are national in scope. Therefore, the entire country is typically considered within the scope of analysis. However, there may be reasons to consider a rule's distributional effects at a sub-national level. For example, for a regulation of hazardous waste sites it may be appropriate to conduct separate state-level analyses due to differences in implementation of state-level regulations. A rule may also affect a limited part of the country. The 2011 Cross-State Air Pollution Rule (U.S. EPA 2011b), for example affects mainly eastern states.[35] In such cases the analyst may wish to evaluate the effects of the regulation at a regional level. Finally, for some regulations, such as those governing the use of a household chemical or as a product ingredient, geography may not be as relevant for determining how health and environmental outcomes vary across population groups of concern. Two main issues to consider when comparing impacts of a rulemaking on minority, low-income, or indigenous populations across geographic areas are:

- Unit of analysis (e.g., facilities or aggregate emissions to which a population group is exposed within a designated geographic area); and
- Geographic area of analysis used to characterize impacts (e.g., county or census tract).[36]

The unit of analysis refers to how the environmental harm is characterized. For instance, in a proximity-based analysis the unit of analysis could be an individual facility or the

34 Information on owner-occupied homes versus renters may be useful when exploring issues of gentrification, where renters could be worse off due to rising housing costs.

35 See http://www.epa.gov/airtransport/ for details. (accessed December 1, 2011.)

36 This is often referred to in the literature as geographic scale.

total number of facilities within a particular geographic area (e.g., a county or census tract). In an exposure-based analysis the unit of analysis could be emissions aggregated within a particular geographic area to which the population is exposed. The unit of analysis is often identical to the geographic scale used to aggregate and compare effects on minority, low-income, or indigenous populations in one area to another (see Section 10.2.7 regarding how to select an appropriate comparison group).[37] The choice will vary depending on the nature of the pollutant (e.g., point sources may use a facility as the unit of analysis, while area sources may use a geographic unit). In considering various units, an important consideration is whether the data are sufficiently disaggregated to pick up potential variation in impacts across socioeconomic characteristics. More aggregated units of analysis (e.g., metropolitan statistical area (MSA) or county) may mask variation in impacts across socioeconomic groups compared to more disaggregated levels (e.g., facility or census tract).

The geographic area of analysis is the area used to characterize impacts (e.g., distance around a facility). Outcomes are aggregated by population groups within geographic areas to compare across groups. As with unit of analysis, choice of options for defining the geographic area will vary depending on pollutant and rule. Some air pollutants, for example, may travel hundreds of miles away from the source, making it appropriate to choose a large area for measuring impacts. In contrast, water pollutants or waste facilities may affect smaller areas, making it appropriate to consider a smaller area for analysis. Likewise, an assessment of outcomes from specific industrial point sources may require more spatially resolved air quality, demographic and health data than one that affects regional air quality, where coarser air quality, demographic and health data may suffice. Using more than one geographic area of analysis to compare effects across population groups may also be useful since outcomes are unlikely to be neatly contained within geographic boundaries. The literature has demonstrated that results are sensitive to the choice of the geographic area of analysis (Mohai and Bryant 1992; Baden et al. 2007).

37 In Fowlie et al. (2012), for example, the scale of the analysis varies between 0.5, 1 and 2 miles of the facility (which is the unit of analysis).

Commonly used geographic areas of analysis include:

Counties: The United States has more than 3,000 counties according to the 2007 Census of Governments. Although counties are well-defined units of local government and provide complete coverage of the United States, they vary in size from a few to thousands of square miles and population density ranges from less than one person per square mile in some Alaskan counties to over 66,000 in New York County. In addition, spatial considerations associated with using counties present concerns for an analysis of distributional impacts in the context of EJ. A facility located in one corner of a county may have greater effects on neighboring counties than on residents of the county where the plant is located.[38, 39]

Metropolitan and Micropolitan Statistical Areas: The U.S. Census Bureau publishes data on metropolitan and micropolitan statistical areas, as defined by OMB (OMB 2009). Metropolitan statistical areas include an urban core and adjacent counties that are highly integrated with the urban core. A micropolitan statistical area corresponds to the concept of a metropolitan statistical area but on a smaller scale. Metropolitan statistical areas have an urban core of at least 50,000 persons; micropolitan statistical areas have an urban core population between 10,000 and 50,000 persons. Rural areas of the United States are not covered by these statistical designations, though according to the U.S. Census Bureau, almost 94 percent of the U.S. population lived in a metro- or micropolitan statistical area in 2010.

Zip codes: Zip codes are defined by the U.S. Post Office for purposes of mail delivery and may change over time. They also may cross state, county, and other more disaggregated Census

38 These same advantages and disadvantages can apply to other units of government.

39 For criteria pollutants, baseline health data may be available at the county level (e.g., baseline death rates, hospital admissions, and emergency department visits).

statistical area definitions, making them difficult to use for analysis. Zip code tabulation areas are statistical designations first developed by the U.S. Census Bureau in 2000 to approximate the zip code using available census block level data on population and housing characteristics. Data are readily available for the approximately 33,000 U.S. zip code tabulation areas. While smaller than counties, they also vary greatly in size and population. As a result, they may often be less preferable than other geographic areas for analyzing distributional effects across population groups of concern.

Census tracts/block groups/blocks: Census tracts are small statistical subdivisions of a county, typically containing from 1,500 to 8,000 persons. The area encompassed within a census tract may vary widely, depending on population density. Census tracts in denser areas cover smaller geographic areas, while those in less dense areas cover larger geographic areas. Census tract boundaries were intended to remain relatively fixed. However, they are divided or aggregated to reflect changes in population growth within an area over time. Although they were initially designed to be homogeneous with respect to population characteristics, economic status, and living conditions, they may have become less so over time as demographics have changed.

Analysts may also choose to use census blocks or block groups. A census block is a subdivision of a census tract and the smallest geographic unit for which the U.S. Census Bureau tabulates data, containing from 0 to 600 persons. Many blocks correspond to individual city blocks bounded by streets, but may include many square miles, especially in rural areas. And census blocks may have boundaries that are not streets, such as railroads, mountains or water bodies. The U.S. Census Bureau established blocks covering the entire nation for the first time in 1990. Census block groups are a combination of blocks that are within — and a subdivision of — a given census tract. Block groups typically contain 600 to 3,000 persons.[40]

40 Other Census statistical area definitions (e.g., public use microdata areas or PUMAs) are also available.

GIS methods: Because Census-based definitions often reflect topographical features such as rivers, highways, and railroads, they may exclude affected populations that, although separated by some physical feature, receive a large portion of the adverse impacts being evaluated. Since Census-based definitions vary in geographic size due to differences in population density, Geographic Information System (GIS) software and methods may enable the use of spatial buffers around an emissions source that are more uniform in size and easier to customize to reflect the appropriate scale and characteristics of emissions being analyzed for a given rulemaking.

Analysts should be aware that there are a number of challenges typical of working with geospatial data. In some cases, statistical techniques rely on assumptions that often are violated by these types of data (Chakraborty and Maantay 2011). For instance, spatial autocorrelation — when locations in closer proximity are more highly correlated than those further away from each other — violates the assumption that error terms are independently distributed (an assumption that underlies ordinary least squares).

10.2.6 Defining the Baseline

Proper definition of the baseline is crucial for evaluating a rule's distributional effects. OMB (2003) defines the baseline as "the best assessment of the way the world would look absent the proposed action." The baseline allows one to determine how a rule's effects are distributed across population groups of concern and to assess whether some groups may be disproportionately affected. Baseline assumptions used in a distributional analysis should be consistent with those used in the benefit-cost analysis. See Chapter 5 for a more detailed discussion of baseline issues.

10.2.7 Comparison groups

The choice of a relevant comparison group is important for evaluating changes in health, risk, or exposure effects across population groups of concern relative to a baseline. Within-group comparisons involve comparing effects on the same demographic group across different areas in the state, region or nation, while across-group comparisons examine effects for different socioeconomic groups within an affected area. From the perspective of EO 12898, across-group comparisons may be most relevant. The literature suggests using more than one comparison group to analyze whether a finding of disproportionate impacts is sensitive to how it is defined. Bowen (2001) also argues that restricting the comparison group to alternative locations within the same metropolitan area may be more defensible than a national level comparison in some instances, given heterogeneity across geographic regions in industrial development and economic growth over time and inherent differences in socioeconomic composition (e.g., relatively more Hispanics reside in the Southwest). Ringquist (2005), however, notes that placing restrictions on comparison groups in this way may "reduce the power of statistical tests by reducing sample sizes" or bias results against a finding of disproportionate impacts because such restrictions reduce variation in socioeconomic variables of interest.

10.2.8 Measuring and estimating impacts

This section presents a range of potentially useful approaches for describing distributions in regulatory analysis. To the extent feasible, basic summary statistics of a regulation's impacts on relevant endpoints by race and income are recommended for distributional analyses. Summary statistics may be straightforward to calculate when data are available, and providing such information promotes consistency across EPA analytical efforts. A related document, the Interim Guidance on *Considering Environmental Justice During the Development of an Action* (U.S. EPA 2010a), suggests conducting a screening process for determining when an action may require evaluation. For economically significant actions, it is recommended that the results of the screening be demonstrated through the use of summary statistics. Summary statistics can be supplemented with other approaches described below when a screening analysis indicates that a more careful evaluation is needed.

The health effects of exposure to pollution may vary across populations (likewise, with costs). One way to capture these effects is to use information regarding variation in risk and incidence by groups, when available, to characterize the baseline and projected response to a change in exposure (for example, see Fann et al. 2011). However, available scientific literature and data (which also often requires some level of spatial resolution) may not allow for a full characterization. In these cases, it is recommended that the analyst qualitatively discuss conditions that are not adequately accounted for in the risk and exposure characterization used to assess health effects for minority populations or low-income populations and the key sources of uncertainty highlighted in the literature (U.S. EPA 2010a). When data are available to approximate risk or exposure, for instance location of emitting facilities, some level of quantitative analysis may be possible.

Text Box 10.1 discusses the potential usefulness of social welfare functions and inequality indices for ranking distributions. While these methods are useful for combining efficiency and equity considerations into one measure, these tools are not sufficiently developed for application to regulatory analysis. For a more detailed discussion of the advantages and disadvantages of methods commonly used to rank environmental outcomes see Maguire and Sheriff (2011).

10.2.8.1 Simple Summary Statistics

Simple summary measures can characterize potential differences in baseline and regulatory options within and across populations of concern relative to appropriate comparison groups. Such statistics can be calculated, if data are available, to address the three questions outlined in Section 10.2.2. It is important to note, however, that summary statistics alone do not necessarily provide a complete description of differences across groups. Omitted variables are one important limitation of examining single statistics. In addition, summary statistics (e.g., means) can mask important details about the tails of the distribution which can be important for identifying potential EJ concerns

Text Box 10.1 - Social Welfare Functions and Inequality Indices

The costs, benefits, and distributional effects of a regulation can be evaluated by a single social welfare function (SWF). A SWF provides a way to aggregate welfare or utility across individuals into a single value, thus allowing simple, direct comparisons in ranking alternative allocations. Such comparisons are potentially useful in evaluating whether a change from the baseline to a regulatory option makes society better off. Likewise, they can also facilitate comparisons between possible regulatory options (see Adler 2008, 2012 for a discussion). Sen (1970), Arrow (1977), and Just et al. (2004) provide theoretical discussions of SWFs, and Norland and Ninassi (1998) provide an example of an application to energy markets. Adler (2012) addresses practical issues of incorporating both health and income effects in a SWF.

Any ranking of alternative outcomes uses an implicit set of normative criteria; a SWF makes the criteria explicit regarding how society prefers to distribute resources across individuals. Since there is no consensus regarding those preferences, a universally-accepted SWF does not exist. For example, suppose an increase in exposure to a particular pollutant results in an average loss of 0.1 IQ points across a population of 1,000 children (100 IQ points total). It is not obvious how society should rank alternative distributions of this loss. Is it worse to have 250 individuals suffer a loss of 0.1 each, 250 suffer a 0.3 loss, and 500 suffer no loss? Or 500 individuals suffer a loss of 0.01 and 500 suffer a loss of 0.19? Many sensible SWFs could be specified; some may prefer the first outcome, some may prefer the second, and some may be indifferent between the two.

An inequality index is a related concept used to assign a numerical value to distributions of a single "good" or "bad" (e.g., income or pollution), independent of the total amount produced. A distribution with a higher index value is less "equal" than one with a lower number. Commonly used indices are based on simple SWFs and are subject to the same limitations (Blackorby and Donaldson 1978, 1980). However, unlike a SWF, an index number value has cardinal significance, i.e., the magnitudes, not just the rankings, contain information about how much society would be willing to give up in exchange for the rest to be equally distributed.

Inequality indices were originally developed for ranking "goods," like income. In general, it is inappropriate simply to use positive values of a bad outcome (e.g., pollution exposure) in the formula for an index, since doing so would imply that the underlying SWF is increasing in pollution, i.e., it would rank scenarios with higher overall pollution as more desirable. Since indices cannot accommodate negative values, some commonly used income inequality measures, such as the Gini coefficient, and Atkinson index, are inappropriate for evaluating distributions of adverse outcomes. The Kolm index (Kolm 1976a, 1976b), in contrast, does not suffer from this problem (see Maguire and Sheriff 2011). Given that the peer-reviewed literature does not yet contain environmental applications of the Kolm Index, and the Atkinson Index is undefined for "bads," we do not recommend inequality indices be used in regulatory analysis of distributional impacts in the context of EJ at this time.

(see Gochfeld and Burger 2011). Nonetheless, such information can provide useful information on potential differences.

After reviewing the available data and feasible methods for developing information on potential differences, the analyst should present information in a transparent and accessible manner such that the decision maker can consider:

- Population groups of concern for the regulatory action,
- Geographic scale and unit of analysis, when relevant,
- Primary conclusions (e.g., statistical differences),
- Sources of uncertainty across alternative results (e.g., comparison groups and geographic scale), and
- Data quality and limitations of the results.

A variety of measures can be used to characterize an action's distributional effects for population groups of concern.

Means and quantiles

Reporting mean outcomes by group at the baseline and for each regulatory option is a straightforward way to display information. Tests for statistical significance across means provide additional information about differences (see Been and Gupta 1997 and Wolverton 2009). However, mean estimates can mask what might be important information in the tails of the distribution. For example, the baseline outcomes could be uniformly distributed across the population but concentrated around the mean for the regulatory scenario. Examining differences around the central tendency only would not reveal this information. Presenting data using different quantiles can provide additional information illuminating these effects.

Ratios

A simple ratio can be calculated to determine whether certain groups are relatively more exposed to an environmental hazard. For instance, the probability that an individual is minority conditional on being exposed can be divided by the probability that an individual is not minority conditional on being exposed. Alternatively, one can also create a ratio of the probability that an individual is exposed to an environmental risk conditional on being minority divided by the probability that an individual is not exposed conditional on being in the same demographic group. Because ratios may mask absolute differences, ratios should be used in conjunction with other statistics. For example, a ratio may show a 100-fold difference between two groups' exposure to an environmental hazard but the absolute difference could be small. Ratios may exaggerate the importance of differences.

Tests for Differences

Statistical tests can determine whether a significant disparity exists across demographic groups. One of the simplest is a *t*-test of the difference in means. However, a *t*-test assumes a normal distribution so it would be inappropriate for non-normal distributions. For non-normal distributions, nonparametric methods may be used. In cases where comparisons are made based on the difference in probabilities between two groups, tests such as the Kendall test and the Fisher Exact test (for small samples) may be used. These tests compare standard errors of two separate and independent statistics to determine how likely it is that the calculated distribution is the actual one. More sophisticated tests are needed when making comparisons across more than two groups or a more formal examination of the full distribution is desired.

Correlation coefficients

Simple pair-wise correlations between impacts and relevant demographic groups may be useful information for characterizing distributional effects (e.g., Brajer and Hall 2005). It is important to note, however, that the value of a Pearson correlation coefficient, for example, is a measure of how closely the distribution of the relationship between two variables (e.g., percent minority population and ambient pollution concentrations) can be represented by a straight line. It does not provide information regarding the slope of the line, apart from being positive or negative. Similarly, a Spearman rank correlation coefficient measures how closely the relationship can be captured by a generic monotonically increasing or decreasing function. Determination of what constitutes a "strong" or "weak" correlation is somewhat arbitrary, and caution should be used when comparing coefficients across socio-economic variables of interest.

Counts

A count of geographic areas (e.g., counties) where the incidence of an environmental outcome affected by a rule, disaggregated by race/ethnicity and income, exceeds the overall average is a useful measure. For comparison, this count should be accompanied by a count of geographic areas where the incidence does not exceed the overall average.

These counts do not account for magnitude of differences, but can help identify the need for more detailed analysis.

10.2.8.2 Visual Displays

Maps, charts, graphs, and other visual displays are commonly used in EJ analyses (see Shadbegian et al. 2007, for example). With increased access to GIS software and built-in graphical functions in spreadsheet or statistical software, it is relatively easy to produce a variety of visual displays of EJ-related information. Visual displays can be helpful in displaying baseline levels of pollutants or locations of certain facilities, and the distribution, demographic profile and baseline health status of population groups of concern.

There are several challenges with GIS analysis of distributional information. These include spatial and data deficiencies as well as geographic considerations that can lead to misleading or inaccurate results.[41] It may be difficult to discern differences that arise between baseline and regulatory options, unless such differences are stark. While the use of visual displays in an analysis of distributional impacts in the context of EJ may be useful for helping to communicate the geographic distribution of impacts, this information may be more effective if it is accompanied by other analytical information.

10.2.8.3 Proximity-Based Analysis

Proximity- or distance-based analysis is an approach commonly used in the EJ literature as a surrogate for more direct measures of risk or exposure when such information is not easily available. This approach examines demographic and socioeconomic characteristics in proximity to a particular location, typically a waste site, permitted facility, or some other polluting source (for instance, see Baden and Coursey 2002, Cameron et al. 2012, and Wolverton 2009). While a simplistic approach is to examine the population within a Census-defined geographic boundary of a location, it is also possible to use GIS methods to draw a concentric buffer around an emission source, such as a one mile radius around a site to approximate the distance that a particular pollutant may travel. In some cases, it may also be possible to use dispersion models to select a buffer that approximates the effect of atmospheric conditions (for instance, wind direction and weather patterns) on exposure, though these types of models are data-intensive (Chakraborty and Maantay 2011).

Several analytical considerations are important for conducting a proximity-based analysis.[42] First, accurate information is needed for the location of polluting sources. Addresses or latitude/longitude coordinates must reflect physical locations of polluting facilities, and not the location of a headquarters building, for example. Second, a decision must be made regarding the appropriate distance from the facility to examine community characteristics. A solid waste facility with strict monitoring and safety controls is likely to have a limited geographic impact, whereas a permitted air pollution source may have the potential for a more widespread geographic impact. In general, Census-defined geographic boundaries (e.g., county, MSA) are unlikely to provide an accurate portrayal of the relevant affected population because emission sources are often not found in the center of the area (i.e., they are sometimes along a boundary and thus mostly affect a neighboring jurisdiction) and pollutant exposures do not conform to these boundaries.[43] In addition, Census-defined areas often vary widely in size, implying that they may differ in how well they proxy for actual exposure. Defining proximity or distance using buffer-based approaches (e.g., through GIS or fate and transport modeling) around an emissions source has the potential to more closely approximate actual risk and exposure, but the appropriate distance measure can vary by situation. The literature has demonstrated that results in proximity-based analyses can vary substantially with the choice

41 See Chakraborty and Maantay (2011) for further discussion of the limitations of using GIS for EJ analyses.

42 For an overview of proximity analysis, including a discussion of various spatial analysis techniques used in the literature see Chakraborty and Maantay (2011) and Mohai and Saha (2007).

43 Mohai and Saha (2007) refer to this as the "unit-hazard coincidence" approach because the analyst uses the available geographic units and determines whether they are coincident with an environmental hazard instead of first identifying the exact location of the hazard and then examining effects within a particular distance.

of the geographic area of analysis (see Rinquist 2005; Mohai and Saha 2007). For this reason, it is recommended that the analyst explore the potential value of defining and applying more than one specification for distance or proximity.[44]

When this approach is used, it is important to be aware of biases and limitations introduced when proximity or distance is used as a substitute for risk and exposure modeling and that these limitations be clearly discussed (see Chakraborty and Maantay 2011). In particular, it may only be possible to make limited observations with regard to the possibility of disproportionate impacts based on proximity-based analysis alone.

10.2.8.4 Exposure Assessment

Spatial patterns associated with environmental burdens across individuals or communities are difficult to analyze when pollution is diffuse. Air and water pollution, for example, are typically dispersed widely and subject to atmospheric or geologic features. As such, identifying the "proximity" to the hazards via some type of GIS analysis, as described above, is less useful. However, monitoring and/or modeling data may generate distributional effects at a disaggregated level.

Criteria air pollutants (i.e., carbon monoxide, lead, nitrogen dioxide, ozone, particulate matter and sulfur dioxide) are monitored nationally. EPA's National Air Toxics Assessment (NATA) data provide an assessment of hazardous air pollutants across the U.S. at the census tract level.[45] Data from these monitoring networks may potentially be combined with demographic data and dispersion models to generate baseline and regulatory distributions of pollutants by population groups of concern.[46]

44 The analysis of distributional impacts in the context of EJ completed for EPA's proposed Definition of Solid Waste is an example of this type of analysis in a rule-making context. See EPA's Draft Environmental Justice Methodology for the Definition of Solid Waste Final Rule, January 13, 2009, available at: http://www.epa.gov/epawaste/hazard/dsw/ej-meth.pdf (accessed on December 1, 2011).

45 See Apelberg et al. (2005) for an application to Maryland and Morello-Frosch et al. (2002) for an application to southern California.

46 See, for example, U.S. EPA (2011b), Fann et al. (2011), and Post et al. (2011).

While this approach is promising due to spatial detail associated with monitoring data, it is currently only available for certain air pollutants. In addition, it is important to note that monitoring data measure emissions, not individual exposures or health effects associated with the pollutant under consideration. As such, these data are a proxy for actual effects associated with a particular regulation. Further, all individuals within a grid cell are assigned the same emissions (or concentrations based on air quality modeling). Actual exposures or health effects may differ across individuals for a variety of reasons discussed throughout this chapter.

10.2.8.5 Risk Considerations

Certain factors make some populations more susceptible (i.e., experience a greater biological response to a specific exposure) to a particular environmental stressor (see Adler and Rehkopf 2008, Sacks et al. 2011 and Schwartz et al. 2011a).[47, 48] These factors can be genetic or physiological (such as sex and age). They may also be acquired due to variation in factors such as health-care access, nutrition, fitness, stress, housing quality, other pollutant exposures, or drug and alcohol use.[49] For instance, many populations face exposures from multiple pollutants or exposures that have accumulated in ways that may affect their susceptibility to a particular pollutant and introduce complex considerations when attempting to address EJ concerns.[50]

47 A special issue of the *American Journal of Public Health* (Volume 101, Issue S1, December 2011) provides a set of papers exploring these and other issues.

48 EPA's Integrated Risk Information System (IRIS) defines susceptibility as "increased likelihood of an adverse effect, often discussed in terms of relationship to a factor that can be used to describe a human subpopulation (e.g., life stage, demographic feature, or genetic characteristic)." See http://www.epa.gov/iris/help_gloss.htm#s (accessed on December 1, 2011).

49 Sexton (1997) suggests that low-income families may be more susceptible to environmental stressors due to differences in quality of life and lifestyle. Centers for Disease Control data show higher incidences of asthma-related emergency room visits and asthma-related deaths among African-American populations. See http://minorityhealth.hhs.gov/templates/content.aspx?ID=6170 (accessed December 1, 2011).

50 EPA's *Framework for Cumulative Risk Assessment* may serve as a useful reference when assessing how prior exposures may affect the impacts of emission changes from the rule being analyzed, available at http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=36941 (accessed November 2, 2010).

In addition, activities linked to a specific cultural background or socioeconomic status could expose populations to higher levels of pollution. For example, some indigenous peoples and immigrant populations rely on subsistence fishing which could result in higher mercury levels from consumption of fish or expose these populations to other forms of pollution if fishing occurs in contaminated waters (see Donatuto and Harper 2008).[51]

10.3 Children's Environmental Health

Distributional analysis may shed light on differential effects of regulation on children, a lifestage-defined group characterized by a multitude of unique behavioral, physiological, and anatomical attributes. There are two sets of important differences between children and adults regarding health benefits. First, there are differences in exposure to pollutants and in the nature and magnitude of health effects resulting from the exposure. Children may be more vulnerable to environmental exposures than adults because their bodily systems are still developing; they eat, drink, and breathe more in proportion to their body size; their metabolism may be significantly different — especially shortly after birth; and their behavior can expose them more to chemicals and organisms (e.g., crawling leads to greater contact with contaminated surfaces while hand-to-mouth and object-to-mouth contact is much greater for toddler age children). Second, individuals may systematically place a different economic value on reducing health risks to children than on reducing such risks to adults (U.S. EPA 2003).

EO 13045 requires that each federal agency address disproportionate health risks to children. In addition, EPA's Children's Health Policy requires the Agency "consider the risks to infants and children consistently and explicitly as a part of risk assessments generated during its decision making process, including the setting of standards to protect public health and the environment."[52]

Generally, many approaches described earlier in this chapter to characterize the distribution of impacts may be adapted to evaluate children's environmental health risks.[53] For example, when proximity-based analysis is appropriate for evaluating environmental justice impacts, it might also be used to examine whether children are disproportionately located near facilities of concern. In such a case, the considerations described earlier about geography, defining the baseline and comparison groups, and use of summary statistics would all apply.

10.3.1 Childhood as a Lifestage

Evaluating distributional impacts of regulatory actions on children differs in an important way from evaluating the same impacts on population groups of concern for EJ. When EPA evaluates disproportionate health risk impacts from environmental contaminants, it views childhood as a sequence of lifestages from conception through fetal development, infancy, and adolescence, rather than a distinct "subpopulation."

Use of the term "subpopulation" is ingrained in both EPA's past practices as well as various laws that EPA administers such as the Safe Drinking Water Act Amendments. Prior to publication of revised risk assessment guidelines in 2005,[54] EPA described all groups of individuals as "subpopulations." In the 2005 guidelines, the Agency recognizes the importance of distinguishing between groups that form a relatively fixed portion of the population, such as those described Section 3 of this document, and

51 It is also worth considering conditions that reduce a community's ability to participate fully in the decision-making process such as time and resource constraints, lack of trust, lack of information, language barriers, and difficulty in accessing and understanding complex scientific, technical, and legal resources (see Dietz and Stern 2008).

52 See http://yosemite.epa.gov/ochp/ochpweb.nsf/content/policy-eval_risks_children.htm (accessed on December 1, 2011).

53 In principle there is a potential distinction in distributional analysis to be made between factors that are fixed, such as race and sex, and those defined by lifestages. The latter raises the possibility, at least, of examining distribution concerns through the lens of differences in lifetime utility or well-being rather than focusing on a single lifestage. See Adler (2008) for one proposal consistent with this approach.

54 See http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=55907 (accessed on December 1, 2011).

lifestages or age groups that are dynamic groups drawing from the entire population.

The term "lifestage" refers to a distinguishable time frame in an individual's life characterized by unique and relatively stable behavioral and/or physiological characteristics associated with development and growth. Thus, since 2005 EPA characterizes childhood as a sequence of lifestages.[55]

10.3.2 Analytical Considerations

Assessing distributional consequences of policies that affect children's health requires considerations that span risk assessment, action development, and economic analysis. In each case there are existing Agency documents that can assist in the evaluation.

10.3.2.1 Risk Assessment

Effects of pollution can differ depending upon age of childhood exposure. Analysis of disproportionate impacts to children or from childhood lifestages begins with health risk assessment, but also includes exposure assessment. Many risk guidance and related documents address how to consider children and childhood lifestages in risk assessment.

A general approach to considering children and childhood lifestages in risk assessment is found in *A Framework for Assessing Health Risks of Environmental Exposures to Children* (U.S. EPA 2006a). The framework identifies existing guidance, guidelines and policy papers that relate to children's health risk assessment. It emphasizes the importance of an iterative approach between hazard, dose response, and exposure analyses. In addition, it includes a discussion of principles for weight of evidence consideration across life stages.

EPA's 2005 *Cancer Guidelines* (U.S. EPA 2005a) explicitly call for consideration of possible sensitive subpopulations and/or lifestages such as childhood. The *Cancer Guidelines* were augmented by *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens*.[56] Recommendations from this supplement include calculating risks utilizing lifestage-specific potency adjustments in addition to lifestage-specific exposure values which should be considered for all risk assessments.

EPA's *Child-Specific Exposures Handbook* (U.S. EPA 2008b)[57] and *Highlights of the Child-Specific Exposure Factors Handbook* (U.S. EPA 2009a)[58] help risk assessors understand children's exposure to pollution. The handbook provides important information for answering questions about lifestage specific exposure through drinking, breathing, and eating. EPA's guidance to scientists on selecting age groups to consider when assessing childhood exposure and potential dose to environmental contaminants is identified in *Guidance on Selecting Age Groups for Monitoring and Assessing Childhood Exposures to Environmental Contaminants* (U.S. EPA 2005c).

10.3.2.2 Action Development

Disproportionate impacts during fetal development and childhood are considered in EPA guidance on action development, particularly the *Guide to Considering Children's Health When Developing EPA Actions: Implementing Executive Order 13045 and EPA's Policy on Evaluating Health Risks to Children* (U.S. EPA 2006b). The guide helps determine whether EO 13045 and/or EPA's Children's Health Policy applies to an EPA action and, if so, how to implement the Executive Order and/or EPA's Policy. The guide clearly integrates EPA's Policy on Children's Health with the Action Development Process and provides an updated listing of additional guidance documents.

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55 The 2005 Risk Assessment Guidelines "view childhood as a sequence of lifestages rather than viewing children as a subpopulation, the distinction being that a subpopulation refers to a portion of the population, whereas a lifestage is inclusive of the entire population." (U.S. EPA 2005, p 1-15).

56 Available at http://www.epa.gov/cancerguidelines/guidelines-carcinogen-supplement.htm (accessed on December 1, 2011).

57 Available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=199243 (accessed on December 1, 2011).

58 Available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=200445 (accessed on December 1, 2011).

10.3.2.3 Economic Analysis

While these *Economic Guidelines* provide general information on benefit-cost analyses of policies and programs, many issues concerning valuation of health benefits accruing to children are not covered. Information provided in the *Children's Health Valuation Handbook* (U.S. EPA 2003), when used in conjunction with the *Guidelines*, allows analysts to characterize benefits and impacts of Agency policies and programs that affect children.

The *Handbook* is a reference tool for analysts conducting economic analyses of EPA policies when those policies are expected to affect risks to children's health. A major emphasis of the *Handbook* is ensuring that a regulation or policy's economic impacts on children are fully considered in supporting analyses. This analysis includes incorporating children's health considerations in an assessment of efficiency, as well as in any distributional analysis focused on children. Decision makers may also find it useful to have information on a policy's specific impact on children's health, regardless of whether the impact heavily influences overall benefit-cost analysis.

Economic factors may also play a role in other analyses that evaluate children's environmental health impacts. For example, if a higher proportion of children live in poverty, the ability of households with children to undertake averting behaviors might be compromised. This type of information could inform the exposure assessment.

10.3.3 Intersection Between Environmental Justice and Children's Health

The burden of health problems and environmental exposures is often borne disproportionately by children from low-income communities and minority communities (e.g., Israel et al. 2005; Lanphear et al. 1996; Mielke et al. 1999; Pastor et al. 2006).

The challenge for EPA is to integrate both environmental justice and lifestage susceptibility considerations for children where appropriate when conducting distributional analysis. This is especially true when short-term exposure to environmental contaminants such as lead or mercury early in life can lead to life-long health consequences.

10.4 Other Distributional Considerations

10.4.1 Elderly

Another important lifestage to consider is that of the elderly.[59] While there are no standard procedures for including the elderly in a distributional analysis, EPA stresses the importance of addressing environmental issues that may adversely impact them. Most of the Agency's work in this area has been related to risk and exposure assessment.

Older adults may be more susceptible to adverse effects of environmental contaminants due to differential exposures arising from physiological and behavioral changes with age, disease status, drug interactions, as well as the body's decreased capacity to defend against toxic stressors. These considerations are highlighted in EPA's *Exposure Factors Handbook* (U.S. EPA 2011d) and have led EPA's Office of Research and Development to consider an exposure factors handbook specifically for the aging (see U.S. EPA 2007). Additionally, the toxicokinetic and toxicodynamic impacts of environmental agents in older adults have been considered in EPA's document entitled *Aging and Toxic Response: Issues Relevant to Risk Assessment* (U.S. EPA 2005b).[60]

10.4.2 Intergenerational Impacts

Concern for intergenerational impacts arises when those affected by a policy are not yet alive when the policy is developed. If a policy's benefits, costs, and impacts primarily fall upon the current

59 There is a lack of broad agreement about the beginning of the "elderly" lifestage. The U.S. and other countries typically define this lifestage to begin at the traditional retirement age of 65, but, for example, the U.N. defines "elderly" to begin at age 60 (U.S. EPA 2005b).

60 Available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=156648 (accessed on December 1, 2011).

generation, or if policy decisions are reversible within this time frame, there is little need for explicit consideration of intergenerational impacts. However, in other cases, benefits and/or costs of the policy will be borne by future generations, and it is important to consider impacts on these generations. One such case would be policies to reduce greenhouse gases, which are expected to result in benefits related to reduced changes in climate for future generations. Other examples may relate to toxic chemical exposures. Exposures to parents prior to their child's conception can result in adverse health effects in the child, including effects that may not become apparent until the child reaches adulthood.[61]

Assessing intergenerational impacts can be related to the social welfare function approach, described in Text Box 10.1 of this chapter, and to social discounting. In both cases, normative judgments need to be made about which there is no consensus. Under the Ramsey approach to intergenerational discounting, this judgment is reflected in a "pure rate of time preference" parameter that weighs the welfare of current and future generations. See Section 6.3.1 for more information on intergenerational discounting and debate about the value of this parameter. One way to clarify distributional consequences if intergenerational impacts are important is to display time paths of benefits and costs without discounting, as recommended in Chapter 6 of these *Guidelines*.

10.5 Conclusion

This chapter provides a variety of tools, analytical considerations and guidance for conducting distributional analyses for environmental justice, children's environmental health and other factors. Tools and methods are intended to be flexible enough to accommodate various data and other constraints associated with particular scenarios, while introducing consistency and rigor in the way regulatory analyses consider distributional effects.

Methods for analyzing distributional impacts in the context of EJ, in particular, are continually being discussed, debated, and improved. For instance, EPA is in the process of developing more specific guidance on considering environmental justice concerns when planning human health risk assessments (U.S. EPA 2012b). Updates to this chapter about strengths and limitations of various analytical options, as well as new approaches, will be added when appropriate.

61 See U.S. EPA (2006a) and WHO (2007). The latter is available at http://www.who.int/ipcs/features/ehc/en/index.html (accessed on January 11, 2013).

Chapter 11

Presentation of Analysis and Results

This chapter provides some general guidance for presenting analytical results to policy makers and others interested in environmental policy development. Economic analyses play an important role throughout the policy development process. From the initial, preliminary evaluation of potential options through the preparation of a final economic analysis document, economic analysts participate in an interactive process with policy makers. The fundamental goal of this process is to collect, analyze, and present information useful for policy makers.

Economic analysis is often motivated by a desire to find an optimal outcome, such as a degree of stringency in a regulation, or a level of provision of a public good that yields the largest possible net benefits. Environmental statutes sometimes mandate criteria other than economic efficiency, such as best available control technology or lowest achievable emission rate. Policy makers rely on quantitative analysis to promulgate these approaches. In particular they rely on analyses that delineate the costs, benefits, or other impacts of a wide range of control options.

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. Conveying uncertainty effectively and reporting critical assumptions and key unquantified effects to decision makers is critical at all points in the policy-making process.

This chapter begins by providing general guidance on how to present the results of economic analyses, with a particular emphasis on presenting benefits and costs, including those that cannot be quantified and/or put into dollar terms. The chapter then discusses the components, or inputs, of an economic analysis, and how their effect on the economic analysis can best be communicated.

11.1 Presenting Results of Economic Analyses

The presentation of the results of an economic analysis should be thorough and transparent. The reader should be able to understand:

- What the primary conclusions of the economic analysis are;
- How the benefits and costs were estimated;
- What the important non-quantified or non-monetized effects are;
- What key assumptions were made for the analysis;
- What the primary sources of uncertainty are in the analysis; and
- How those sources of uncertainty affect the results.

An economic analysis of regulatory or policy options should present all identifiable costs and benefits that are incremental to the regulation or policy under consideration. These should include directly intended effects and associated costs, as well as ancillary (or co-) benefits and costs.

Benefits and costs should be reported in monetary terms whenever possible. In reality, however, there are often effects that cannot be monetized, and the analysis needs to communicate the full richness of benefit and cost information beyond what can be put in dollar terms. Benefits and costs that cannot be monetized should, if possible, be quantified (e.g., expected number of adverse health effects avoided). Benefits and costs that cannot be quantified should be presented qualitatively (e.g., directional impacts on relevant variables). Section 11.1.2 contains more detailed guidance on presenting this information in EPA's economic analyses.

Agencies are also required to provide OMB with an accounting statement reporting benefit and cost estimates when sending over each economically significant rule. Analysts should rely upon these *Guidelines* and *Circular A-4* for developing these estimates. *Circular A-4* describes the accounting statement on pages 44-46 and contains a suggested format for this accounting statement.[1]

In addition to requirements under *Circular A-4*, the 2010 OMB *Annual Report to Congress on the Costs and Benefits of Federal Regulations* asks agencies to provide a "simple, clear table of aggregated costs and benefits" of each economically significant rule in the regulatory Preamble of the Federal Register Notice and in the Executive Summary of the RIA (OMB 2010a, p. 51). EPA's guidance for satisfying these criteria is described more fully in Section 11.1.2 as part of the Agency's general guidance on reporting the results of benefit-cost analysis (BCA).

The results of economic analyses of environmental policies should generally be presented in three sections.

- **Results from BCA.** Estimates of the net social benefits should be presented based on the benefits and costs expressed in monetary terms. Non-monetized and unquantifiable benefits and costs should also be included and described in the presentation.
- **Results from cost-effectiveness analysis (CEA).** Under OMB *Circular A-4*, CEA should generally be performed for rules in which the primary effect is human health or safety. Results of these analyses should also be presented when they are conducted.[2]
- **Results from economic impact analysis (EIA) and distributional assessments.** Results of the EIA should be reported, including predicted effects on prices, profits, plant closures, employment, and any other effects. Distributional impacts for particular groups of concern, including small entities, governments, and environmental justice populations should also be presented.

The relative importance of these three sections will depend on the policy and statutory context of the analysis.

11.1.1 Presenting the Results of Benefit-Cost Analyses

When presenting the results of a BCA, the expected benefits and costs of the preferred regulatory option should be reported, together with the expected benefits and costs of alternative approaches. OMB's *Circular A-4* requires that at least one alternative be more stringent and one less stringent than the preferred option, and the incremental costs and benefits would be reported for each increasingly stringent option. Separate time streams of benefits and costs should be reported, in constant (inflation-adjusted), undiscounted dollars. Per the discussion in

1 The accounting statement is on page 47 of *Circular A-4*, available at www.whitehouse.gov/sites/default/files/omb/assets/omb/circulars/a004/a-4.pdf (accessed on January 21, 2011).

2 The Institute of Medicine (IOM) (2006) recently issued recommendations to regulatory agencies on how to perform health-based CEA. Recent examples of CEA can be found in appendices of several recent RIAs including those for PM NAAQS [see Appendix G listed at http://www.epa.gov/ttn/ecas/ria.html (accessed March 13, 2011)] and the Ground Water Rule [see Appendix H listed at http://www.epa.gov/safewater/disinfection/gwr/regulation.html (accessed March 13, 2011)].

Chapter 6, appropriately discounted benefits and costs should be reported as well.

Ideally, all benefits and costs of a regulation would be expressed in monetary terms, but this is almost never possible because of data gaps, unquantifiable uncertainties, and other challenges. It is important not to exclude an important benefit or cost category from BCA even if it cannot be placed in dollar terms. Instead, such benefits and costs should be expressed quantitatively if possible (e.g., avoided adverse health impacts). If important benefit or cost categories cannot be expressed quantitatively, they should be discussed qualitatively (e.g., a regulation's effect on technological innovation).

Quantifiable benefits and costs, properly discounted, should be compared to determine a regulation's net benefits, even if important benefits or costs cannot be monetized. However, an economic analysis should assess the likelihood that non-monetized benefits and costs would materially alter the net benefit calculation for a given regulation.

Incremental benefits, costs, and net benefits of moving from less to more stringent regulatory alternatives should also be presented. If a regulation has particularly significant impacts on certain groups or sub-populations, the various options' incremental impacts on these subpopulations or source categories should be reported. This should include a discussion of incremental changes in quantified and qualitatively described benefits and costs.

Given the number of potential models presented in Chapters 7 and 8, the analyst should take care to clearly indicate the correspondence between the benefit and cost estimates. For example, the cost analysis may include results from a general equilibrium model but the benefit analysis may only include partial equilibrium effects.[3] In this case, the cost side of the equation includes general equilibrium feedback effects while the benefit side does not. This difference should be clearly presented and explained.

The tables at the end of this chapter contain templates for presenting information on regulatory benefits and costs, including those benefits that cannot be quantified or put into dollar terms. The analyst's primary goal, using these tables, is to communicate the full richness of benefit and cost information instead of focusing narrowly on what can be put in dollar terms. Some guiding principles for constructing these tables follow.

- *All meaningful benefit and costs are included in all of the tables* even if they cannot be quantified or monetized. Not only does this provide consistency for the reader, but it also maintains important information on the context of the quantified and monetized benefits.
- *The types of benefits and costs are described briefly in plain terms* to make them clearer to the public and to decision makers, and they should be well-defined and mutually exclusive, to the extent possible. Benefits should be grouped a manner consistent with the categories in Table 7.1 of Chapter 7, although the order and specific characterization can be expected to vary by rule as needed.
- *The benefits are expressed first in natural or physical units* (i.e., number) to provide a more complete picture of what the rule accomplishes. These units are not discounted as they would be in a CEA because the goal here is to describe what might be termed the "physical scope" of the rule's benefits. It may be the case that physical or natural units are not relevant for presenting costs.
- *Explanatory notes accompany each benefit and cost entry* and can be used to describe whatever the most salient or important points are about scientific uncertainty, the type of benefit or cost, how it is estimated, or the presentation.

The benefit categories in these templates (e.g., improved human health, improved environment, and other benefits,) will need to be revised to reflect the benefits categories for the rule under

3 While there have been some attempts to include benefit estimates in general equilibrium models, these efforts are nascent (Sieg et al. 2004, Yang et al. 2004, and Jena et al. 2008).

Table 11.1 - Template for Regulatory Benefits Checklist

Overview of Benefits			
Benefits	**Effect can be Quantified?** (put in numeric terms)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Improved Human Health			
• **Reduced incidence of adult premature mortality** from exposure to $PM_{2.5}$	√	√	e.g., see Section 5.2 of the economic analysis
• **Reduced incidence of fetal loss** from reduced exposure to disinfection byproducts	√	--	*Notes and reference to section of the economic analysis*
• ***Unquantified human health benefit*** *with a brief description*	--	--	*Notes and reference*
Improved Environment			
• **Fewer fish killed** from reduced nutrient loadings into waterways	√	√	*Notes and reference*
• **Improved timber harvest** from lower tropospheric ozone concentrations	√	√	*Notes and reference*
• ***Other environmental benefit*** *with a brief description*	--	--	*Notes and reference*
Other Benefits			
• **Fuel savings** from improved efficiency in automobiles and light trucks	√	√	*Notes and reference*
• ***Other benefit*** *with a brief description*	--	--	*Notes and reference*

consideration. Simpler analyses may need only the overview (Table 11.1) and the final summary (Table 11.4).

Table 11.1 is a quick-glance summary of regulatory benefits and costs, the extent to which they could be quantified and monetized, and a reference to where they are more fully characterized or estimated in the economic analysis. Some benefits may be described only qualitatively.

Table 11.2 reports benefits in non-monetary terms along with the units and additional explanatory notes. The goal of this table is to communicate the physical scope of the regulation's benefits and costs rather than the dollar equivalent. Benefits here do not need to be discounted to present value, but the time associated with the quantities should be made clear (e.g., "annual" or "more than ten years").

Table 11.3 reports benefits in monetary terms along with a total for dollar-valued benefits. Here it is important to specify the reference year for the dollars (i.e., real terms), the discount rate(s) used, and the unit value and/or source.

Table 11.4 contains a template for bringing all this information together in summary that includes the type of benefit or cost, how it is measured, its quantity, and dollar benefits. When multiple regulatory options are included in this table, it is appropriate for including in the regulatory preamble as requested by OMB.

Consistent with recommendations in these *Guidelines* for communicating uncertainty, quantitative entries should generally include a central or best estimate in addition to a range or confidence interval. The ability to do this, of course, may be limited by data availability.

11.1.2 Presenting the Results of Cost-Effectiveness Analyses

When BCA is not possible, CEA may be the best available option. The cost-effectiveness of a policy

Table 11.2 - Template for Quantified Regulatory Benefits

Quantified Benefits			
Benefits	**Quantified Benefits** (confidence interval or range)	**Units**	**More Information** *(w/possible reference to section of the economic analysis)*
Improved Human Health			
• **Reduced incidence of adult premature mortality** from exposure to $PM_{2.5}$	estimate *(range)*	expected avoided expected premature deaths per year	e.g., range represents confidence interval
• **Reduced incidence of fetal loss** from reduced exposure to disinfection byproducts	estimate *(range)*	expected avoided fetal losses per year	e.g., confidence interval cannot be estimated. Range based on alternative studies
• ***Unquantified human health benefit*** *with a brief description*	*	*	e.g., data do not allow for quantification
Improved Environment			
• **Fewer fish killed** from reduced nutrient loadings into waterways	estimate *(range)*	thousands of fish per year	Notes *(reference)*
• **Improved timber harvest** from lower tropospheric ozone concentrations	estimate *(range)*	thousands of board feet per year	Notes *(reference)*
• ***Other environmental benefit*** *with a brief description*	*	*	Notes *(reference)*
Other Benefits			
• **Fuel savings** from improved efficiency in automobiles and light trucks	estimate *(range)*	millions of gallons of gasoline reduced per year	Notes *(reference)*
• ***Other benefit*** *with a brief description*	*	*	Notes *(reference)*

Note: * indicates the benefit cannot be quantified with available information

option is calculated by dividing the annualized cost of the option by non-monetary benefit measures. Options for such measures range from quantities of pollutant emissions reduced, measured in physical terms, to a specific improvement in human health or the environment, measured in reductions in illnesses or changes in ecological services rendered.

In the context of RIA, or other analyses of specific regulatory or policy options, CEA is most informative when several different options are analyzed. The analysis should include at least one option that is less stringent and at least one option that is more stringent than the preferred option. The incremental costs and non-monetary benefit yield of each option, in order of increasing stringency, should be reported.

The non-monetary measure of benefits used in a CEA must be chosen with great care to facilitate valid comparisons across options. The closer the chosen measure is to the variable that directly impacts social welfare, the more robust a CEA will be. Consider the following steps that a typical environmental economic assessment follows:

- Changes in emissions are estimated (e.g., tons of emissions); then

Table 11.3 - Template for Dollar-Valued Regulatory Benefits

Dollar-Valued Benefits			
Benefit	**Dollar Benefits** (millions per year)	**Basis of Value**	**More Information** *(w/possible reference)*
Improved Human Health			
• **Reduced incidence of adult premature mortality** from exposure to $PM_{2.5}$	$ estimate *($ range)*	e.g., $X based on Agency guidance	Notes *(reference)*
• **Reduced incidence of fetal loss** from reduced exposure to disinfection byproducts	*	Not available	Notes *(reference)*
• ***Unquantified human health benefit*** *with a brief description*	*	*	e.g., data insufficient to quantify (reference)
Improved Environment			
• **Fewer fish killed** from reduced nutrient loadings into waterways	$ estimate *($ range)*	e.g., $X based on WTP for recreational fishing	e.g., range reflects two different valuation approaches *(reference)*
• **Improved timber harvest** from lower tropospheric ozone concentrations	$ estimate *($ range)*	e.g., change in consumer and producer surplus	e.g., estimated from market model across several species *(reference)*
• ***Other environmental benefit*** *with a brief description*	*	*	Notes *(reference)*
Other Benefits			
• **Fuel savings** from improved efficiency in automobiles and light trucks	$ estimate *($ range)*	e.g., $X, based on net-of-tax average per gallon price	e.g., there is debate on how well fuel savings represent consumer benefits *(reference)*
• ***Other benefit*** *with a brief description*	*	*Not available*	Notes *(reference)*
TOTAL Benefits that can be monetized ($millions per year)	**$ estimate** *($ range)*		

Note: * indicates the benefit cannot be quantified with available information.

- Changes in environmental quality (e.g., changes in ambient concentrations of a given air pollutant) are estimated; then
- Changes in human health or welfare (e.g., changes in illness or visibility) are estimated.

Each successive step in this sequence yields a better measure for CEA.

To illustrate, consider a typical air pollution scenario. Depending on where and when air pollutants are released into the atmosphere, a given ton of a particular pollutant can have widely divergent impacts on ambient air quality. Similarly, depending on when and where air quality changes, widely different levels of human health impacts may result. Particularly when different regulatory approaches are under consideration (e.g., regulation of different source categories in different locations), failing to standardize the analyses on the benefit measure that directly affects human health or welfare will significantly reduce

Table 11.4 - Template for Summary of Benefits and Costs

Benefits Notes: e.g., "annual average numbers; 2006 dollars annualized at 3% discount rate" Best estimate, with range							
	Option 1		Proposed Option		Option 3		***Source, limitations, or other key notes***
	Number	***$ Millions***	***Number***	***$ Millions***	***Number***	***$ Millions***	
Improved Human Health							
• **Reduced incidence of adult premature mortality** from exposure to $PM_{2.5}$	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	*highlight most important points, as needed*
• **Reduced incidence of fetal loss** from reduced exposure to disinfection byproducts	estimate *(range)*	*	estimate *(range)*	*	estimate *(range)*	*	e.g., no valuation data exist. Effects are sensitive to dose-response model.
• ***Unquantified human health benefit*** *with a brief description*	*	*	*	*	*	*	e.g., risk data insufficient for quantification
Improved Environment							
• **Fewer fish killed** from reduced nutrient loadings into waterways	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	*Notes*
• **Improved timber harvest** from lower tropospheric ozone concentrations	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	*Notes*
• ***Other environmental benefit*** *with a brief description*	*	*	*	*	*	*	*Notes*
Other Benefits							
• **Fuel savings** from improved efficiency in automobiles and light trucks	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	estimate *(range)*	$ estimate *(range)*	*Notes*
• ***Other benefit*** *with a brief description*	*	*	*	*	*	*	*Notes*
TOTAL Benefits that can be monetized *(annualized, millions $2006)*	**$ estimate** *(range)*		**$ estimate** *(range)*		**$ estimate** *(range)*		e.g., total range may be overstated because of aggregation *(See Section 8.1 of economic analysis)*

Note: * indicates the benefit cannot be quantified with available information.

Table 11.4 - Template for Summary of Benefits and Costs (continued)

Costs 2006 dollars annualized at 3% discount rate Best estimate, with range				
	Option 1	Proposed Option	Option 3	***Source, limitations, or other key notes***
	$ Millions	***$ Millions***	***$ Millions***	
• **Initial capital costs** *with any brief description and units.*	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	*e.g., estimated from engineering cost models*
• **Type of cost** *with a brief description and units.* (This could include non-monetized costs.)	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	*Notes*
• **Type of cost** *with a brief description and units.* (This could include non-monetized costs.)	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	*Notes*
TOTAL Costs that can be monetized *(annualized, millions $2006)*	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	
TOTAL Net Benefits that can be monetized *(annualized, millions $2006)*	**$ estimate *(range)***	**$ estimate *(range)***	**$ estimate *(range)***	

the value of the analysis to decision makers (and the public).

When presenting the results of a CEA, the rationale for the selection of the non-monetary benefit measure must be described in detail. The presentation of results should also include a discussion of the limitations of the analysis, especially if an inferior measure, such as cost per ton of pollutant, must be used.

CEA is most useful when the policy or regulation in question affects a single endpoint. When multiple endpoints are affected (e.g., cancer and kidney failures), combining endpoints into a single effectiveness measure is impossible unless appropriate weighting factors exist for the multiple endpoints. The theoretically correct weights to apply are the dollar values associated with each endpoint, but generally it is the absence of these values that necessitates CEA. Therefore, it is not possible to compare a policy or regulation that reduces relatively more expected cancers, but fewer expected cases of kidney failure, with one that has the opposite relative effects. When this occurs, the effects of each option for each endpoint should be reported. A single endpoint may be selected for calculating cost-effectiveness, while other endpoints can be listed as ancillary benefits (or, if possible, their monetary value should be subtracted from the option's cost prior to calculating its cost-effectiveness) (OMB 2003).

The most cost-effective option — i.e., the option with the lowest cost per unit of benefit — is not necessarily the most economically efficient. 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, CEA can indicate which control measures or policies are inferior options.

11.1.3 Presenting the Results of EIA and Distributional Analyses

EIA and distributional outcomes 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 distributional outcomes.

The results of the EIA should also be reported for important sectors within the affected population — 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 in distributional assessments may include the expected allocation of benefits, costs, or both for specific subpopulations including those highlighted in the various mandates. These include minorities, low-income populations, small businesses, governments, not-for-profit organizations, 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.

11.1.4 Reporting the Effects of Uncertainty on Results of Economic Analyses

Estimates of costs, benefits and other economic impacts should be accompanied by indications of the most important sources of uncertainty embodied in the estimates, and, if possible, a quantitative assessment of their importance. OMB requires formal quantitative analysis of uncertainties for rules with annual economic effects of $1 billion or more.

In economic analysis, uncertainty encompasses two different concepts:

- Statistical variability of key parameters; and
- Incomplete understanding of important relationships.

Economic analyses of environmental policies and regulatory options will frequently have to accommodate both concepts. The importance of statistical variability is commonly assessed using Monte Carlo analyses (see U.S. EPA 1997). Delphic panels, or expert elicitation techniques, can help close knowledge gaps surrounding key relationships (see IEc 2004).

Ideally, an economic analysis would present results in the form of probability distributions that reflect the cumulative impact of all underlying sources of uncertainty. When this is impossible, due to time or resource constraints, results should be qualified with descriptions of major sources of uncertainty. If at all possible, information about the underlying probability distribution should be conveyed. (A forthcoming section of these *Guidelines* will more fully address uncertainty issues.)

As recommended in Chapter 6, many EPA analyses will employ more than one discount rate to reflect different underlying approaches to discounting. When the choice of discount rate affects the outcome of the analysis, analysts should take extra care to convey the underlying theory and assumptions to decision makers. See Chapter 6 for more information.

11.2 Communicating Data, Model Choices and Assumptions, and Related Uncertainty

An economic analysis of an environmental regulation should carefully describe the data used in the analysis, the models it relies on, major assumptions that were made in running the models, and all major areas of uncertainty in each of these elements. Presentations of economic analyses should strive for clarity and transparency. An analysis whose conclusions can withstand close scrutiny is more likely to provide policy makers with the information they need to develop robust environmental policies.

11.2.1 Data

An economic analysis should clearly describe all important data sources and references used. Unless the data are confidential business information or some other form of private data, they should be available to policy makers, other researchers, policy analysts and the public. Providing documentation and access to the data used in an analysis is crucial to the credibility and reproducibility of the analysis.

EPA Order 5360.1 A2 (U.S. EPA 2000a) and the applicable federal regulations established a mandatory quality system for EPA. As required by the quality system, all EPA offices have developed quality management plans to ensure the quality of their data and information products.

Until recently, federal quality assurance (QA) requirements only applied to measurement and collection of *primary* environmental data. This meant that QA requirements often did not apply to economic analyses, which usually rely on the use of secondary data. However, this changed with the introduction of QA requirements regarding use of secondary data. In 2002 the Agency released QA guidelines regarding use of secondary data, and released Agency guidance, *Guidance for Quality Assurance Project Plans,* that includes procedures for documenting secondary data (U.S. EPA 2002f).

In any economic analysis, there should be a clear presentation of how data are used and a concise explanation of why the data are suitable for the selected purpose. The data's accuracy, precision, representativeness, completeness, and comparability should be discussed when applicable. When data are available from more than one source, a rationale for choosing the source of the data should be provided.

11.2.2 Model Choices and Assumptions

An economic analysis of an environmental regulation should carefully describe the models it relies on, the major assumptions made in running the models (to be discussed more fully below), and any areas of outstanding uncertainty. The analyst should take particular care to explain any results that might be viewed as counter-intuitive. In particular, analysts should be careful not to accept model output blindly. Any model that is used without proper thought given to both its input and output may become a "black box" insofar as nonsensical results may result from a misspecified scenario, a coding error, or any of a number of other causes.

In the process of conducting an economic analysis, it is sometimes necessary to bridge an information gap by making an assumption. Analysts should not simply note the information gap, but should also justify the chosen assumption and provide a rationale for choosing one assumption over other plausible options. The analyst should take care not to overlook information gaps that are filled with a piece of information that is only slightly related to the desired information. Analysts are advised to keep a running list of assumptions. This will make it easier to identify "key assumptions" for the final report. The likely impact of errors in assumptions should be characterized both in terms of direction and magnitude of effect when feasible.

Maintaining a list of assumptions can benefit the analysis in several ways. In the short run, a list can serve to focus analysts' attention on those assumptions with the greatest potential to affect net benefits, possibly leading to new approaches to bridging an information gap. In the long run, highlighting information gaps may encourage EPA or others to devote attention and resources to generating that information.

Whenever the likely errors in a particular assumption can be characterized numerically or statistically, the factor is a good candidate for sensitivity analysis or uncertainty analysis, respectively. In many cases, only a narrative description of the impact of errors in assumptions is possible. The analyst should include a table that clearly lays out all of the key assumptions and the potential magnitude and direction of likely errors in assumptions in the summary of results.

11.2.3 Addressing Uncertainty Driven by Assumptions and Model Choice

Every analysis should address uncertainties resulting from the choices the analyst has made. For example, many economic analyses performed at EPA include assessments of economic impacts expected to occur decades into the future. Estimates of the future costs and benefits of a regulation will be sensitive to assumptions about growth rates for populations, source categories, economic activity, and technological change, as well as many other factors. Sensitivity analyses on key variables in the baseline scenario should be performed and reported when possible. This allows the reader to assess the importance of the assumptions made for the central case. Some of these variables may be affected by a regulation, particularly the assumed rate of technological innovation. (Please see Chapter 5 for additional guidance on specifying baselines.)

The impact of using alternative assumptions or alternative models can be assessed quantitatively in many cases.

- **Alternative analysis.** An analysis of alternative assumptions or "alternative analysis" is the substitution of one of the key assumptions with another. In presenting the results, the alternative analysis is presented with equal weight as the primary analysis and is presented alongside of the primary analysis, even if the probability of the alternative assumption differs from that of the primary analysis. Because performing an alternative analysis on all the assumptions in an analysis is prohibitively resource intensive, the analyst should focus on the assumptions that have the largest impact on the final results of the particular analysis. Thus, keeping a running list of the "key assumptions" in an analysis is recommended.
- **Sensitivity analysis.** A sensitivity analysis is used to assess how the final results or other aspects of the analysis change as input parameters change, particularly when only point estimates of parameters are available. A regulatory impact analysis benefits from knowing how the cost-effectiveness of a particular technology changes as fuel prices change, or how the net benefits of a BCA change as one of the model coefficients change. Typically, a sensitivity analysis measures how the model's output changes as one of the input parameters change. Joint sensitivity analysis (varying more than one parameter at a time) is sometimes useful as well.
- **Model uncertainty.** In addition to explaining the uncertainty in a model's parameters, analysts should discuss the uncertainty generated by the choice of model. Multiple models are often available to the analyst, and choosing among them is similar to making an assumption. Implicit in the choice of a model are many factors. For example, one model may take long-run effects into account while another model does not. When possible, presenting results of an alternate model can inform the reader. When resource limitations prevent the use of an alternative model, it is still often possible to predict the direction and likely magnitude of the use of an alternate model, and the analyst should present this information to the reader.

11.3 Use of Economic Analyses

The primary purpose of conducting economic analysis is to provide policy makers and others with detailed information on a 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. For this reason, these *Guidelines* reflect updated information associated with 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 is often 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. Even if the criterion of economic efficiency were the sole guide to policy decisions, social benefit and costs 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, these *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 trade-offs among them. Ultimately statutory requirements dictate if and how the analytic results are used in standard setting. In any case, these results, along with other analyses and considerations, serve as important inputs for the broader policy-making process and serve as important resources for the public.

Appendix A
Economic Theory

This appendix provides a brief overview of the fundamental theory underlying the approaches to economic analysis discussed in Chapters 3 through 9. The first section summarizes the basic concepts of the forces governing a market economy in the absence of government intervention. Section A.2 describes why markets may behave inefficiently. If the preconditions for market efficiency are *not* met, government intervention can be justified.[1] The usefulness of benefit-cost analysis (BCA) as a tool to help policy makers determine the appropriate policy response is discussed in Section A.3. Sections A.4 and A.5 explain how economists measure the economic impacts of a policy and set the optimal level of regulation. Section A.6 concludes and provides a list of additional references.

A.1 Market Economy

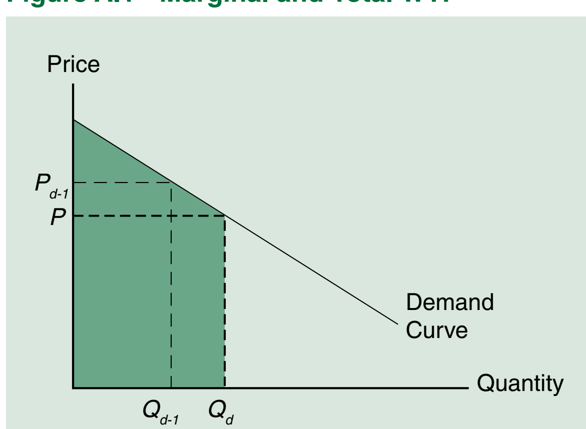
The economic concept of a market is used to describe any situation where exchange takes place between consumers and producers. Economists assume that consumers purchase the combination of goods that maximizes their well-being, or "utility," given market prices and subject to their household budget constraint. Economists also assume that producers (firms) act to maximize their profits. Economic theory posits that consumers and producers are rational agents who make decisions taking into account *all* of the costs — the full opportunity costs — of their choices, given their own resource constraints.[2] The purpose of economic analysis is to understand how the agents interact and how their interactions add up to determine the allocation of society's resources: what is produced, how it is produced, for whom it is produced, and how these decisions are made. The simplest tool economists use to illustrate consumers' and producers' behavior is a market diagram with supply and demand curves.

1 EPA's mandates frequently rely on criteria other than economic efficiency, so policies that are not justified due to a lack of efficiency are sometimes adopted.

2 *Opportunity cost* is the next best alternative use of a resource. The full opportunity cost of producing (consuming) a good or service consists of the maximum value of other goods and services that could have been produced (consumed) had one not used the limited resources to produce (purchase) the good or service in question. For example, the full cost of driving to the store includes not only the price of gas but also the value of the time required to make the trip.

The demand curve for a single individual shows the quantity of a good or service that the individual will purchase at any given price. This quantity demanded assumes the condition of holding all else constant, i.e., assuming the budget constraint, information about the good, expected future prices, prices of other goods, etc. remain constant. The height of the demand curve in Figure A.1 indicates the maximum price, P, an individual with Q_d units of a good or service would be willing to pay to acquire an additional unit of a good or service. This amount reflects the satisfaction (or utility) the individual receives from an additional unit, known as the *marginal benefit* of consuming the good. Economists generally assume that the marginal benefit of an additional unit is slightly less than that realized by

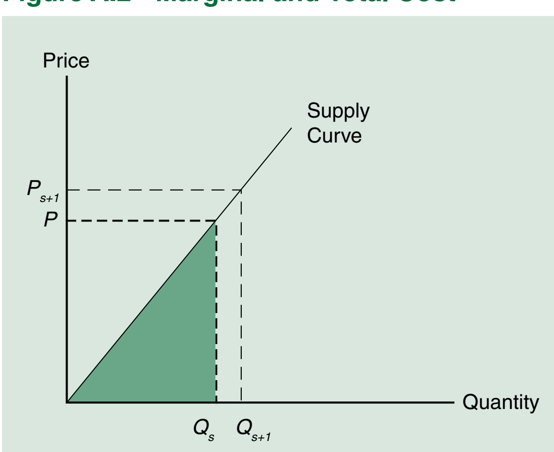
Figure A.1 - Marginal and Total WTP



the previous unit. The amount an individual is willing to pay for one more unit of a good is less than the amount she paid for the last unit; hence, the individual demand curve slopes downward. A market demand curve shows the total quantity that consumers are willing to purchase at different price levels, i.e., their collective willingness to pay (WTP) for the good or service. In other words, the market demand curve is the horizontal sum of all of the individual demand curves.

The concept of an individual's WTP is one of the fundamental concepts used in economic analyses, and it is important to distinguish between total and marginal WTP. Marginal WTP is the additional amount the individual would pay for one additional unit of the good. The total WTP is the aggregate amount the individual is willing to pay for the total quantity demanded (Q_d). Figure A.1 illustrates the difference between the marginal and total WTP. The height of the demand curve at a quantity Q_{d-1} gives the marginal WTP for the $Q_{d-1}{}^{th}$ unit. The height of the demand curve at a quantity Q_d gives the marginal WTP for the $Q_d{}^{th}$ unit. Note that the marginal WTP is greater for the $Q_{d-1}{}^{th}$ unit. The *total* WTP is equal to the sum of the marginal WTP for each unit up to Q_d. The shaded area under the demand curve from the origin up to Q_d shows total WTP.

Figure A.2 - Marginal and Total Cost

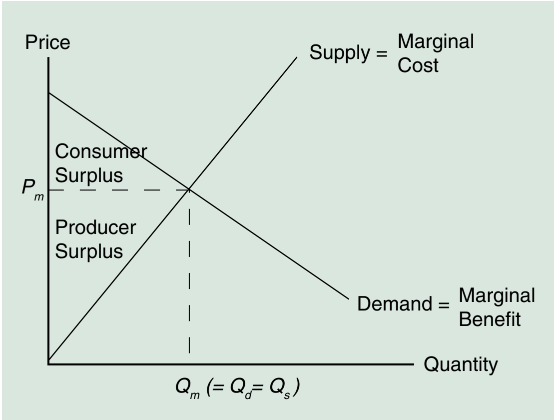


An individual producer's supply curve shows the quantity of a good or service that an individual or firm is willing to sell (Q_s) at a given price. As a profit-maximizing agent, a producer will only be willing to sell another unit of the good if the market price is greater than or equal to the cost of producing that unit. The cost of producing the additional unit is known as the *marginal cost*. Therefore, the individual supply curve traces out the marginal cost of production and is also the marginal cost curve. Economists generally assume that the cost of producing one additional unit is greater than the cost of producing the previous unit because resources are scarce. Therefore the supply curve is assumed to slope upward. In Figure A.2, the marginal cost of producing the $Q_s{}^{th}$ unit of the good is given by the height of the supply curve at Q_s. The marginal cost of producing the $Q_{s+1}{}^{th}$ unit of the good is given by the height of the supply curve at Q_{s+1}, which greater than the cost of producing the $Q_s{}^{th}$ unit, and greater than the price, P. The *total cost* of producing Q_s units is equal to the shaded area under the supply curve from the origin to the quantity Q_s. The market supply curve is simply the horizontal summation of the individual producers' marginal cost curves for the good or service in question.

In a competitive market economy, the intersection of the market demand and market supply curves determines the equilibrium price and quantity of a good or service sold. The demand curve reflects the marginal benefit consumers receive from purchasing an extra unit of the good (i.e., it reflects their marginal WTP for an extra unit). The supply curve reflects the marginal cost to the firm of producing an extra unit. Therefore, at the competitive equilibrium, the price is where the marginal benefit equals the marginal cost. This is illustrated in Figure A.3, where the supply curve intersects the demand curve at equilibrium price P_m and equilibrium quantity Q_m.

A counter-example illustrates why the equilibrium price and quantity occur at the intersection of the market demand and supply curves. In Figure A.3, consider some price greater than P_m where Q_s is greater than Q_d (i.e., there is *excess supply*). As producers discover that they cannot sell off their inventories, some will reduce prices slightly, hoping to attract more customers. At lower prices consumers will purchase more of the good (Q_d increases) although firms will be willing to sell less (Q_s

Figure A.3 - Market Equilibrium



decreases). This adjustment continues until Q_d equals Q_s. The reverse situation occurs if the price becomes lower than P_m. In that case, Q_d will exceed Q_s (i.e., there is *excess demand*) and consumers who cannot purchase as much as they would like are willing to pay higher prices. Therefore, firms will begin to increase prices, causing some reduction in the Q_d but also increasing Q_s. Prices will continue to rise until Q_s equals Q_d. At this point no purchaser or supplier will have an incentive to change the price or quantity; hence, the market is said to be in equilibrium.

Economists measure a consumer's net benefit from consuming a good or service as the excess amount that she is willing to spend on the good or service over and above the market price. The net benefit of all consumers is the sum of individual consumer's net benefits — i.e., what consumers are willing to spend on a good or service over and above that required by the market. This is called the *consumer surplus*. In Figure A.3, the market demands price P_m for the purchase of quantity Q_m. However, the demand curve shows that there are consumers willing to pay more than price P_m for all units prior to Q_m. Therefore, the consumer surplus is the area under the market demand (marginal benefit) curve but above the market price. Policies that affect market conditions in ways that decrease prices by decreasing costs of production (i.e., that shift the marginal cost curve to the right) will generally increase consumer surplus. This increase can be used to measure the benefits that consumers receive from the policy.[3]

On the supply side, a producer can be thought to receive a benefit if he can sell a good or service for more than the cost of producing an additional unit — i.e., its marginal cost. Figure A.3 shows that there are producers willing to sell up to Q_m units of the good for less then the market price P_m. Hence, the net benefit to producers in this market, known as *producer surplus*, can be measured as the area above the market supply (marginal cost) curve but below the market price. Policies that increase prices by increasing market demand for a good (i.e., that shift the marginal benefit curve to the right) will generally increase producer surplus. This increase can be used to measure the benefits that producers receive from the policy.

Economic efficiency is defined as the maximization of social welfare. In other words, the efficient level of production is one that allows society to derive the largest possible net benefit from the market. This condition occurs where the (positive) difference between the total WTP and total costs is the largest. In the absence of externalities and other market failures (explained below), this occurs precisely at the intersection of the market demand and supply curves where the marginal benefit equals the marginal cost. This is also the point where total surplus (consumer surplus plus producer surplus) is maximized. There is no way to rearrange production or reallocate goods so that someone is made better off without making someone else worse off — a condition known as *Pareto optimality*. Notice that economic efficiency requires only that net benefits be maximized, *irrespective of to whom those net benefits accrue*. It does not guarantee an "equitable" or "fair" distribution of these surpluses among consumers and producers, or between sub-groups of consumers or producers.

Economists maintain that *if the economic conditions are such that there are no market imperfections* (as discussed in Section A.2), then this condition of Pareto-optimal economic efficiency

3 Section A.4.2 provides a more technical discussion of how consumer surplus serves as a measure of benefits.

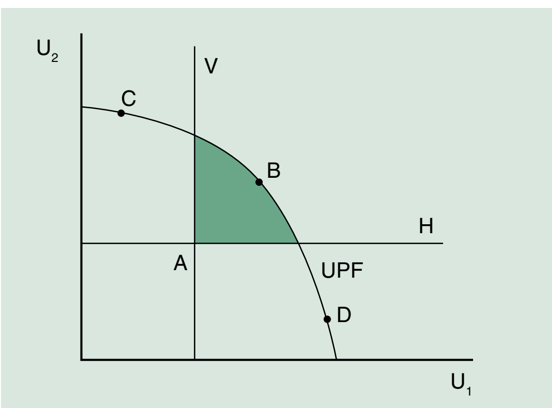
occurs automatically.[4] That is, no government intervention is necessary to maximize the sum of consumer surplus and producer surplus. This theory is summarized in the two Fundamental Theorems of Welfare Economics, which originate with Pareto (1906) and Barone (1908):

1. **First Fundamental Welfare Theorem.** Every competitive equilibrium is Pareto-optimal.

2. **Second Fundamental Welfare Theorem.** Every Pareto-optimal allocation can be achieved as a competitive equilibrium after a suitable redistribution of initial endowments.

One graphical representation of these results is given in Figure A.4, which shows utility (welfare) levels in a two-person economy.[5] The curve shown is the utility possibility frontier (UPF) curve; the area within it represents the set of all possible welfare outcomes. Each point on the negatively sloped UPF curve is Pareto optimal since it is not possible to increase the utility of one person without decreasing the utility of the other. If the initial allocation is at point A, then the set of Pareto-superior (welfare-enhancing) outcomes include all points in the shaded area, bordered by *H*, *V*, and the UPF curve.[6] If trading is permitted, the First Welfare Theorem applies and the market will move the economy to a superior, more efficient point such as *B*. Then the Second Welfare Theorem simply says that for any chosen point along the UPF curve, given a set of lump sum taxes and transfers, an initial allocation can be determined inside the UPF from which the market will achieve the desired outcome.[7]

Figure A.4 - Utility Possibility Frontier



A.2 Reasons for Market or Institutional Failure

If the market supply and demand curves reflect society's true marginal social cost and WTP, then a laissez-faire market (i.e., one governed by individual decisions and not government authority) will produce a socially efficient result. However, when markets do not fully represent social values, the private market will not achieve the efficient outcome (see Mankiw 2004, or any basic economics text); this is known as a *market failure*. Market failure is primarily the result of externalities, market power, and inadequate or asymmetric information. Externalities are the most likely cause of the failure of private and public sector institutions to account for environmental damages.

Externalities occur when markets do not account for the effect of one individual's decisions on another individual's well-being.[8] In a free market producers make their decisions about what and how much to produce, taking into account the cost of the required inputs — labor, raw materials,

4 Technically, there are two types of efficiency. *Allocative efficiency* means that resources are used for the production of goods and services most wanted by society. *Productive efficiency* implies that the least costly production techniques are used to produce any mix of goods and services. Allocative efficiency requires that there be productive efficiency, but productive efficiency can occur without allocative efficiency. Goods can be produced at the least-costly method without being most wanted by society. Perfectly competitive markets in the long run will achieve both of these conditions, producing the "right" goods (allocative efficiency) in the "right" way (productive efficiency). These two conditions imply Pareto-optimal economic efficiency. (See Varian 1992 or any basic economics text for a more detailed discussion.)

5 Another, perhaps more commonly used, graphical tool to explain the First and Second Welfare Theorems is an Edgeworth box. See Varian (1992) or other basic economic textbook for a detailed discussion.

6 Note that efficiency could be obtained by moving along the vertical line *V*, which keeps utility of person 1 (U_1) constant while increasing utility of person 2 (U_2), or by moving along the horizontal line *H*, which only shows improvements in utility for person 1. Moving to point *B* improves the utility for both individuals.

7 Note that outcomes on the frontier such as *C* and *D*, although efficient, may not be desired on equity, or fairness, grounds.

8 More formally, an externality occurs when the production or consumption decision of one party has an unintended negative (positive) impact on the profit or utility of a third party. Even if one party compensates the other party, an externality still exists (Perman et al. 2003). See Baumol and Oates (1988) or any basic economics textbook for similar definitions and more detailed discussion.

machinery, energy. Consumers purchase goods and services taking into account their income and their own tastes and preferences. This means that decisions are based on the private costs and private benefits to market participants. If the consumption or production of these goods and services poses an external cost or benefit on those not participating in the market, however, then the market demand and supply curves no longer reflect the true marginal social benefit and marginal social cost. Hence, the market equilibrium will no longer be the socially (Pareto) efficient outcome.

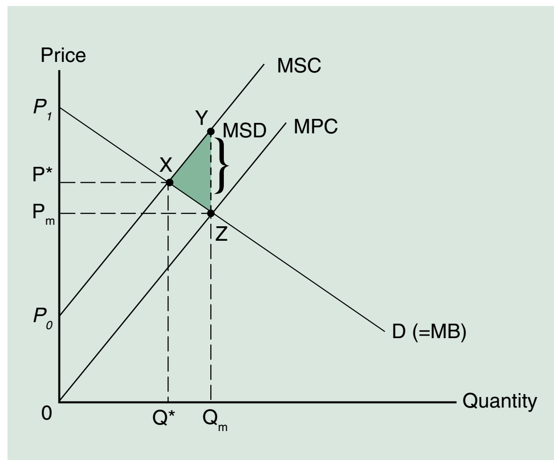
Externalities can arise for many reasons. Transactions costs or poorly defined property rights can make it difficult for injured parties to bargain or use legal means to ensure that the costs of the damages caused by polluters are internalized into their decision making.[9] Activities that pose environmental risks may also be difficult to link to the resulting damages and often occur over long periods of time. Externalities involve goods that people care about but are not sold in markets.[10] Air pollution causes ill health, ecological damage, and visibility impacts over a long time period, and the damage is often far from the source(s) of the pollution. The additional social costs of air pollution are not included in firms' profit maximization decisions and so are not considered when firms decide how much pollution to emit. The lack of a market for clean air causes problems and provides the impetus for government intervention in markets involving polluting industries.

9 A property right can be defined as a bundle of characteristics that confer certain powers to the owner of the right: the exclusive right to the choice of use of a resource, the exclusive right to the services of a resource, and the right to exchange the resource at mutually agreeable terms. Externalities typically arise from the violation of one or more of the characteristics of well-defined property rights. This implies that the distortions resulting from an externality can be eliminated by appropriately establishing these rights. This insight is summarized by the famous "Coase theorem" which states that if property rights over an environmental asset are clearly defined, and bargaining among owners and prospective users of the asset is allowed, then externality problems can be corrected and the efficient outcome will result regardless of who was initially given the property right. The seminal paper is Coase (1960).

10 Often these are goods that exhibit public good characteristics. Pure public goods are those that are non-rivalrous in consumption and non-excludable. [See Perman et al. (2003) for a detailed discussion of these, as well as congestible and open access resources — i.e., goods that are neither pure public nor pure private goods.] Because exclusive property rights cannot be defined for these types of goods, pure private markets cannot provide for them efficiently.

Figure A.5 illustrates a negative externality associated with the production of a good. For example, a firm producing some product might also be generating pollution as a by-product. The pollution may impose significant costs — in the form of adverse health effects, for example — on households living downwind or downstream of the firm. Because those costs are not borne *by the firm,* the firm typically does not consider them in its production decisions. Society considers the pollution a cost of production, but the firm typically will not. In this figure:

Figure A.5 - Negative Externality



- *D* is the market demand (marginal benefit) curve for the product;
- *MPC* is the firm's marginal private real-resource cost of production, excluding the cost of the firm's pollution on households;
- *MSD* is the marginal social damage of pollution (or the marginal external cost) that the firm is not considering; and
- *MSC* is society's marginal social cost associated with production, including the cost of pollution ($MSC = MPC + MSD$).

In an incomplete market, producers pay no attention to external costs, and production occurs where market demand (D) and the marginal private real-resource cost (MPC) curves intersect — at a price P_m and a quantity Q_m. In this case, net social welfare (total WTP minus total social costs) is equal to the area of the triangle P_0P_1X less the area of triangle *XYZ*.[11] If the full social cost of production, including the cost of pollution, is taken into consideration, then the marginal cost curve should be increased by the amount of the marginal social damage (*MSD*) of pollution.[12] Production will now occur where the demand and marginal social cost (*MSC*) curves intersect — at a price *P** and a quantity *Q**. At this point net social welfare (now equal to the area of the triangle, P_0P_1X, alone) is maximized, and therefore the market is at the socially efficient point of production. This example shows that when there is a negative externality such as pollution, and the social damage (external cost) of that pollution is not taken into consideration, the producer will oversupply the polluting good.[13] The shaded triangle (*XYZ*), referred to as the *deadweight loss (DWL)*, represents the amount that society loses by producing too much of the good.

A.3 Benefit-Cost Analysis

If a negative externality such as pollution exists, an unregulated market will not account for its cost to society, and the result will be an inefficient outcome. In this case, there may be a need for government intervention to correct the market failure. A correction may take the form of dictating the allowable level of pollution or introducing a market mechanism to induce the optimal level of pollution.[14] Figure A.5 neatly summarizes this in a single market diagram. To estimate the *total* costs and benefits to society of an activity or program, the costs and benefits in each affected market, as well as any non-market costs or benefits, are added up. This is done through BCA.

11 Recall from Section A.1 that total WTP is equal to the area under the demand curve from the origin to the point of production ($0P_1ZQ_m$). Total costs (to society) are equal to the area under the MSC curve from the origin to the point of production ($0P_0YQ_m$).

12 When conducting BCA related to resource stocks, the MSD or marginal external cost is the present value of future net benefits that are lost to due to the use of the resource at present. That is, exhaustible resources used today will not be available for future use. These foregone future benefits are called *user costs* in natural resource economics (see Scott 1953, 1955). The marginal user cost is the user cost of one additional unit consumed in the present, and is added together with the marginal extraction cost to determine the MSC of resource use.

13 Similarly, the private market will undersupply goods for which there are positive externalities, such as parks and open space.

14 Chapter 4 discusses the various regulatory techniques and some non-regulatory means of achieving pollution control.

BCA can be thought of as an accounting framework of the overall social welfare of a program, which illuminates the trade-offs involved in making different social investments (Arrow et al. 1996). It is used to evaluate the favorable effects of a policy action and the associated opportunity costs. The favorable effects of a regulation are the benefits, and the foregone opportunities or losses in utility are the costs. Subtracting the total costs from the total monetized benefits provides an estimate of the regulation's net benefits to society. An efficient regulation is one that yields the maximum net benefit, assuming that the benefits can be measured in monetary terms.

BCA can also be seen as a type of market test for environmental protection. In the private market, a commodity is supplied if the benefits that society gains from its provision, measured by what consumers are willing to pay, outweigh the private costs of producing the commodity. Economic efficiency is measured in a private market as the difference between what consumers are willing to pay for a good and what it costs to produce it. Since clean air and clean water are public goods, private suppliers cannot capture their value and sell it. The government determines their provision through environmental protection regulation. BCA quantifies the benefits and costs of producing this environmental protection in the same way as the private market, by quantifying the WTP for the environmental commodity. As with private markets, the efficient outcome is the option that maximizes net benefits.

The key to performing BCA lies in the ability to measure both benefits and costs in monetary terms so that they are comparable. Consumers and producers in regulated industries and the governmental agencies responsible for implementing and enforcing the regulation (and by extension, taxpayers in general) typically pay the costs. The total cost of the regulation is found by summing the costs to these individual sectors. (An example of this, excluding the costs to the government, is given in Section A.4.3.) Since environmental regulation usually addresses some externality, the benefits of a regulation often occur *outside* of markets. For example, the primary benefits of drinking water regulations are improvements in human health. Once the expected reduction in illness and premature mortality associated with the regulation is calculated, economists use a number of techniques to estimate the value that society places on these health improvements.[15] These monetized benefits can then be summed to obtain the total benefits from the regulation.

Note that in BCA gains and losses are weighted equally regardless of to whom they accrue. Evaluation of the fairness, or the equity, of the net gains cannot be made without specifying a social welfare function. However there is no generally agreed-upon social welfare function, and assigning relative weights to the utility of different individuals is an ethical matter that economists strive to avoid. Given this dilemma, economists have tried to develop criteria for comparing alternative allocations where there are winners and losers without involving explicit reference to a social welfare function. According to the Kaldor-Hicks compensation test, named after its originators Nicholas Kaldor and J.R. Hicks, a reallocation is a welfare-enhancing improvement to society if:

1. The winners could theoretically compensate the losers and still be better off; and

2. The losers could not, in turn, pay the winners to not have this reallocation and still be as well off as they would have been if it did occur (Perman et al. 2003).

While these conditions sound complex, they are met in practice by assessing the net benefits of a regulation through BCA. The policy that yields the highest positive net benefit is considered welfare enhancing according to the Kaldor-Hicks criterion. Note that the compensation test is stated in terms of *potential* compensation and does not solve the problem of evaluating the fairness of the distribution of well-being in society. Whether and how the beneficiaries of a regulation should compensate the losers involves a value judgment and is a separate decision for government to make.

Finally, BCA may not provide the *only* criterion used to decide if a regulation is in society's best interest. There are often other, overriding considerations for promulgating regulation. Statutory instructions, political concerns, institutional and technical feasibility, enforceability, and sustainability are all important considerations in environmental regulation. In some cases a policy may be considered desirable even if the benefits to society do not outweigh its costs, particularly if there are ethical or equity concerns.[16] There are also practical limitations to BCA. Most importantly, this type of analysis requires assigning monetized values to non-market benefits and costs. In practice it can be very difficult or even impossible to quantify gains and losses in monetary terms (e.g., the loss of a species, intangible effects).[17] In general, however, economists believe that BCA provides a systematic framework for comparing the social costs and benefits of proposed regulations, and that it contributes useful information to the decision-making process about how scarce resources can be put to the best social use.

A.4 Measuring Economic Impacts

A.4.1 Elasticities

The net change in social welfare brought about by a new environmental regulation is the sum of the negative effects (i.e., loss of producer and consumer surplus) and the positive effects (or social benefits) of the improved environmental quality. This is shown graphically for a single market in Figure A.5 above. The use of demand and supply curves highlights the importance of assessing how individuals will respond to changes in market conditions. The net benefits of a policy will depend on how responsively producers and consumers react to a change in price. Economists

15 Chapter 7 discusses a variety of methods economists use to value environmental improvements.

16 Chapter 9 addresses equity assessment and describes the methods available for examining the distributional effects of a regulation.

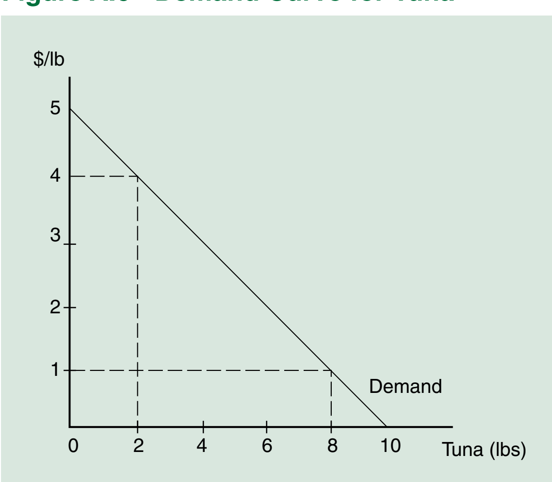
17 Kelman (1981) argues that it is even unethical to try to assign quantitative values to non-marketed benefits.

measure this responsiveness by the supply and demand elasticities.

The term "elasticity" refers to the sensitivity of one variable to changes in another variable. The price elasticity of demand (or supply) for a good or service is equal to the percentage change in the quantity demanded (or supplied) that would result from a 1 percent increase in the price of that good or service. For example, a price elasticity of demand for tuna equal to -1 means that a 1 percent increase in the price of tuna results in a 1 percent decrease in the quantity demanded. 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 consumed or produced. For example, according to the demand curve for tuna shown in Figure A.6, at a price of $1 per pound, a 10 percent increase in price would reduce quantity demanded by 2.5 percent (from 8 lbs to 7.8 lbs). At a price of $4 per pound, a 10 percent increase in price would result in a 40 percent decrease in quantity demanded (from 2 to 1.2 lbs). This implies that the price elasticity of demand is -0.25 when tuna costs $1/lb but -4 when the price is $4/lb. When calculating elasticities it is important realize where one is on the supply or demand curve, and the price or quantity should be stated when reporting an elasticity estimate.

Elasticities are important in measuring economic impacts because they determine how much of a

Figure A.6 - Demand Curve for Tuna



price increase will be passed on to the consumer. For example if a pollution control policy leads to an increase in the price of a good, multiplying the price increase by current quantity sold generally will not provide an accurate measure of impact of the policy. Some of the impact will take the form of higher prices for the consumer, but some of the impact will be a decrease in the quantity sold. The amount of the price increase that is passed on to consumers is determined by the elasticity of demand relative to supply (as well as existing price controls). "Elastic" demand (or supply) indicates that a small percentage increase in price results in a larger percentage decrease (increase) in quantity demanded (supplied).[18] All else 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. In determining the economic impacts of a rule, supply characteristics in the industries affected by a regulation can be as important as demand characteristics. For highly elastic *supply* curves relative to the demand curves, it is likely that cost increases or decreases will be passed on to consumers.

The many variables that affect the elasticity of demand include:

- The cost and availability of close substitutes;
- The percentage of income a consumer spends on the good;
- How necessary the good is for the consumer;
- The amount of time available to the consumer to locate substitutes;
- The expected future price of the good; and
- The level of aggregation used in the study to estimate the elasticity.

The availability of close substitutes is one of the most important factors that determine demand elasticity. A product with close substitutes at similar prices tends to have an elastic demand,

18 Demand (or supply) is said to be "elastic" if the absolute value of the price elasticity of demand (supply) is greater than one and "inelastic" if the absolute value of the elasticity is less than one. If a percentage change in price leads to an equal percentage change in quantity demanded (supplied) (i.e., if the absolute value of elasticity equals one), demand (supply) is "unit elastic."

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. Narrowly defined markets (e.g., salmon) will have more elastic demands than broadly defined markets (e.g., food) since there are more substitutes for narrow goods.

Another factor that affects demand elasticities is whether the affected product represents a substantial or necessary portion of customers' costs or budgets. Goods that account for a substantial portion of consumers' budgets or disposable income tend to be relatively price elastic. This is because consumers are more aware of small changes in the price of expensive goods compared to small changes in the price of inexpensive goods, and therefore may be more likely to seek alternatives. 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. If the good is a necessity item, the quantity demanded is unlikely to change drastically for a given change in price. Demand will be relatively inelastic.

Elasticities tend to increase over time, as firms and customers have more time to respond to changes in prices. Although a company may face an inelastic demand curve in the short run, it could experience greater losses in sales from a price increase in the long run. Over time customers begin to find substitutes or new substitutes are developed. However, temporary price changes may affect consumers' decisions differently than permanent ones. The response of quantity demanded during a one-day sale, for example, will be much greater than the response of quantity demanded when prices are expected to decrease permanently. Finally, it is important to keep in mind that elasticities differ at the firm versus the industry level. It is not 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 cost.

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 relatively elastic supply curves, it is likely that cost increases or decreases will be passed on to consumers. The elasticity of supply depends, in part, on how quickly per unit costs rise as firms increase their output. Among the many variables that influence this rise in cost are:

- The cost and 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.

Similar to the determinants of demand elasticity, the factors influencing the price elasticity of supply all relate to a firm's degree of flexibility in adjusting production decisions in response to changing market conditions. The more easily a firm can adjust production levels, find input substitutes, or adopt new production technologies, the more elastic is supply. Supply elasticities tend to increase over time as firms have more opportunities to renegotiate contracts and change production technologies. When production takes time, the quantity supplied may be more responsive to expected future price changes than to current price changes.

Demand and supply elasticities are available for the aggregate output of final goods in most industries. They are usually published in journal articles on research pertaining to a particular industry.[19]

19 Another useful source of elasticity estimates is the recently developed EPA Elasticity Databank (U.S. EPA 2007d). In the absence of an encyclopedic "Book of Elasticities" the Elasticity Databank serves as a searchable database of elasticity parameters across a variety of types (i.e., demand and supply elasticities, substitution elasticities, income elasticities, and trade elasticities) and economic sectors/product markets. The database is populated with EPA-generated estimates used in Environmental Impact Assessment studies conducted by the Agency since 1990, as well as estimates found in the economics literature. It can be accessed from the Technology Transfer Network Economics and Cost Analysis Support website: http://www.epa.gov/ttnecas1/Elasticity.htm.

When such information is unavailable, as is often the case for intermediate goods, elasticities may be quantitatively or qualitatively assessed.[20] Econometric tools are frequently used to estimate supply and demand equations (thereby the elasticities) and the factors that influence them.

A.4.2 Measuring the Welfare Effect of a Change in Environmental Goods

As introduced in Section A.1 changes in consumer surplus are measured by the trapezoidal region below the ordinary, or Marshallian, demand curve as price changes. This region reflects the benefit a consumer receives by being able to consume more of a good at a lower price. If the price of a good decreases, some of the consumer's satisfaction comes from being able to consume more of a commodity when its price falls, but some of it comes from the fact that the lower price means that the consumer has more income to spend. However, the change in (Marshallian) consumer surplus only serves as a monetary measure of the welfare gain or loss experienced by the consumer under the strict assumption that the marginal utility of income is constant.[21] This assumption is almost never true in reality. Luckily, there are alternative, less demanding monetary measures of consumer welfare that prove useful in treatments of BCA. Intuitively, these measures determine the size of payment that would be necessary to compensate the consumer for the price change. In other words, they estimate the consumer's WTP for a price change.

As mentioned above, a price decline results in two effects on consumption. The change in relative prices will increase consumption of the cheaper good (the substitution effect), and consumption will be affected by the change in overall purchasing power (the income effect). A Marshallian demand curve reflects both substitution and income effects. Movements along it show how the quantity demanded changes as price changes (holding all other prices and income constant), so it reflects both the substitution and the income effects. The Hicksian (or "compensated") demand curve, on the other hand, shows the relationship between quantity demanded of a commodity and its price, holding all other prices and *utility* (rather than income) constant. This is the correct measure of a consumer's WTP for a price change. The Hicksian demand curve is constructed by adjusting income as the price changes so as to keep the consumer's utility the same at each point on the curve. In this way, the income effect of a price change is eliminated and the substitution effect can be considered alone. Movements along the Hicksian demand function can be used to determine the monetary change that would compensate the consumer for the price change.

Hicks (1941) developed two correct monetary measures of utility change associated with a price change: compensating variation and equivalent variation. *Compensating variation* (CV) assesses how much money must be taken away from consumers after a price decrease occurred to return them to the original utility level. It is equal to the amount of money that would 'compensate' the consumer for the price decrease. *Equivalent variation* (EV) measures how much money would need to be given to the consumer to bring her to the higher utility level instead of introducing the price change. In other words, it is the monetary change that would be 'equivalent' to the proposed price change.

Before examining the implications of these measures for valuing environmental changes, it is useful to understand CV and EV in the case of a reduction in the price of some normal, private good, C_1.[22] This is shown with indifference curves and a budget line, as seen in Figure A.7.

Assume that the consumer is considering the trade-off between C_1 and all other goods, denoted by a composite good, C_2. The indifference curve, U_0, depicts the different combinations of the two goods that yield the same level of utility. Because of

20 Final goods are those that are available for direct use by consumers and are not utilized as inputs by firms in the process of production. Goods that contribute to the production of a final good are called intermediate goods. It is of course possible for a good to be final from one perspective and intermediate from another (Pearce 1992).

21 See Perman et al. (2003), Just et al. (2005) or any graduate level text for a more thorough exposition of this issue.

22 The notation and discussion in this section follow Chapter 12 of Perman et al. (2003).

Figure A.7 - Indifference Curve

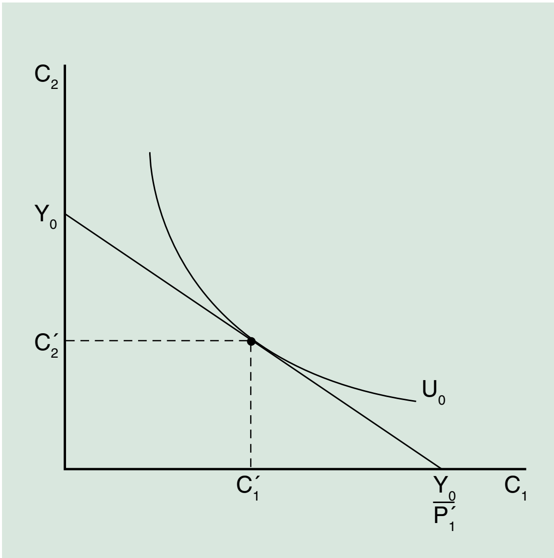
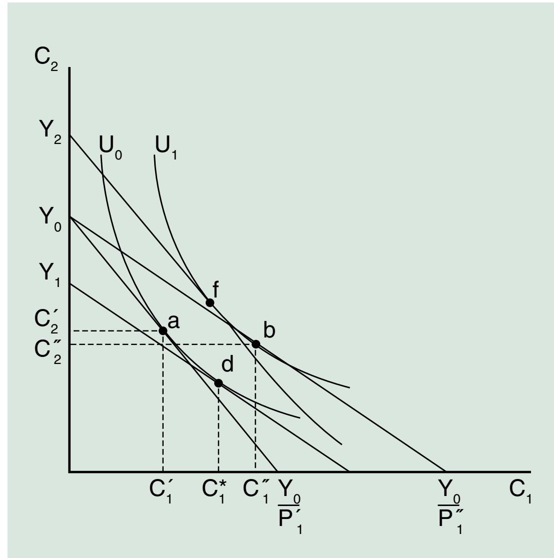


Figure A.8 - Change in Optimal Consumption Bundle



diminishing marginal utility, the curve is concave, where increasing amounts of C_1 must be offered for each unit of C_2 given up to keep the consumer indifferent. The budget line on the graph reflects what the consumer is able to purchase given her income, Y_0, and the prices of the two goods — P_1' and P_2', respectively.[23] A utility-maximizing consumer will choose quantities C_1'and C_2', the point where the indifference curve is tangent to the budget constraint.[24]

Figure A.8 shows the change in the optimal consumption bundle resulting from a reduction in the price of C_1. If the price of C_1 falls, the budget line shifts out on the C_1 axis because more C_1 can be purchased for a given amount of money. The consumer now chooses C_1" and C_2" at point *b* and moves to a new, higher utility curve, U_1. CV then measures how much money must be taken away at the new prices to return the consumer to the old utility level. That is, starting at point b and keeping the slope of the budget line fixed at the new level, by how much must it be shifted downward to make it tangent to the initial indifference curve, U_0? It is, therefore, the maximum amount the consumer would be willing to pay to have the price fall occur — i.e., the precise monetary measure of the welfare change.[25] In Figure A.8, CV is simply given by the amount $Y_0 - Y_1$. EV, on the other hand, measures how much income must be given to the individual at the old price set to maintain the same level of well-being as if the price change did occur. That is, keeping the slope of the budget line fixed at the old level, by how much must it be shifted upwards to make it tangent to U_1? EV is, then, the minimum amount of money the consumer would accept in lieu of the price fall. This too is a proper monetary measure of the utility change resulting from the price decrease. In Figure A.8 then EV is the amount $Y_2 - Y_0$, leaving the individual at point *f*.

CV and EV are simply measures of the distance between the two indifference curves. However, the amount of money associated with CV, EV, and Marshallian consumer surplus (MCS) is generally not the same. For a price fall, it can be shown that CV < MCS < EV, and for a price increase, CV > MCS > EV.[26] Notice that in the case of a price decrease, the CV measures the consumer's willingness to pay (WTP) to receive the price reduction and EV measures the consumer's

23 In Figure A.7, C_2 is considered the numeraire good (i.e., prices are adjusted so that P_2' is equal to 1).

24 For a review of the utility maximizing behavior of consumers, see any general microeconomics textbook.

25 In Figure A.8, this would result in a shift from C_1" to C_1*. This is known as the *income effect* of the price change. The shift from C_1' to C_1* is considered the *substitution effect*.

26 This can be seen by redrawing Figure A.8 using a graph of Marshallian and Hicksian demand curves. See Perman et al. (2003) for a detailed explanation.

willingness to accept (WTA) to forgo the lower price. If the price of C_1 were to increase, then the relationships between WTP/WTA and CV/EV would be reversed. CV would measure the consumer's WTA to suffer the price increase and EV would be the individual's WTP to avoid the increase in price.

In order to examine the implications of these measures for valuing changes in environmental conditions, one can think of C_1 in the above discussion as an environmental commodity, henceforth denoted by E. Then an improvement in environmental quality (or an increase in an environmental public good) resulting from some policy is reflected by an increase in the amount of E. Holding all else constant, such an increase is equivalent to a decrease in the price of E and can be depicted as a shifting outward of the budget line along the E axis.

Welfare changes due to an increase in E follow along the lines of the previous discussion. However, because E is generally non-exclusive and non-divisible, the consumer consumption level cannot be adjusted. Therefore, the associated monetary measures of the welfare change are not technically CV and EV, but are referred to as *compensating surplus* (CS) and *equivalent surplus* (ES). In practice, however, the process is the same; a Hicksian demand curve is estimated for the unpriced environmental good. Analogous to the preceding discussion, if there is an environmental improvement, then CS measures the amount of money the consumer would be willing to pay for the improvement that would result in the pre-improvement level of utility. For the purposes of environmental valuation, this is the primary measure of concern when considering environmental improvements. ES measures how much society would have to pay the consumer to give him the same utility as if the improvement had occurred. In other words, this is how much he would be willing to accept to not experience the gain in environmental quality. If valuing an environmental degradation, then CS measures the WTA and ES measures WTP.

Whereas statements can be made about the relative size of CV, EV, and MCS for price changes of normal goods, Bockstael and McConnell (1993) find that it is not possible to make similar statements about CS, ES, and MCS for a change in environmental quality.[27] Given that environmental quality is generally an unpriced public good, ordinary Marshallian demand functions cannot be estimated, so it may seem irrelevant that one cannot say anything about how MCS approximates the proper measure. However, Bockstael and McConnell's results are important in relation to indirect methods for environmental valuation. However, most indirect valuation studies are based on Marshallian demand functions in practice, in the hope of keeping the associated error small.

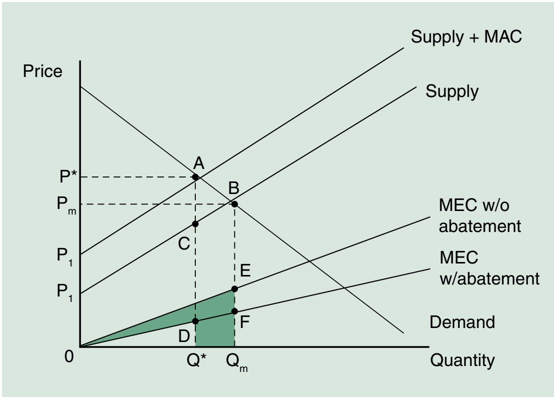
A.4.3 Single Market, Multi-Market, and General Equilibrium Analysis

Both supply and demand elasticities are affected by the availability of close complements and substitutes. This highlights the fact that regulating one industry can have an impact on other, non-regulated markets. However, this does not necessarily imply that all of these other markets must be modeled. Changes due to government regulation can be captured using only the equilibrium supply and demand curves for the affected market, assuming: (1) there are small, competitive adjustments in all other markets; and (2) there are no distortions in other markets. This is referred to as *partial equilibrium analysis*.

For example, suppose a new environmental regulation increases per unit production costs. The benefits and costs of abatement in a partial equilibrium setting are illustrated in Figure A.9 where the market produces the quantity Q_m in equilibrium without intervention. The external costs of production are shown by the marginal external costs (MEC) curve without

27 Willig (1976) shows that ordinary, or Marshallian, demand curves can provide an approximate measure of welfare changes resulting from a price change. In most cases the error associated with using MCS, with respect to CV or EV, will be less than 5 percent (see Perman et al. 2003).

Figure A.9 - Benefits and Costs of Abatement



any abatement. Total external costs are given by the area under the MEC curve up to the market output, $Q_{m,}$ or the area of triangle Q_mE0.

With required abatement production, costs are the total of supply plus marginal abatement costs (MAC), shown as the new, higher supply curve in the figure. These higher costs result in a new market equilibrium quantity shown as Q^*. The social cost of the requirement is the resulting change in consumer and supplier surplus, shown here as the total observed abatement costs (parallelogram P_0P_1AC) plus the area of triangle ABC, which can be described as deadweight loss.

Abatement also produces benefits by shifting the MEC curve downward, reflecting the fact that each unit of production now results in less pollution and social costs. Additionally, the reduced quantity of the output good results in reduced external costs. The reduced external costs, i.e., the benefits, are given by the difference between triangle Q_mE0 and triangle Q^*D0, represented by the shaded area in the figure.

The net benefits of abatement are the benefits (the reduced external costs) minus the costs (the loss in consumer and producer surplus). In the figure this would equal the shaded area (the benefits) minus total abatement costs and deadweight loss as described above.

While the single market analysis is theoretically possible, it is generally impractical for rulemaking. As mentioned in Section A.3, this is often because the gains occur outside of markets and cannot be linked directly to the output of the regulated market. Therefore BCA is frequently done as two separate analyses: a benefits analysis and a cost analysis.

When a regulation is expected to have a large impact outside of the regulated market, then the analysis should be extended beyond that market. If the effects are significant but not anticipated to be widespread, one potential improvement is to use multi-market modeling in which vertically or horizontally integrated markets are incorporated into the analysis. The analysis begins with the relationship of input markets to output markets. A multi-market analysis extends the partial equilibrium analysis to measuring the losses in other related markets.[28]

In some cases, a regulation can have such a significant impact on the economy that a general equilibrium modeling framework is required.[29] This may be because regulation in one industry has broad indirect effects on other sectors, households may alter their consumption patterns when they encounter increases in the price of a regulated good, or there may be interaction effects between the new regulation and pre-existing distortions, such as taxes on labor. In these cases, partial equilibrium analyses are likely to result in an inaccurate estimation of total social costs. Using a general equilibrium framework accounts for linkages between all sectors of the economy and all feedback effects, and can measure total costs comprehensively.[30]

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28 An example of the use of multi-market model for environmental policy analysis is contained in a report prepared for EPA on the regulatory impact of control on asbestos and asbestos products (U.S. EPA 1989).

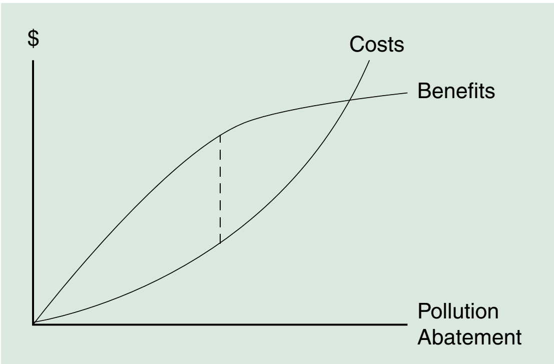
29 *General equilibrium analysis* is built around the assumption that, for some discrete period of time, an economy can be characterized by a set of equilibrium conditions in which supply equals demand in all markets. When this equilibrium is "shocked" through a change in policy or a change in some exogenous variable, prices and quantities adjust until a new equilibrium is reached. The prices and quantities from the post-shock equilibrium can then be compared with their pre-shock values to determine the expected impacts of the policy or change in exogenous variables.

30 Chapter 8 provides a more detailed discussion of partial equilibrium, multi-market, and general equilibrium analysis.

A.5 Optimal Level of Regulation

Following from the definition in Section A.1, the most economically efficient policy is the one that allows for society to derive the largest possible social benefit at the lowest social cost. This occurs when the *net* benefits to society (i.e., total benefits minus total costs) are maximized. In Figure A.10, this is at the point where the distance between the benefits curve and the costs curve is the largest and positive.

Figure A.10 - Maximized Net Benefits



Note that this is *not* necessarily the point at which:

- Benefits are maximized;
- Costs are minimized;
- Total benefits = total costs (i.e., benefit-cost ratio = 1);
- Benefit-cost ratio is the largest; or
- The policy is most cost-effective.

If the regulation were designed to maximize benefits, then any policy, no matter how expensive, would be justified if it produced any benefit, no matter how small. Similarly, minimizing costs would, in most cases, simply justify no action at all. A benefit-cost ratio equal to one is equivalent to saying that the benefits to society would be exactly offset by the cost of implementing the policy. This implies that society is indifferent between no regulation and being regulated; hence, there would be no net benefit from adopting the policy. Maximizing the benefit-cost ratio is not optimal either. Two policy options could yield equivalent benefit-cost ratios but have vastly different net benefits. For example, a policy that cost $100 million per year but produced $200 million in benefits has the same benefit-cost ratio as a policy that cost $100,000 but produced $200,000 in benefits, even though the first policy produces substantially more net benefit for society.[31] Finally, finding the most cost-effective policy has similar problems because the cost-effectiveness ratio can be seen as the inverse of the benefit-cost ratio. A policy is cost effective if it meets a given goal at least cost — i.e., minimizes the cost per unit of benefit achieved. Cost-effectiveness analysis (CEA) can provide useful information to supplement existing BCA and may be appropriate to rank policy options when the benefits are fixed and cannot be monetized, but it provides no guidance in setting an environmental standard or goal.

Conceptually, net social benefits will be maximized if regulation is set such that emissions are reduced up to the point where the benefit of abating one more unit of pollution (i.e., marginal social benefit)[32] is equal to the cost of abating an additional unit (i.e., marginal abatement cost).[33] If the marginal

31 Benefit-cost ratios are useful when choosing one or more policy options subject to a budget constraint. For example, consider a case where five options are available and the budget is $1,000. The first option will cost $1,000 and will deliver benefits of $2,000. Each of the other four will cost $250 and deliver benefits of $750. If options are selected according to the net benefits criterion, the first option will be selected, because its net benefits are $1,000 while the net benefits of each of the other options are $500. However if options are selected by the benefit-cost ratio criterion, the other four options will be selected, as each of their benefit-cost ratios equal 3, versus a benefit-cost ratio of 2 for the first option. In this case, choosing options by the net benefits criterion will yield $1,000 in total net benefits, while choosing options by the benefit-cost ratio criterion will yield $500 in total net benefits. In most cases, choosing options in decreasing order of benefit-cost ratios will yield the largest possible net benefits given a fixed budget. This method will guarantee the optimal solution if the benefits and costs of each option are independent, and if each option can be infinitely subdivided: simply select the options in decreasing order of their benefit-cost ratios and once the budget is exceeded subdivide the last option selected such that the budget constraint is met exactly (see Dantzig 1957). Also note that this strategy does not require measuring benefits and costs in the same units, which means that it is directly useful for CEA (Hyman and Leibowitz 2000), while the net-benefit criterion is not.

32 The benefits of pollution reduction are the reduced damages from being exposed to pollution. Therefore, the marginal social benefit of abatement is measured as the additional reduction in damages from abating one more unit of pollution.

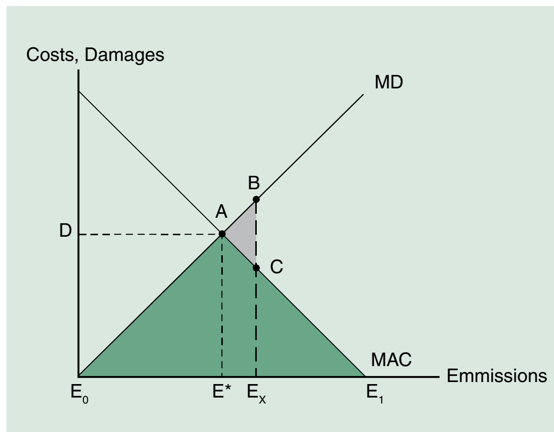
33 The idea that a given level of abatement is efficient — as opposed to abating until pollution is equal to zero — is based on the economic concept of diminishing returns. For each additional unit of abatement, marginal social benefits decrease while marginal social costs of that abatement increase. Thus, it only makes sense to continue to increase abatement until the point where marginal abatement benefits and marginal costs are just equal. Any abatement beyond that point will incur more additional costs than benefits. (Alternatively, one can understand the efficient level of abatement as the amount of regulation that achieves the efficient level of pollution. If one considers a market for pollution, the socially-efficient outcome would be the point where the marginal WTP for pollution equals the marginal social cost of polluting.)

benefits are greater than the marginal costs, then additional reductions in pollution will offer greater benefits than costs, and society will be better off. If the marginal benefits are less than marginal costs, then additional reductions in pollution will cost society more than they provide in benefits, and will make society worse off. When the marginal cost of abatement is equal to society's marginal benefit, no gains can be made from changing the level of pollution reduction, and an efficient aggregate level of emissions is achieved. In other words, *a pollution reduction policy is at its optimal, most economically efficient point when the marginal benefits equal the marginal costs of the rule.*[34]

The condition that marginal benefits must equal marginal costs assumes that the initial pollution reduction produces the largest benefits for the lowest costs. As pollution reduction is increased (i.e., regulatory stringency is increased), the additional benefits decline and the additional costs rise. While it is not always true, a case can be made that the benefits of pollution reduction follow this behavior. The behavior of total abatement costs, however, will depend on how the pollution reduction is distributed among the polluters since firms may differ in their ability to reduce emissions. The aggregate marginal abatement cost function shows the least costly way of achieving reductions in emissions. It is equal to the horizontal sum of the marginal abatement cost curves for the individual polluters. Although each firm faces increasing costs of abatement, marginal cost functions still vary across sources. Some firms may abate pollution relatively cheaply, while others require great expense. To achieve economic efficiency, the lowest marginal cost of abatement must be achieved first, and then the next lowest. Pollution reduction is achieved at lowest cost only if firms are required to make equiproportionate cutbacks in emissions. That is, at the optimal level of regulation, the cost of abating one more unit of pollution is equal across all polluters.[35]

Figure A.11 illustrates why the level of pollution that sets the marginal benefits and marginal costs of abatement equal to each other is efficient.[36] Emissions are drawn on the horizontal axis and increase from left to right. The damages from emissions are represented by the marginal damage (MD) curve. Damages may include the costs of worsened human health, reduced visibility, lower property values, and loss of crop yields or biodiversity. As emissions rise, the marginal damages increase. E_1 represents the amount of emissions in the absence of regulation on firms. The costs of controlling emissions are represented by the marginal abatement cost curve (MAC). As emissions are reduced below E_1, the marginal cost of abatement rises.

Figure A.11 - Efficient Level of Pollution



The total damages associated with emissions level E^* are represented by the area of the triangle AE_0E^*, while the total abatement costs are represented by area AE_1E^*. The total burden on

34 It is important to reemphasize the word "marginal" in this statement. Marginal, in economic parlance, means the extra or next unit of the item being measured. If regulatory options could be ranked in order of regulatory stringency, then marginal benefits equal to marginal costs means that the additional benefits of increasing the regulation to the next degree of stringency is equal to the additional cost of that change.

35 Thus a regulation that requires all firms to achieve the same level of reduction will probably result in different marginal costs for each firm and not be efficient. (See Field and Field 2005 or any other environmental economics text for a detailed explanation and example.)

36 Figure A.11 illustrates the simplest possible case, where the pollutant is a flow (i.e., it does not accumulate over time) and marginal damages are independent of location. When pollution levels and damages vary by location, then the efficient level of pollution is reached when marginal abatement costs adjusted by individual transfer coefficients are equal across all polluters. Temporal variability also implies an adjustment to this equilibrium condition. In the case of a stock pollutant, marginal abatement costs are equal across the discounted sum of damages from today's emissions in all future time periods. In the case of a flow pollutant, this condition should be adjusted to reflect seasonal or daily variations (see Sterner 2003).

society of this level is equal to the total abatement costs of reducing emissions from E_1 to E^* plus the total damages of the remaining emissions, E^*. That is, the total burden is the darkly shaded triangle, E_0AE_1.

Now assume that emissions are something other than E^*. For example, suppose emissions were E_X, which is greater than E^*. Total damages for this level of emissions are equal to the area of the triangle BE_0E_x, while total costs of abatement to this level is equal to the area CE_xE_1. The total burden on society of this level is the sum of the areas of the darkly shaded and the lightly shaded triangles. This means that the excess social cost of choosing emissions E_X rather than E^* is equal to the area of the lightly shaded triangle, *ABC*. A similar analysis could be done if emissions levels were below level, E^*. Here, the additional abatement costs would be greater than the decrease in damages, resulting in excess social costs. The policy that sets the emissions level at E^* — at the point where marginal benefits of pollution reduction (represented by the MD curve) and the MAC curve intersect — is economically efficient because it imposes the least net cost on, and yields the highest net benefits for, society. That is, the triangle E_0AE_1 is the smallest shaded region that can be obtained.

This section has focused on first-best optimal regulation when there are no pre-existing market distortions. However, it is important to note that realizable policy outcomes will often be "second best" due to information constraints, political constraints, imperfect competition, and market distortions created by tax and other government interventions. For example, many of the emissions-based policies emphasized in these *Guidelines* may be less feasible for addressing nonpoint source pollution, such as agriculture, which is less observable and more stochastic than emissions from point sources. Agriculture is also subject to multiple non-environmental policy distortions that must be considered in the measurement of the social benefits and costs of regulating agriculture.

A.6 Conclusion

The purpose of this appendix is to present a brief explanation of some of the fundamental economics relevant to Chapters 3 through 9. It is not intended to provide a comprehensive discussion of all microeconomic theory and its application to environmental issues. The interested reader can turn to undergraduate or graduate level textbooks for a more thorough exposition of the topics covered here. At the undergraduate level, Field and Field (2005) provide an introduction to the basic principles of environmental economics. Tietenberg's (2002) and Perman et al.'s (2003) presentations are more technical but still used primarily for undergraduate courses. Freeman (2003) is the standard text for graduate courses in environmental economics and deals with the methodology of non-market valuation. Supplemental texts that provide a good handle on environmental economics with less technical detail include Stavins (2000a), and Portney and Stavins (2000). Finally, general microeconomics textbooks (Mankiw 2004, and Varian 2005 at the undergraduate level; and Mas-Colell et al. 1995, Kreps 1990, and Varian 2005 at the graduate level), and applied welfare economics textbooks (Just et al. 2005) are useful references as well.

Appendix B
Mortality Risk Valuation Estimates

Some EPA policies are designed to reduce the risk of contracting a potentially fatal health effect such as cancer. Reducing these risks of premature death provides welfare increases to those individuals affected by the policy. These policies generally provide marginal changes in relatively small risks. That is, these policies do not provide assurance that an individual will not die prematurely from environmental exposures; rather, they marginally reduce the probability of such an event. For BCA, analysts generally aggregate these small risks over the affected population to derive the number of statistical lives saved (or the number of statistical deaths avoided) and then use a "value of statistical life" (VSL) to express these benefits in monetary terms.

The risk reductions themselves can generally be classified according to the characteristics of the risk in question (e.g., voluntariness or controllability) and the characteristics of the affected population (e.g., age and health status). These dimensions may affect the *value* of reducing mortality risks. Ideally the VSL would account for all possible risk and demographic characteristics that matter. 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 exposed population in the area and would reflect the changes in life expectancy that would result. Unfortunately, time and resource constraints make it difficult if not impossible to obtain such unique valuation estimates for each EPA policy. Instead, analysts need to draw from existing VSL estimates obtained using well-established methods (see Chapter 7).

This appendix describes the default VSL estimate currently used by the Agency and its derivation, as well as how analysts should characterize and assess benefit transfer issues that may arise in its application. Benefit transfer considerations that are common to all valuation applications, including the effect of most demographic characteristics of the study and policy populations, are described in Chapter 7 Section 7.3 and will not be repeated here.

B.1 Central Estimate of VSL

Table B.1 contains the VSL estimates that currently form the basis of the Agency's recommended central VSL estimate. Fitting a Weibull distribution to these estimates yields a central estimate (mean) of $7.4 million ($2006) with a standard deviation of $4.7

Table B.1 - Value of Statistical Life Estimates (mean values in millions of 2006 dollars)

Study	Method	Value of Statistical Life
Kniesner and Leeth (1991 - US)	Labor Market	$0.85
Smith and Gilbert (1984)	Labor Market	$0.97
Dillingham (1985)	Labor Market	$1.34
Butler (1983)	Labor Market	$1.58
Miller and Guria (1991)	Contingent Valuation	$1.82
Moore and Viscusi (1988)	Labor Market	$3.64
Viscusi, Magat, and Huber (1991)	Contingent Valuation	$4.01
Marin and Psacharopoulos (1982)	Labor Market	$4.13
Gegax et al. (1985)	Contingent Valuation	$4.86
Kniesner and Leeth (1991 - Australia)	Labor Market	$4.86
Gerking, de Haan, and Schulze (1988)	Contingent Valuation	$4.98
Cousineau, Lecroix, and Girard (1988)	Labor Market	$5.34
Jones-Lee (1989)	Contingent Valuation	$5.59
Dillingham (1985)	Labor Market	$5.71
Viscusi (1978)	Labor Market	$6.07
R.S. Smith (1976)	Labor Market	$6.80
V.K. Smith (1983)	Labor Market	$6.92
Olson (1981)	Labor Market	$7.65
Viscusi (1981)	Labor Market	$9.60
R.S. Smith (1974)	Labor Market	$10.57
Moore and Viscusi (1988)	Labor Market	$10.69
Kniesner and Leeth (1991 - Japan)	Labor Market	$11.18
Herzog and Schlottman (1987)	Labor Market	$13.36
Leigh and Folsom (1984)	Labor Market	$14.21
Leigh (1987)	Labor Market	$15.31
Garen (1988)	Labor Market	$19.80

Derived from U.S. EPA (1997a) and Viscusi (1992). Updated to 2006$ with GDP deflator.

million.[1, 2] EPA recommends that the central estimate, updated to the base year of the analysis, be used in all benefits analyses that seek to quantify mortality risk reduction benefits.

This approach was vetted and endorsed by the Agency when the 2000 *Guidelines for Preparing Economic Analyses* were drafted.[3] It remains EPA's default guidance for valuing mortality risk changes although the Agency has considered and presented alternatives.[4]

1 The VSL was updated from the $4.8 million ($1990) estimate referenced in the 2000 *Guidelines* by adjusting the individual study estimates for inflation using a GDP deflator and then fitting a Weibull distribution to the estimates. The updated Weibull parameters are: location = 0, scale = 7.75, shape = 1.51 (updated from location = 0; scale = 5.32; shape = 1.51). The Weibull distribution was determined to provide the best fit for this set of estimates. See U.S. EPA 1997a for more details.

2 This VSL estimate was produced using the GDP deflator inflation index. Some economists prefer using the Consumer Price Index (CPI) in some applications. The key issue for EPA analysts is to ensure that the chosen index is used consistently throughout the analysis.

3 The studies listed in Table B.1 were published between 1974 and 1991, and most are hedonic wage estimates that may be subject to considerable measurement error (Black et al. 2003, and Black and Kniesner 2003). Although these were the best available data at the time, they are sufficiently dated and may rely on obsolete preferences for risk and income. The Agency is currently considering more recent studies as it evaluates approaches to revise its guidance.

4 EPA is in the process of revisiting this guidance and has recently engaged the SAB-EEAC on several issues including the use of meta-analysis as a means of combining estimates and approaches for assessing mortality benefits when changes in longevity may vary widely (U.S. EPA 2006d). The Agency is committed to using the best available science in its analyses and will revise this guidance in response to SAB recommendations (see U.S. EPA 2007g for recent SAB recommendations).

B.2 Other VSL Information

For most of mortality risk reductions EPA uniformly applies the VSL estimate discussed above. For a period of time (2004-2008), the Office of Air and Radiation (OAR) valued mortality risk reductions using a VSL estimate derived from a limited analysis of some of the available studies. OAR arrived at a VSL using a range of $1 million to $10 million (2000$) consistent with two meta-analyses of the wage-risk literature. The $1 million value represented the lower end of the interquartile range from the Mrozek and Taylor (2002) meta-analysis of 33 studies. The $10 million value represented the upper end of the interquartile range from the Viscusi and Aldy (2003) meta-analysis of 43 studies. The mean estimate of $5.5 million (2000$) was also consistent with the mean VSL of $5.4 million estimated in the Kochi et al. (2006) meta-analysis. However, the Agency neither changed its official guidance on the use of VSL in rulemakings nor subjected the interim estimate to a scientific peer-review process through the Science Advisory Board (SAB) or other peer-review group.

During this time, the Agency continued work to update its guidance on valuing mortality risk reductions. EPA commissioned a report from meta-analytic experts to evaluate methodological questions raised by EPA and the SAB on combining estimates from the various data sources. In addition, the Agency consulted several times with the SAB Environmental Economics Advisory Committee (SAB-EEAC) on the issue. With input from the meta-analytic experts, the SAB-EEAC advised the Agency to update its guidance using specific, appropriate meta-analytic techniques to combine estimates from unique data sources and different studies, including those using different methodologies such as wage-risk and stated preference (U.S. EPA 2007g).

Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently best reflects the SAB-EEAC advice received to date. Therefore, the VSL described above that was vetted and endorsed by the SAB should be applied in relevant analyses while the Agency continues its efforts to update its guidance on this issue.

B.3 Benefit Transfer Considerations

Policy analysts valuing mortality risk reductions should account for differences in risk and population characteristics between the policy and study scenarios and their potential effect on the overall results. The ultimate objective of the benefit transfer exercise is to account 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 correcting for some relevant factors, but not for others, recognizing that it may not be feasible to account for all factors.

B.4 Adjustments Associated with Risk Characteristics

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:[5]

- voluntary/involuntary
- ordinary/catastrophic
- delayed/immediate
- natural/man-made
- old/new
- controllable/uncontrollable
- necessary/unnecessary
- occasional/continuous

Transferring VSL estimates among these categories may introduce bias. There have been some recent efforts attempting to quantitatively assess these sources of bias.[6] These studies generally conclude that voluntariness, control and responsibility affect individual values for safety, although there is no consensus on the direction and magnitude of these effects.

5 A review of issues in risk perception is found in Lichtenstein and Slovic (2006). Other informative sources include Slovic (1987), Rowe (1977), Otway (1977), and Fischoff et al. (1978).

6 Examples include Hammitt and Liu (2004), Sunstein (1997), Mendeloff and Kaplan (1990), McDaniels et al. (1992), Savage (1993), Jones-Lee and Loomes (1994, 1995, 1996), and Covey et al. (1995).

In addition, 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. As part of the Agency's review of our mortality risk guidance we are evaluating the literature from which the studies are drawn.

Support for quantitative adjustments in the empirical literature is lacking for most of these factors. The SAB reviewed an Agency summary of the available empirical literature on the effects of risk and population characteristics on WTP for mortality risk reductions (U.S. EPA 2000d). The SAB review concludes 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. The review supports making the following adjustments to primary benefits estimates: (1) adjusting WTP estimates to account for higher future income levels, though not for cross-sectional differences in income; and (2) discounting risk reductions that are brought about in the future by current policy initiatives (that is, after a cessation lag), using the same rates used to discount other future benefits and costs. All other adjustments, if made, should be relegated to sensitivity analyses.

Increases in income over time. The economics literature shows that the income elasticity of WTP to reduce mortality risk is positive, based on cross-sectional data. As a result, benefits estimates of reduced mortality risk accruing in future years may be adjusted to reflect anticipated income growth, using the range of income elasticities (0.08, 0.40 and 1.0) employed in *The Benefits and Costs of the Clean Air Act, 1990-2010.*[7] Recent EPA analyses have assumed a triangular distribution from these values and used the results in a probabilistic assessment of benefits.[8] At the time of this writing, EPA is engaged in a consultation with the SAB-EEAC on the appropriate range of income elasticities and will update this guidance as needed.

7 For details see Kleckner and Neuman (2000).

8 See, for example, pp. 6-84 of the Final Economic Analysis for the Stage 2 DBPR (U.S. EPA 2005a).

Timing of reduced exposure and reduced risk. Many environmental policies are targeted at reducing the risk of effects such as cancer, where there may be an extended period of time between the reduced exposure and the reduction in the risk of death from the disease.[9] This delay between the change in exposure and realization of the reduced risk may affect the value of that risk reduction. Most existing VSL estimates are based on risks of relatively immediate fatalities making them an imperfect fit for a benefits analysis of many environmental 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.

B.5 Effects on WTP Associated with Demographic Characteristics

Two population characteristics are particularly noteworthy for their potential effect on mortality risk valuation estimates: age and health status of the exposed population. In September 2006, the Agency requested an additional advisory from the SAB-EEAC on issues related to valuing changes in life expectancy for which age and baseline health status are close correlates.[10] Because the outcome of this review is not yet available, we focus here on previous advice received from the SAB on related questions.

Age. It has sometimes been posited that older individuals should have a lower WTP for changes in mortality risk given the fewer years of life expectancy remaining compared to younger individuals. This hypothesis may be confounded, however, 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). Several authors have attempted to explore

9 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.

10 U.S. EPA (2006d) summarizes much of the literature related to the effects of age and health status on WTP for changes in mortality risk and includes the charge questions put to the SAB-EEAC on these issues.

potential differences in mortality risk valuation estimates associated with differences in the average age of the affected population using theoretical models of life-cycle consumption.[11] In general this literature has shown that the relationship between age and WTP for mortality risk changes is ambiguous, requiring strong assumptions to even sign the relationship.[12] Empirical evidence is also mixed. A number of empirical studies (mostly hedonic wage studies) suggest that the VSL follows a consistent "inverted-U" life-cycle, peaking in the region of mean age.[13] Others find no such statistically significant relationship and still others show WTP increasing with age.[14] Stated preference results are also mixed, with some studies showing declining WTP for older age groups and others finding no statistically significant relationship between age and WTP.[15]

In spite of the ambiguous relationship between age and WTP, two alternative adjustment techniques have been derived from this literature. The first technique, *value of statistical life-years (VSLY)*, is derived by dividing the estimated VSL by expected remaining life expectancy. This is by far the most common approach and presumes that: (1) the VSL 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 and prospective studies of the costs and benefits of the Clean Air Act (U.S. EPA 1997a, and U.S. EPA 1999).

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.[16]

Neither approach enjoys general acceptance in the literature as they both require large assumptions to be made, some of which have been contradicted in empirical studies. Since published support is lacking, neither approach is recommended at this time.

Analysts are advised to note the age distribution of the affected population when possible, especially when children are found to be a significant portion of the affected population.[17] Although the literature on the valuation of children's health risks is growing, there is still not enough information currently to derive age-specific valuation estimates.

Health status. Individual health status may also affect WTP for mortality risk reduction. This is an especially 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. Again, both the theoretical and empirical literatures on this point are mixed with some studies showing a declining WTP for increased longevity with a declining baseline health state (Desvousges et al. 1996) and other

11 See, for example, Shepard and Zeckhauser (1982), Rosen (1988), Cropper and Sussman (1988, 1990), and Johannson (2002).

12 See Evans and Smith (2006) for a recent summary.

13 See Jones-Lee et al. (1985), Aldy and Viscusi (2008), Viscusi and Aldy (2007a and b), and Kniesner et al. (2006).

14 Viscusi and Aldy (2003) review more than 60 studies of mortality risk estimates from 10 countries and discuss eight hedonic wage studies that explicitly examine the age-WTP relationship. Only five of the eight studies found a statistically significant, negative relationship between age and the return to risk. Smith et al. (2004) and Kniesner et al. (2006) find that WTP increases with age.

15 Krupnick et al. (2002) report that WTP for mortality risk reductions changes significantly with age after age 70. Alberini et al. (2004) find no difference in the WTP for younger age groups and find a 20 percent reduction for those aged 70 and older. However this difference was not statistically significant.

16 This second approach was illustrated in one EPA study (U.S. EPA, 2002d) for valuation of air pollution mortality risks, drawing upon adjustments measured in Jones-Lee et al. (1985).

17 See U.S. EPA (2003a) for more information on the valuation of children's health risks. OMB's *Circular A-4* advises agencies to use estimates of mortality risk valuation for children that are at least as large as those used for adult populations (OMB 2003).

studies showing no statistically significant effects (Krupnick et al. 2002).[18]

Application of existing VSLY approaches implicitly assumes a linear relationship in which each discounted life year is valued equally. As OMB (1996) notes "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. An 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.[19]

B.6 Conclusion

Due to current limitations in the existing economic literature, these *Guidelines* conclude that, for the present time, the appropriate default approach for valuing these benefits is 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, and 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. In addition, EPA will continue to conduct periodic reviews of the risk valuation literature and will reconsider and revise the recommendations in these *Guidelines* accordingly. EPA will seek advice from the SAB as guidance recommendations are revised.

18 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).

19 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 imply support for variation in protection from environmental risks. Another consideration is that existing economic methods may not capture social WTP to reduce health risks. Chapter 9 details how some these considerations may be informed by a separate assessment of equity.

Appendix C

Accounting for Unemployed Labor in Benefit-Cost Analysis

In very rare cases, the implementation of a rule or policy may result in the job implications for the structurally unemployed. This appendix (under development) will review the literature on estimating the value unemployed individuals place on their time and will describe what estimates of the costs of labor are most appropriate for use in regulatory impact analysis (RIA) under this scenario.

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