Guidelines for Preparing Economic Analyses

External Review Draft

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National Center for Environmental Economics
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Acronyms and Abbreviations

AC	annualized costs
ACN	AirControlNET
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
CAAA	Clean Air Act Amendments
CAIR	Clean Air Interstate Rule
CAMR	Clean Air Mercury Rule
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
CFR	Code of Federal Regulations
CGE	computable general equilibrium
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
DOE	Department of Energy
DOT	Department of Transportation
DWL	dead weight loss
EA	economic analysis
EBIT	earnings before interest and taxes
EEAC	Environmental Economics Advisory Committee
EIA	economic impact analysis
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
FTE	full-time equivalent employment
GDP	gross domestic product
GIS	Geographic Information System
HCFC	hydrochlorofluorocarbon
ICR	Information Collection Request
I/O	input-output
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
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
NIOSH	National Institute of Occupational Safety and Health
NOAA	National Oceanic and Atmospheric Administration
NOx	nitrogen oxide
NPDES	National Pollutant Discharge Elimination System
NPV	net present value
OCC	opportunity cost of capital
OECD	Organization for Economic Cooperation and Development
OGC	Office of General Counsel
OLS	ordinary least squares
OMB	Office of Management and Budget
OSHA	Occupational Safety and Health Administration
PACE	Pollution Abatement Costs and Expenditures
PRA	Paperwork Reduction Act
POTW	publicly-owned (wastewater) treatment work
PVC	present value of costs
QA	quality assurance
QALY	quality-adjusted life year
RAPIDS	Rule and Policy Information Development System
RCRA	Resource Conservation and Recovery Act
RDD	random digit dialing
RFA	Regulatory Flexibility Act
RIA	regulatory impact analysis
RP	revealed preference
RUM	random utility model
SAB	Science Advisory Board
SAM	social accounting matrix
S&P	Standard & Poor's
SBA	Small Business Administration
SBREFA	Small Business Regulatory Enforcement Fairness Act
SIC	Standard Industrial Classification
SISNOSE	significant economic impact on a substantial number of small entities
SO_2	sulfur dioxide
SP	stated preference
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
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 one would have to pay at the end of each period *t* to add up to the same cost in present value terms as the stream of costs being annualized. Similarly, the annualized benefit is the amount one 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 which is used for comparison with one or more alternative scenarios. For example, an economic analysis of a policy or regulation compares "the world with the policy or regulation" (the policy scenario) with "the world absent the policy or regulation" (the baseline scenario).

Benefit-cost analysis

A benefit-cost analysis 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 of a policy or action. Economists define benefits by focusing on measures of individual satisfaction or well-being, referred to as measures of welfare or utility. Willingness to pay is the preferred approach to valuing benefits.

Benefit/cost ratio

A benefit/cost ratio is the ratio of the net present value of benefits associated with a project or proposal, relative to the net present value 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 as two projects with the same benefit/cost ratio may 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. These regulations usually impose the same requirements on all sources, although new and existing sources 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 the production process by using different inputs or different mixtures of inputs, or (3) capturing the 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

Cost effectiveness analysis expresses the costs associated with 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, and hence 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 the transaction. The bearers of such 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 and an 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

An economic impact analysis examines the distribution of monetized effects such as changes in profitability or in government revenues, as well as non-monetized effects such as increases in unemployment rates or numbers of plant closures.

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 demand 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 demand falls as price rises, the greater 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, however, 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 supply that occurs in response to a percentage change in price. For many goods the supply can be increased over time by locating alternative sources, investing in an expansion of production capacity, or developing competitive products which 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 time.

Emission taxes
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 may 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]

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 (e.g., 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 involved.

Flow pollutants
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 a flow pollutant.

Hotspot
A hotspot is a geographic area with a high level of pollution/contamination within a larger 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

[1] Definition taken from http://www.epa.gov/compliance/basics/ejbackground.html (Accessed April 17, 2008).

amount the gainers are prepared to accept as a bribe to forgo the change. In other words, the Hicks compensation test is from the losers' point of view, while the Kaldor compensation test is from the gainers' point of view. The Kaldor-Hicks criterion is widely applied in welfare economics and managerial economics. For example, it forms an underlying rationale for cost-benefit analysis.

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
Displacement of pollution from one location to another as a result of the imposition of tighter pollution controls. Under tradable permits, leakages occur when pollution is displaced to an area not affected by the cap.

Marginal benefit
The benefit received from a small 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 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 benefit received by the producer of a good (marginal private benefit) plus the marginal benefit received by other members of society (external benefit).

Marginal social cost
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 system under which sources are required to have emissions permits matching their actual emissions, with each permit specifying how much the firm is allowed to emit and transferable.

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

Meta-analysis
Meta-analysis is a statistical method of combining data from a set of comparable studies of a problem in order to provide a larger sample size for evaluation and to produce 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 net present value, 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 the policy's effects.

Net present value
The net present value 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 individuals may attach to the mere knowledge of the existence of a good or resource, as opposed to enjoying its direct use. It can be motivated for a variety of reasons, including bequest values for future generations, existence values and values of paternalistic altruism for others' enjoyment of the resource.

Opportunity cost
Opportunity cost is the value of the second best alternative to a particular activity or resource. Opportunity cost need not be assessed in monetary terms, but rather can be assessed in terms of anything which is of value to the person or persons doing the assessing. For example, using a grove of trees to produce paper may have as a second best alternative 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 may produce the illusion that its 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)
QALYs are a composite measure used to convert different types of health effects to a common, integrated unit, incorporating both the quality and quantity of life lived in different health states. These metrics are 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 may 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, is 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, in order to encourage certain behavior. For example, polluters may be paid 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 may 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

The sum of all costs associated with a given activity.

Use value

An economic value based on the tangible human use of some environmental or natural resource.

Value of a Statistical Life

The value of a statistical life (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 identifiable life is valued at this amount, but rather that the aggregate value of reducing a collection of small individual risks is worth $5 million in this case.

Value of a Statistical Life Year

The value of a statistical life year (VSLY) is the estimated dollar value for a year of statistical life. In practice, it 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

Willingness to accept (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, willingness to accept is the least amount of money an individual would accept to forego the improvement (or endure the decrement).

Willingness to Pay

Willingness to pay (WTP) is the largest amount of money that an individual or group could pay, along with a change in policy, 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 the environmental effects of concern.

1 Introduction

1.1 Background to the *Guidelines for Performing Economic Analyses*

The Guidelines for Preparing Economic Analyses is part of a continuing effort by the 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* (US EPA 1983) and later revised in the late 1990s. In September of 2000, the EPA issued its *Guidelines for Preparing Economic Analyses* (US EPA 2000) (*EA Guidelines*), 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, the EPA committed to periodically revise the *EA 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 *EA Guidelines*, describes new Executive Orders and recent guidance documents that impose new requirements on analysts, and fills information gaps by providing more expansive information on selected topics. Furthermore, to facilitate the adoption of new information in the future, this document will be released electronically and in a loose-leaf format. This new, more flexible format will allow future updates and additions without requiring a wholesale revision of the document.

While economic analysis may 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, Executive Orders and guidance materials. With a few exceptions, the collection of Executive Orders and statutes that govern the conduct of economic analysis and distributional analysis has remained largely unchanged since 2000. Executive Order 12866 (and its recent amendment EO 13422), directing federal agencies to perform a benefit-cost analysis for economically significant rules (those with an economic impact of $100 million or more), still provides the primary impetus for much of the formal benefit-cost analysis within the Agency.[2] However, new guidance documents and handbooks on how to comply with a number of executive orders 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 guidance documents are referenced in the revised *EA Guidelines* where appropriate.

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[2] EO 13422, a recent amendment to EO 12866, requires 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." See http://www.whitehouse.gov/news/releases/2007/01/20070118.html (accessed October 5, 2007) for more information.

In addition, the *EA Guidelines* have been and will continue to be updated to keep pace with the evolving emphases policy makers place on different economic and social concerns affected by environmental policies. Chapters on the conduct of Cost Effectiveness Analysis and Uncertainty Analysis are under development and will be added upon their completion. These new chapters will assist analysts in complying with OMB's requirements to conduct cost-effectiveness analysis for economically significant rules that have improved human health as their primary benefit and to conduct formal probabilistic uncertainty analysis for rules with economic impacts exceeding $1 billion.

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

Underlying these efforts is the recognition that a thorough and careful economic analysis is an important component in designing sound environmental policies. Preparing high quality economic analysis can greatly enhance the effectiveness of environmental policies by providing policy makers with the ability to systematically assess the consequences of regulatory and non-regulatory (i.e., voluntary) actions. An economic analysis can describe the implications of policy alternatives not just for economic efficiency, but also for 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 may provide useful insights.

It is important to note that economic analysis is but one component in the decision making process and under some statutes 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. Still, economic analysis provides a means to organize information and 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 *EA Guidelines*

The scope of the *EA Guidelines* is on economic analysis typically conducted for environmental policies using regulatory or non-regulatory management strategies. Other guidance documents exist for related analyses, some of which are inputs to economic assessments. No attempt is made here to summarize these other guidance materials. Instead, their existence and content are noted in the appropriate sections.

As with the 2000 *EA Guidelines,* the presentation of economic concepts and applications in this document assumes the reader has some background in microeconomics as applied to environmental and natural resource policies. Thus, to fully understand and apply the approaches and recommendations presented in the *EA Guidelines,* readers should be familiar with basic applied microeconomic analysis, the concepts and measurement of consumer and producer surplus, and the economic foundations of benefit-cost evaluation. Appendix A provides the reader with a brief review of economic foundations and the Glossary defines selected key terms. Persons lacking environmental economics skills, but seeking to better understand the economics, may require additional training or reading.

The *EA Guidelines* are designed to provide assistance to analysts in the economic analysis of environmental policies, but they do not provide a rigid blueprint or a “cookbook” for all policy assessments. The most productive and illuminating approaches for particular situations will depend on a variety of case-specific factors and will require professional judgment. The *EA 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 resources, there is no substitute for reviewing the original source materials.

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

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

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

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 policymaking, 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 to a new state of equilibrium with the regulation in effect. The differences between the old and new states – measured as changes in prices, quantities produced and consumed, income and other economic quantities – can be used to characterize the net welfare changes for each affected group identified in the model. Analysts can rely on different outputs and conclusions from the general equilibrium framework to assess issues of both *efficiency* and *distribution*. At EPA these issues often take the form of three distinct questions:

- Is it theoretically possible for the "gainers" from the policy to fully compensate the "losers" and still remain better off?

- Who are the gainers and losers from the policy and associated economic changes?

- And how did a particular group – especially a group that may be considered to be disadvantaged – fare as a result of the policy change?

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

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

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

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

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

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

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

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

The assumptions and modeling framework developed for the BCA, constrain and limit the estimation techniques used to examine gainers and losers (in an EIA) or to examine impacts on disadvantaged sub-populations (in an equity assessment). To estimate these two categories of impacts we rely on a multiplicity of estimation techniques. The constraints faced by these analyses as well as details regarding estimation techniques are given by Chapter 9.

1.4 Organization of the *EA Guidelines*

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

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

- Chapter 3: Statement of Need for the Proposal provides guidance on procedures and analyses for clearly identifying the environmental problem to be addressed and for justifying Federal intervention to correct it;

- Chapter 4: Regulatory and Non-Regulatory Approaches to Consider discusses the variety of regulatory and non-regulatory approaches analysts and policy makers ought to consider in developing strategies for environmental improvement;

- Chapter 5: 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 Social Costs presents the basic theoretical approach for assessing the social costs of environmental policies and describes how this can be applied in practice;

- Chapter 9: Distributional Analyses: Economic Impact Analyses and Equity Assessment provides guidance for performing a variety of different assessments of the economic impacts and equity effects of environmental policies; and

- Chapter 10: Presentation of Analysis and Results concludes the main body of the *EA Guidelines* with suggestions for presenting the quantified and unquantified results of the various economic analyses to policy makers.

2 Statutory and Executive Order Requirements for Conducting Economic Analyses

Various statutes and executive orders direct agencies to conduct specific types of economic analyses.[3] 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.[4]

The scope of requirements for economic analysis can vary substantially. In some cases, the statute or executive order may contain language that limits its applicability to only those regulatory actions that fall above a threshold in significance or impact. Economic analysis may be useful in determining if a regulatory action exceeds a significance or impact threshold, i.e., if it is in the class of regulatory actions the statute or executive order is targeting. For example, provisions of Executive Order 12866 apply to rules that have an annual effect on the economy of $100 million or more or are otherwise considered "economically significant."[5] If a regulatory action must comply with the requirements of a given statute or executive order, additional economic analysis (e.g., Executive Order 12866 requires an analysis of benefits and costs), procedural steps (e.g., Executive Order 13132 may require consultation with affected State and local governments), or a combination of economic analysis and procedural steps may be required. This chapter describes the general requirements for economic analysis contained in each statute and executive order, and identifies thresholds beyond which a regulatory action must follow additional economic analysis requirements.[6] Requirements of the statutes and executive orders that do *not* necessitate economic analysis are not covered in this chapter.

Guidance for the development of regulatory economic analysis is provided in the Office of Management and Budget's (OMB) recent *Circular A-4* (OMB 2003), which replaces its 1996 *Best Practices* document and its 2000 *OMB Guidelines*. These guidance documents have helped to shape EPA's methodology for analytical and empirical economic analysis as described in EPA's own guidance documents. For each executive order 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 your program's Office of General Counsel (OGC) attorney.[7]

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[3]For the text of each statute and executive order appearing in this chapter and guidance specific to them, or for more information on their implications for EPA rule development generally, 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 executive orders can easily be found using http://usasearch.gov/.

[4]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.

[5] Note that the threshold is defined in terms of annual costs or annual benefits of the option that is proposed or finalized and applies regardless of whether the rule is regulatory or deregulatory in nature. See Section 2.1.1 for a more complete definition.

[6]Note that for some statutes and executive orders, requirements for *proposed* regulatory actions may vary slightly from the requirements for *final* regulatory actions.

[7]See (US EPA 2005b) for more information.

2.1 Executive Orders

2.1.1 Executive Order 12866, "Regulatory Planning and Review" (as amended by Executive Order 13422)

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.

While any one of the four criteria 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 <u>based on the annual costs or benefits of the option that is proposed or finalized</u>. So, if one option poses costs or benefits in excess of $100 million, but the 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 added 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 executive order 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) is required. The requirements for benefit-cost analysis 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 Executive Order 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 alternatives approaches also be assessed (Section 6(a)(3)(C)). Executive Order 13422, issued January 2007, amends Executive Order 12866 and requires analysts to "identify in writing the specific market failure (such as externalities, market power, lack of information) or other specific problem." It also extends the requirement of benefit-cost analyses to "significant" guidance documents.[8]

[8] Note that the following guidance documents are exempted: "Guidance documents on regulations issued in accordance with the formal rulemaking provisions of 5 U.S.C. 556, 557; guidance documents that pertain to a military or foreign affairs function of the United States, other than procurement regulations and regulations involving the import or export of non-defense articles and services; guidance documents on regulations that are limited to agency organization, management, or personnel matters; or any other category of guidance documents exempted by the Administrator of OIRA."

Guidance: Chapters 3 through 8 of this document provide guidance for meeting these requirements. OMB's *Circular A-4* provides guidance to Federal agencies on the development of regulatory analysis of *economically* significant rules as required by Executive Order 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.[9]

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

Threshold: Programs, policies, and activities that have disproportionately high and adverse human health or environmental effects on minority populations, including Native American populations, and/or on low-income populations.

Requirements contingent on threshold: No specific requirements for additional economic analysis; rather, certain procedural steps are required.

Guidance: 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.[10] These materials provide guidance on key terms in the Executive Order. Chapter 9 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 Executive Order 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 Executive Order 13045 (U.S. EPA 1998b). EPA has also prepared the *Children's Health Valuation Handbook* (US EPA 2003b), which discusses special issues related to estimation of the value of health risk reductions to children. Guidance in Chapter 9 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

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

[10] For more information see U.S. EPA 1998a and CEQ 1997.

levels of government." Rules may be considered to impose substantial compliance costs, unless the costs are expressly required by statute or there are federal funds available to cover the State or local governments' compliance costs.

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

Guidance: Interim Guidance on Executive Order 13132: Federalism, January 2001.[11]

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."

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 direct compliance costs of the tribal governments 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.[12] Guidance in Chapter 9 of this document addresses equity analyses focusing on minority populations.

2.1.6 Executive Order 13211, "Energy"

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

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

Guidance: EPA has prepared guidance on what effects might be considered significant. OMB has guidance for implementing Executive Order 13211 as well.[13]

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[11]This document is located at http://intranet.epa.gov/adplibrary/documents/Fedism01.pdf (accessed April 14, 2004, internal EPA document)

[12]Please check the "Action Development Process Library" on EPA's intranet, http://intranet.epa.gov/adplibrary (accessed April 7, 2004, internal EPA document) for the status of this guidance.

[13]U.S. Environmental Protection Agency, *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 Executive Order 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: The EPA must prepare a regulatory flexibility analysis, and comply with a number of procedural requirements to solicit and consider flexible regulatory options that minimize adverse economic impacts on small entities.

Guidance: EPA has issued specific guidance for complying with RFA/SBREFA requirements.[14]

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

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."[15]

Requirements contingent on threshold one: Section 202 of UMRA requires preparation of a written statement that includes the legal authority for the action; a benefit-cost analysis; a distributional analysis; estimates of macroeconomic impacts; and a description of the Agency's consultation with elected representatives of the affected State, local, or tribal governments. Section 205 of UMRA requires the 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 written interim guidance and OMB has provided general guidance on complying with UMRA.[16]

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

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 on or posed to ten or more persons other than federal agencies or employees.

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[14] U.S. Environmental Protection Agency, EPA Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act, November 2006. Available at http://intranet.epa.gov/adplibrary (accessed May 1, 2008, internal EPA document).

[15] Note that it is the threshold in this case that is "adjusted annually for inflation."

[16] See http://intranet.epa.gov/adplibrary (accessed April 14, 2004).

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 (i.e., fewer than 10 persons would be surveyed) in a specific situation, 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 can be found on EPA's intranet site, "ICR Center."[17]

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

3 Statement of Need for Policy Action

A clear statement of the need for policy action should be included in economic analyses of environmental policy prepared for economically significant rules.[18] This chapter discusses the key elements that should comprise this statement, which include:

- A definition of the environmental problem to be addressed (Section 3.2);
- An analysis of the reasons existing legal and other institutions have failed to correct the problem (Section 3.3); and
- A justification of the need for Federal intervention instead of other alternatives (Section 3.4).

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 this document should be applied selectively to be 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.

[18] Executive order 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 extends the requirements in 12866 to guidance documents.

OMB's *Circular A-4* discusses three categories of "market failure" including externalities, market power, and inadequate or asymmetric information.[19] *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 Executive Order 12866, the statement of need should also 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 in which other factors are responsible for the failure of the market or public and private sector institutions to adequately address environmental problems.

3.3 Need for Federal Action

The final component of the statement of need for the 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.[20] 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 executive order 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 the policy action should also 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.

[19] For further discussion of market failure, see Perman et al. (2003), Hanley et al. (1997), and Nicholson (1995).

[20] See Executive Order 13132 on "Federalism" for introductory statements regarding principles of federalism, and a section describing the special requirements for preemption.

4 Regulatory and Non-Regulatory Approaches to Pollution Control

Once Federal action is deemed necessary to address an environmental problem, policymakers have a number of options at their disposal to influence pollution levels. In deciding which approach to implement, policymakers must be cognizant of constraints and limitations of each approach in addressing specific environmental problems. 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 with other existing regulations.

This chapter briefly describes several regulatory and non-regulatory approaches used in environmental policymaking, but does not attempt to detail the relative merits of putting them into practice. Instead, the goal of this chapter is to introduce several important terms and concepts to analysts, describe the conceptual foundations of each approach, and provide additional references for those interested in a more in-depth discussion.[21] Specifically, this chapter discusses the following four general approaches to environmental policymaking: command-and-control regulation, market-based incentives, hybrid approaches, and voluntary initiatives. While command-and-control regulations are by far the most commonly used method of environmental regulation in the U.S., the three other approaches are increasingly employed by EPA. 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 often allow environmental improvements in areas not traditionally regulated by EPA.

The remainder of this chapter is organized as follows:

- Section 4.2 introduces the concept of efficiency;
- Section 4.3 discusses the traditional command-and-control approach to environmental regulation;
- Section 4.4 examines four market-oriented approaches to environmental regulation: marketable permits, emission taxes, subsidies, and tax-subsidy combinations;
- Section 4.5 examines three hybrid approaches: standards-and-pricing, information disclosure, and liability rules;
- Section 4.6 highlights potentially relevant factors in the selection of appropriate market-based or hybrid approaches; and
- Section 4.7 discusses non-regulatory or voluntary approaches to environmental regulation.

The chapter concludes with a discussion of criteria used to evaluate the effectiveness of regulatory and non-regulatory approaches to pollution control.

[21] Baumol and Oates (1988), particularly Chapters 10-14, Kahn (1998), and Sterner (2003) are useful general references on the economic foundations of many of the approaches presented in this chapter.

4.1 Choosing the Efficient Level of Pollution

While any policy option under consideration must balance efficiency with other important policy goals, economic efficiency is a useful concept to frame the discussion and comparison of the many regulatory options presented in the remaining sections of this chapter. [22]

Economic efficiency can be defined as the maximization of social welfare. In other words, an efficient policy is one that allows society to maximize the difference between social benefits and social costs, or maximize net benefits.[23] Conceptually, reductions in emissions should continue to occur 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).[24] 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), we achieve an efficient aggregate level of emissions in which the cost of abating one more unit of pollution is equal across all polluters.

Figure 4.1 illustrates that the economically efficient level of pollution is the one that sets the marginal benefits and marginal costs of abatement equal to each other. The damages from emissions are represented by the marginal external damage curve (MD) while the costs of controlling emissions are represented by the marginal abatement cost curve (MAC). External damages may include the costs of worsened human health, reduced visibility, lower property values, or loss of crop yields or biodiversity.[25] E_1 represents the amount of emissions if there were no regulation on firms. The total damages associated with E^* are represented by area AE_0E^* while the total abatement costs are represented by area AE_1E^*. The total burden on 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 at E^* is represented by the triangle AE_oE_1.

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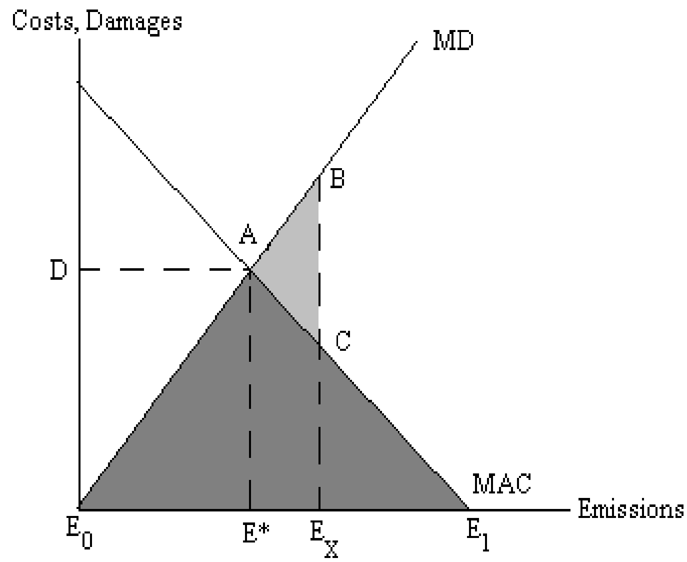
[22] The Appendix to this document includes a detailed discussion of market theory, including economic efficiency.

[23] Note that 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.

[24] 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.

[25] Note that valuing external damages (aka "lost benefits) is the subject of Chapter 7.

Figure 4.1 - The Efficient Level of Pollution



The policy that sets the emissions level at E^* - at the point where MD intersects MAC - is efficient. This is clear if we examine a level above or below E*. Suppose, for example, that emissions are at E_x -- a level greater than E^*.   In this case, total damages are equal to the area of the triangle BE_0E_x while total costs of abatement are equal to the area of the triangle CE_xE_1. The total burden to society is represented by the sum of the two triangles ($BE_0E_x + CE_xE_1$). Compared to our efficient solution, choosing emissions level E_x rather than E^* leads to excess social costs represented by the area of the triangle *ABC*. Similarly, if emission levels are below E^*, then total abatement costs are greater than total damages, again resulting in excess social costs.

It is important to remember that Figure 4.1 illustrates the simplest possible world where the pollutant is a uniformly mixed flow pollutant. In other words, 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. It is also important to note that, in practice, benefit-cost analysis is only one of a number of inputs into the decision-making process. Ultimately, the level of emissions chosen by the decision maker may not be the most economically efficient.

4.2 Traditional Command-and-Control Regulations

Before 1990, most environmental regulation in the U.S. were command-and-control regulations.[26][27] Despite the introduction of other, potentially more efficient methods for regulating emissions, this type of

[26] A number of notable exceptions do exist, however, including the Lead phase-down 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 US EPA (2001b).

regulation is still commonly used and is almost always available as a "backstop" if other approaches do not achieve desired pollution limits. A command-and-control regulation can be defined as a policy that mandates the control of emissions to achieve a specified level of environmental quality or a given emissions standard. These types of regulations are also referred to as direct or standards-based controls.

Two types of command-and-control regulations exist. The first, a *technology* or *design standard*, mandates the specific control technologies or production processes that polluters must use to meet the emissions standard. The second, a *performance-based standard*, also requires that polluters meet an emissions standard, but allows the polluters to choose any available method to meet that standard. Performance-based standards that are technology-based, however, do not specify a particular technology, but rather consider what available and affordable technology can achieve when establishing a limit on emissions.[28] At times, EPA may completely ban or phase out the use or production of a particular product or pollutant, as it has done with chlorofluorocarbons (CFCs) and certain pesticides. This is an example of the most stringent form of command-and-control regulation in which the standard specifies zero allowable emissions.

Note that standards need not be uniform but may vary according to size of the polluting entity, production processes or similar factors. Regulations are often tailored in this manner so that similar regulated entities are treated similarly.

In some sectors, regulators and firms may both prefer command-and-control regulations because they require firms to meet a known standard. When emissions are measurable and the appropriate monitoring technology is in place, determining whether a facility meets or exceeds a given rate of emissions is a straightforward task. In the case of technology standards, monitoring and enforcing the use of a particular technology is also relatively easy. Technology standards are especially useful in cases where the cost of monitoring emissions is high but the cost of monitoring the installation and use of abatement technology is not. Command-and-control regulation is also useful in cases where the level of pollution that maximizes social welfare is at or near zero, firms are not responsive to price signals, or random events or emergencies occur.[29]

Command-and-control regulations require that all polluters (or groups of polluters) meet the same standard. When abatement costs are similar across regulated polluters, this requirement is reasonably efficient. However, when abatement costs vary substantially across polluters, reallocating abatement activities such that some polluters have stricter standards than others would lead to substantial cost savings. If reallocation were possible, a polluter facing relatively high abatement costs would continue to emit at its current level but pay for the damages incurred (e.g., by paying a tax or purchasing permits),

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[27] Goulder and Parry (2008) refer to these as “direct regulatory instruments” because they feel that “command-and-control” has a “somewhat negative connotation.” While this may be true, the bulk of the literature uses the term command and control and we follow that convention in order to minimize confusion.

[28] 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)).

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

while a polluter with relatively low abatement costs would reduce its emissions. A command-and-control regulation does not allow for such reallocation to take place. However, in some cases, older polluters are "grandfathered" from a new regulation and subject to a different, less stringent standard than newer polluters. Grandfathering creates a bias against constructing new facilities and investing in new pollution control technology or production processes.[30] So, grandfathered older facilities with higher emission levels tend to remain active longer than they would if the emissions standard applied equally to all polluters.

Technology standards further constrain firm behavior by mandating how firms must meet the standard irrespective of cost-effectiveness. While pollution is reduced to the desired level, it is accomplished at a higher cost than if firms were to determine on their own the most cost-effective means for meeting the standard given their unique company attributes.

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 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.

In Figure 4.1, a command-and-control regulation could be illustrated by a marginal abatement cost curve that is higher and to the right of the one shown. At any level of emissions, the marginal cost of abatement would be higher, including at E^*. In this case, it is conceivable that the shape of the estimated curve may also differ substantially from the actual abatement cost curve because of limited information available to policymakers on the best method of abatement or most appropriate emissions rate or level. Policymakers may instead limit abatement costs associated with a particular emissions policy to D, but then mandate emission levels at a point higher than E^*.

It is also important to note that a single command-and-control regulation does not directly control the aggregate emission level. For technology standards and performance-based standards that specify an emission rate, aggregate emissions will depend on the number of polluters and the output of each polluter. In other words, as either production or market size increase, so will aggregate emissions. When a performance-based standard is defined in terms of an emission level per polluting firm, aggregate emissions will still be a function of the total number of polluting firms.

Importantly, command-and-control regulations diminish incentives for innovation. Because technology-based 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. 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.[31] In other words, 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. However, in practice, the implementation of performance-based standards based

[30] For a discussion of grandfathering, see Helfand (1991).

[31] 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).

on emission rates often inhibits innovation. Because permitting authority is often delegated to the States, approval of a technology in one state does not ensure its approval in another. Further, regulators may review the existing technologies available and determine that a particular technology meets the criteria set by the standards, essentially requiring it for permit approval. Firm investment in research to develop new, less expensive, and potentially superior technologies is therefore discouraged.[32]

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. For example, when smoking is outlawed within an office building, the non-smoker clearly has the property rights. In a coffeehouse or nightclub that allows smoking, it is instead the smoker that has the property rights. In the first case, the smoker may bribe the non-smoker to allow him to smoke illegally in the office. The bribes need not be in the form of cash but could be payments in kind. In the second case, the non-smoker would have to bribe the smoker to stop smoking.

In each case, the effectiveness of the agreement is contingent on meeting several additional conditions: all parties must have full information, bargaining must be possible, and transaction costs must be low. These conditions are typically met when there are only a small number of individuals involved. If either party is unwilling to negotiate, is not privy to the same information, or faces high transaction costs, then no private agreement will be reached. Going back to our example, if a non-smoker walks into a crowded nightclub in which everyone is smoking, clearly the non-smoker’s ability to negotiate with every smoker is more difficult than negotiating with one smoker.

4.3 Market-Oriented Approaches

Market-oriented 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.[33] Market-oriented approaches may differ from more traditional regulatory methods in terms of economic efficiency 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 allow firms more flexibility than more traditional regulations and capitalize on the heterogeneity of abatement costs across polluters to reduce aggregate pollution efficiently. Market-based approaches discussed in this section include:

- Marketable permit systems;
- Emission taxes;

[32] See Swift (2000) and US EPA (1991) for a detailed discussion of how emission rate-based standards hinder technological innovation.

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

- Subsidies; and
- Tax-subsidy combinations. [34]

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 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.[35]

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 allowances is allowed to vary. In terms of Figure 4.1, the government issues a set number of allowances - one per unit of emissions - equal to E^*. 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 that 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 differ from command-and-control regulations in that they aim to limit aggregate emissions over a compliance period rather than establish an emissions rate.

The equilibrium price of allowances, in theory, adjusts so that it equals the marginal external damages from a unit of pollution, which 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. When only one permit price exists in the market, trading will occur until marginal abatement costs equalize across all firms. [36]

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[34] 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, 2000) 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 policymakers to determine when market-based incentives may result in cost savings over command-and-control regulations. Harrington, Morgenstern, and Sterner (2004) compare the costs and outcomes of command-and-control and incentives-based regulatory approaches to the same environmental problem in the U.S. and Europe. Additional sources include Sterner (2003), Stavins (2003), Tietenberg (1999, 2002), U.S. EPA (2004a, 2001b), OECD (1994a, 1994b), and proceedings published under the "Project 88" forum, Stavins (1988, 1991).

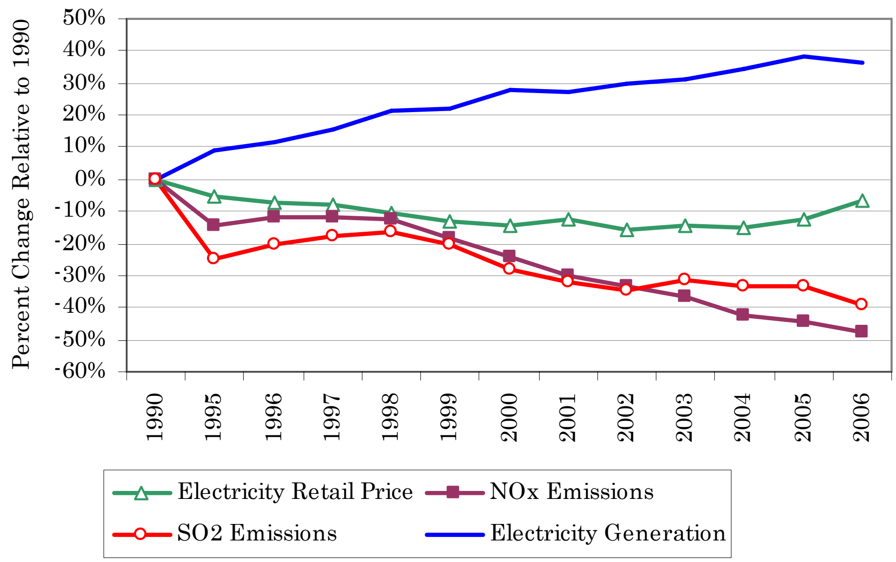
[35] For a more detailed discussion of the various systems and how to design them, see US EPA (2003c)

[36] 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 the EPA's Acid Rain Website at http://www.epa.gov/acidrain (accessed 04/05/2004).

Text Box 4.2- Acid Rain Trading Program for Sulfur Dioxide

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 sulfur dioxide 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 sulfur dioxide generating units into the system. Additional emissions restrictions will occur beginning in 2010.

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 Acid Rain Program’s benefits (including SO_2 trading and direct NO_x controls) reaching upwards of $120 billion annually in 2010 with annual costs around $3 billion (in 2000$) for 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 SO2 and NOx emissions have fallen substantially, electricity retail prices until very recently have declined in real terms.



Source: USEPA 2007a.

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

Generally, allowances initially sold at auction represent income transfers from the purchasers to the government in the amount of the price bid for the allowances. Alternatively, allowances allocated to polluters according to a specified rule represent 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), who receives them and 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 policymaker should also consider potential barrier-to-entry effects. Overall, the treatment of 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 U.S. experience suggests that a market characterized as having 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 may be significantly hindered by burdensome requirements that increase the transaction costs associated with each trade.[37]

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 difficult. Other possible solutions include limiting trading to particular zones that exclude more sensitive areas and establishing pollution “floors.”

However, two recent reviews of the literature (Burtraw et al. 2005 and Harrington et al. 2004) have found 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 in some cases. These results come primarily from studies of the SO_2 and NO_x trading programs and the results may not transfer to other pollutants which have more localized effects if the market-based policy is not carefully designed.

Banking introduces increased flexibility into a trading system by allowing polluters to bank unused permits for future use. For example, 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 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.[38]

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[37] 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 U.S. and the main reasons for its failure.

[38] Notably the U.S. Acid Rain Trading Program has nearly 100 percent compliance and requires a limited number of staff to administer. Specifically, the program only requires about 50 EPA staff.

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, which is then used to accommodate the emissions from the latter. 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 have been used mostly to control air pollution in non-attainment areas, they have historically been 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 (US EPA 2001b).

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 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 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 control mobile source emissions. (US EPA 2003c)

4.3.2 Emission Taxes

Emission taxes are exacted per unit of pollution emitted and induce a polluter to take into account the external cost of its emissions. Under an emission 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. This efficient outcome is illustrated in Figure 4.1 by setting the tax equal to $\$D$/unit of emissions, which results in the achievement of the level of pollution E^* and maximizes social welfare.

As an example of how an emission 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, which may include 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 individually decide 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, reduce other taxes, or redistribute to finance other public services.[39] While difficult to implement in cases where there is temporal and/or spatial variation in

[39] For more information on how the government may use revenues from taxes to offset distortions created by other taxes, see Goulder (1995) and Goulder, Parry, and Burtraw (1997).

emissions, policymakers may 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 U.S.[40] This is because taxing emissions directly may not be feasible when emissions are difficult to measure or accurately estimate, 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.[41] 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 the distribution of economic rents that result from these programs.

User or product charges are a variation on emission taxes that are occasionally utilized in the U.S. These charges may be imposed directly on users of publicly operated facilities or on intermediate or final products whose use or disposal harms the environment, and they 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 usually are 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.

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

4.3.3 Subsidies

Subsidies paid by the government to firms or consumers for their per unit reduction in pollution create the same abatement incentives as emission taxes or charges. For example, 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.

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.[42] 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

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

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

[42] 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 really is 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.

because it requires government outlay, and analysts should consider the opportunity costs associated with using public funds. However, there may be cases in which a subsidy is more feasible than an emissions tax especially when it is difficult to identify polluters, or when research and development activities relevant to emissions abatement would otherwise be under-funded.

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 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. [43]

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. However, 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 emission 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 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.[44] Buy-back programs also exist to promote the scrapping of old, high-emission vehicles.

Environmental subsidies in the U.S. 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

Emissions taxes and environmental subsidies can also be combined to achieve the same level of abatement as when used separately. These instruments are commonly referred to as **deposit-refund systems** in which the deposit operates as a tax and the refund serves as an 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.

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[43] See Sterner (2003) for a more in-depth discussion of how subsidies work and for numerous examples of subsidy programs in the U.S. and other countries.

[44] For more information on the Office of Air's Small Engine Buy-back Program see US EPA (2006c).

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 function 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.[45]

Many examples of deposit-refund systems exist, most of which are designed to encourage consumers to reduce litter and increase the recycling of certain components of municipal solid waste.[46] 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). Deposit-refund systems also have been discussed in the literature as a means of controlling non-point source water pollution, cadmium, mercury, and the removal of carbon from the atmosphere.[47]

The main advantage of a deposit-refund system 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. Because firms have access to better information than government, they may measure and report emissions with greater precision and at a potentially lower cost.

Disadvantages of the deposit-refund 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). Policymakers 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. sulfur dioxide from powerplants) 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.[48] 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.

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

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

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

[48] 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).

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 that 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.[49]

4.4 Hybrid Approaches

In addition to the market-based instruments discussed above, hybrid approaches – those that combine aspects of command-and-control and market-based incentive policies -- are often discussed in the literature and increasingly used in practice. However, hybrid approaches are not always the most economically efficient approach because either the level of abatement or the cost of the policy is greater than what would be achieved through the use of a market-based incentive approach. Nevertheless, such approaches are appealing to policymakers 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 hybrid 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, however, 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 that combines standards with pricing mechanisms.[50] In the case of a standard and tax combination, the same emissions standard is imposed on all polluters and all polluters are then 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 also has several other attractive features. First, if the standard is set properly, proper protection of health and the environment will be assured. This feature of the policy maintains the great advantage of standards: 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.

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

[50] See Roberts and Spence (1976) and Spence and Weitzman (1978).

4.4.2 Information Disclosure

Requiring disclosure of environmental information has increasingly been used as a method of environmental regulation. 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 the government or the public. By collecting and making such information publicly available, firms, government agencies, and consumers may be 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. In other cases, a community with information on a nearby firm's pollution activities may exert pressure on the firm to reduce emissions, even if formal regulations or monitoring and enforcement are weak or nonexistent.[51]

Requirements for information disclosure need not be tied explicitly to an emissions standard; however, they 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. In addition, as is the case with market-based instruments, polluters still have the flexibility to reduce emissions in response to community pressure in the cheapest way possible.

The use of information disclosure or labeling rules has several other advantages. First, 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 may 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.

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 to monitor and report releases.[52] Any required investments in pollution control are in addition to this amount. Second, the amount of pressure a community exerts on a plant may be related to socioeconomic status. Poorer, less educated populations tend to exert far less pressure than communities with richer, well-educated populations.[53] 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 and not on possible ecological or aggregate societal effects, or do not understand how to properly interpret information released in terms of the health risks associated with exposure to particular pollutants.

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[51] For more information on how information disclosure may help to resolve market failures, see Naysnerski and Tietenberg (1992), Pargal and Wheeler (1996), and Tietenberg and Wheeler (2001).

[52] See O'Connor (1996) for information on the costs of monitoring and reporting environmental information. In addition, see World Bank (2000) for a discussion of the main advantages and disadvantages of information disclosure as a policy tool.

[53] See Hamilton (1993) and Arora and Cason (1999).

EPA-led information disclosure efforts include the Toxics Release Inventory (TRI); consumer-based programs on the risks of particular toxic substances, the level of contamination in drinking water; the pesticide-labeling program and the AIRNow program which reports air quality index forecasts for over 300 cities. The TRI requires firms to provide the government and public with information on pollution and abatement activities at each plant on an annual basis if emissions of a set of certain toxic chemicals exceed a threshold. There is some evidence in the literature that the most polluting firms experience small declines in stock prices on the day TRI emissions are released to the public (Hamilton (1995) finds a stock price return of -0.03 percent due to TRI release). Firms that experienced the largest drop in their stock prices also reacted by reducing their reported emissions most in subsequent years.[54]

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: 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 to compensate victims when careful planning does not occur. These rules are used to guide courts in their compensation decisions when the court rules in favor of the victim. Liability rules may serve as an incentive to polluters - to the extent that polluters are aware that they will be held liable before the polluting event occurs - to minimize or prevent involvement in activities that inflict damages on others. However, liability rules differ from other regulatory instruments in two basic ways. First, more uncertainty exists as to the magnitude of payment. Second, liability rules can generate relatively large costs, both in terms of assessing the environmental damage caused, and the amount due.[55] As a result, liability rules are most useful as a regulatory instrument for those cases in which damages requiring compensation are expected to be infrequent, and monitoring firm compliance with regulatory procedures is difficult.

Strict liability and negligence are two types of liability rules relevant to polluters. Under strict liability, polluters are held responsible for all damages whereas under negligence polluters are liable only if they do not exhibit “due standard of care.” Regulations that impose strict liability on polluters for the health and environmental damage caused by their pollution 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 CERCLA (or Superfund) and the Corrective Action provisions of RCRA. These rules have also been used in the redevelopment of potentially contaminated industrial sites, known as brownfields.

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[54] Hamilton (1995) and Konar and Cohen (1997) are two examples of empirical studies that have investigated how the TRI has affected firm behavior and stock market valuation.

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

4.5 Selecting the Appropriate Market-Based Incentive or Hybrid Approach

The selection of the most appropriate market-based incentive or hybrid regulatory approach depends on a wide variety of factors, including: [56]

- The type of market failure being addressed;
- The specific nature of the environmental problem;
- The degree of uncertainty surrounding costs and benefits;
- Concerns regarding market competitiveness;
- Monitoring and enforcement issues;
- Potential for 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 is the failure of firms or consumers 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 is the inability of firms or consumers to make optimal decisions due to lack of information on investment options, available abatement technologies, 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 because 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 non-point 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 sources?[57] Point sources, which emit at identifiable and specific locations, are much easier to identify and control than diffuse and often numerous non-point sources, and therefore are often amenable to the use of a wide variety of market instruments. Although non-point sources are not regulated under EPA, the pollution emitted from a non-point source is. This makes the monitoring and control of non-point source emissions a challenge. In instances where both point and non-point sources contribute to a pollution problem, a good case can be made for a tax-subsidy combination or a tradable permits system. Under such a system, emissions from point sources might be taxed while non-point source controls are subsidized.

[56] Helpful references that discuss aspects to consider when comparing among different approaches include Hahn (1990), Hahn and Stavins (1992), OECD (1994a, 1994b), and Sterner (2003).

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

Flow pollutants tend to dissipate quickly, while stock pollutants persist in the environment and tend to accumulate over time. While it is possible to rely on a wide variety of market and hybrid instruments for the control of flow pollutants, stock pollutants may require strict limits to prevent bioaccumulation or detrimental health effects at small doses, making direct regulation potentially more appealing. If the limit is not close to zero, then 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 potentially practical options. These same instruments are appealing when pollutants are not uniformly mixed across space. In this case, it is important to account for differences in baseline pollution levels, and 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 may 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.

4.5.3 The Degree of Uncertainty

The choice between price-based instruments (e.g. taxes or charges) and quantity-based instruments (e.g. marketable permits) also has been shown theoretically to rest on the degree of uncertainty surrounding the 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 costs of abatement exists and policymakers wish to guard against potential high costs borne by polluters as a result of regulation, then they can limit these costs by using a price instrument. If, on the other hand, more uncertainty associated with the benefits of controlling pollution exist and policymakers wish to guard against high environmental damages, they can limit these damages by using a quantity instrument.[58] The policymaker also should be aware of any discontinuities or threshold values above which sudden large changes in damages or costs could occur due to a small increase in the level of abatement required.

4.5.4 Market Competitiveness

Market power is a type of market failure in and of itself, resulting 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 in which firms have some amount of market power. A combination of market-based instruments may work more effectively than a single instrument in this instance. In addition, to the extent that cost burdens are differentiated, the use of certain market-based instruments may cause a change in market structure to favor existing firms, creating barriers of entry and allowing these firms a certain degree of control over price. Permit systems that set aside a certain number of permits for new firms, for instance, may guard against such barriers.

4.5.5 Monitoring and Enforcement Issues

Market-oriented instruments differ in the degree of difficulty required to monitor and enforce them. For example, subsidies, deposit-refund systems, and information disclosure shift the burden of proof to

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

demonstrate compliance from government to regulated entities. Because firms generally are in a better position than government to monitor and report their own emissions, they may do so at a potentially lower cost. This feature makes these approaches attractive when monitoring is difficult or emissions must be estimated (e.g. when there are non-point 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.

4.5.6 Potential Economy-Wide Distortions

Analysts should also consider the potential distortionary effects of market-based instruments. Instruments that include a revenue-raising component, such as auctioned permits or taxes, may allow for opportunities to direct collected resources to the reduction of market inefficiencies.[59] See Chapter 8 and the Appendix A for a more detailed discussion of economy-wide distortions.

4.5.7 The Goals of the Policymaker

Finally, the goals of policymakers may influence the instrument selected to regulate pollution. Each instrument considered may have different distributional and equity implications for both costs and benefits that should be accounted for when deciding among instruments. For example, policymakers may wish to ensure clean-up of future pollution by firms. In this case, insurance and financial assurance mechanisms may be useful instruments to supplement existing standards and rules when there is a significant risk that sources of future pollution might be incapable of financing the required pollution control or damage mitigation method. In addition, the level at which policymakers allow the market to determine exact outcomes may influence the instrument chosen. Marketable permits, for example, set the total level of pollution control, but the market determines which polluters reduce emissions. On the other hand, taxes let the market determine the extent of control by individual polluters and the total level of control.

4.6 Non-Regulatory Approaches

Analysts are encouraged to consider non-regulatory approaches as potential alternatives to regulation. EPA has pursued a number of non-regulatory approaches that rely on **voluntary initiatives** to achieve improvements in emissions controls and 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. Others have been developed to improve environmental quality in areas that policymakers expect may be regulated in the future but are currently not regulated, such as greenhouse gas emissions and non-point source water pollution.[60]

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[59] 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); Jorgenson (1998a, 1998b).

[60] 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.

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.[61] 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 almost 62 programs as of 2008, involving more than 11,000 organizations. Partner organizations include small and large businesses, citizen groups, state and local governments, universities, and trade associations.[62] The voluntary programs in which these groups participate tend to either have broad environmental objectives and target a variety of firms from different industries, or focus on more specific environmental problems that are often relevant to a single industrial sector. In the U.S., nearly one third of all multi-sector federal voluntary programs focus on energy efficiency and climate change issues, with general pollution prevention efforts being the next most popular issue. 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 attempt to provide targeted and effective technological expertise and assistance to participating firms.

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.

Goal setting is the most common method used in the design of voluntary programs. Implementation-based goals require participants to meet specific targets set by EPA and, provide consistency in objectives across firms. These goals make it easier to design approaches that go beyond standards specified in formal regulation. Additionally, they make it simpler to monitor and measure whether participants are meeting the goal. Target-based goals, where the EPA specifies a qualitative process-oriented goal and firms individually set and then meet a specific target, allow firms increased flexibility. Less information

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[61] Because these programs are voluntary there is no need for formal public comment, however, industry often is consulted during the design phase.

[62] 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 1/30/2008). Additional information may be found at http://www.epa.gov/innovation (Accessed 1/30/2008) and http://www.epa.gov/p2/ (Accessed 1/30/2008).

regarding the individual firm's ability to respond to a particular goal is needed with this approach, as the firm selects a goal it feels is appropriate given its particular cost structure. The EPA's 33/50 program, which set a goal of a 33% reduction of toxic emissions by firms in the chemical industry by 1992 and a 50% reduction by 1995 (relative to a 1988 baseline), is an example of a voluntary program with an implementation-based goal. The EPA's WasteWise and Climate Challenge programs are examples of programs with target-based goals.

Promoting environmental awareness and encouraging process change within firms requires the Agency to evaluate firm's ongoing operations using detailed information and for firms to be willing to follow the government's recommendations for technology investments. Examples of this type of approach are the Green Lights program, which encouraged firms to adopt energy efficient changes that also produced savings in electricity costs, and the Design for the Environment and Green Chemistry programs, which research and encourage firms to use environmentally-friendly processes.

Publicly recognizing firm participation in a program provides 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 their competitions' products.[63] This is the intent of the Green Power Partnership, Climate Leaders, and It All Adds Up to Cleaner Air programs.

Finally, product labeling is used on either the intermediate inputs in a production process or on the final good. Labels on intermediate goods encourage firms to purchase environmentally-responsible inputs. Labels on final goods allow consumers to differentiate between goods produced using a relatively clean production process and those that do not. For example, products deemed energy efficient may apply for the use of the Energy Star label.

[63] See Arora and Cason (1995), Arora and Gangopadhyay (1995), Konar and Cohen (1997, 2001), and Videras and Alberini (2000) for more information on the main arguments for why firms participate in voluntary programs.

Text Box 4.1 - Water Quality Trading of Non-Point Sources

In 2003, EPA issued a "Water Quality Trading Policy " (US 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, the EPA is required to establish Total Maximum Daily Loadings (TMDL) of pollutants for impaired water bodies. The TMDL provides a method for allocating pollutant discharges among point and non-point sources. Point sources are regulated by the EPA and, as such, required to hold National Pollutant Discharge Elimination System (NPDES) permits that limit their discharges. However, many water bodies are still threatened by pollution from unregulated, non-point sources. Nutrients and sediment from urban and agricultural runoff have led to water quality problems that limit recreational uses of rivers, lakes, and streams, create hypoxia in the Gulf of Mexico, and decrease fish populations in the Chesapeake Bay. Allowing for effluent trading between point and non-point sources would lower nutrient and sediment loadings and improve or preserve water quality.

To ensure that the reduction resulting from the trade has the same effect on the water quality than the reduction that would be required without the trade, trade ratios are applied. These ratios attempt to control for the differential effects resulting from the following factors:

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

Through trading, point sources can still meet the discharge limit but at a lower cost that allows continued growth and expansion of production, while non-point sources have an incentive to reduce pollution through participation in the market. To the extent that it is cheaper for a non-point source to reduce pollution than to forgo revenues earned from the sale of any unused credits to point sources, the non-point source will choose to emit less pollution.

As of March 2007, 98 NPDES permits, covering 363 dischargers, included provisions for trading. Only about a third of the dischargers, however, had carried out one or more trades under these permits (US EPA 2007f). The program currently has 37 programs in place (or under development) in 26 states; but to date there have been very few, if any, trades. Trading has been limited for several reasons. While Best Management Practices (BMPs) are typically used to define a pollution reduction credit from a non-point 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 non-point source difficult. Another challenge is encouraging non-point source involvement in trading, given the agriculture industry's distrust of regulators. Further, it is difficult to define appropriate trading ratios between point and non-point sources. Finally, it can be difficult to communicate the advantages of trading to stakeholder groups.

EPA's Water Quality Trading website includes a number of tools to assist stakeholders with establishing trading programs including: the "Water Quality Trading Assessment Handbook" (US EPA 2004b) and the "Water Quality Trading Toolkit for Permit Writers" (US EPA 2007h). These resources, together with more information about water quality trading projects, can be found at http://www.epa.gov/owow/watershed/trading.htm.

4.6.2 Economic Evaluation of Voluntary Approaches

A formal economic analysis is not required for the selection and implementation of a non-regulatory approach. However, given the increased reliance of policymakers on these approaches, EPA has begun to focus on the issue of accountability, or how to identify the voluntary initiatives that function well and those that do not. The Agency formed the EPA Partnership Program Review Workgroup in November 2006 and tasked the workgroup with developing an analytic framework that, when implemented, would improve the effectiveness and efficiency of partnership programs. The framework, completed in 2007, provides direction for the analysis of strategic goals and inter-program analysis to determine common interests across programs and potential for efficiency gains, as well as a systematic method for assigning partnership programs to one of three assessment categories with the largest programs receiving the most scrutiny. In addition, the workgroup developed a set of minimum recommended standards for EPA partnership program management to inform reviews of these programs (US EPA 2007e).

Several factors contribute to difficulties in evaluating voluntary approaches. Many programs target general environmental objectives and thus lack a measurable environmental output. Alternatively, a measurable output may exist, but there may be a lack of data on the firm's environmental outputs. Additionally, a reasonable baseline from which to make a comparison must be established, which will require an extensive analysis comparing the actions of participants versus non-participants in the program.[64] 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 obstacles may be overcome if voluntary approaches use more defined and detailed goal setting and require more complete data collection and reporting from the outset.[65]

4.7 Measuring the Effectiveness of Regulatory or Non-regulatory Approaches

There are several policy criteria that 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, the answers to these questions 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; Goulder and Parry 20008), 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.

Several key questions regarding each policy objective must be considered in determining the overall effectiveness of the approach:

- **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 program?

[64] See Chapter 5 for a discussion of Baselines and specifically section 5.7 for a discussion of behavioral responses.

[65] See also Khanna (2001), OECD (1999, 2003), U.S. EPA (2001), and Brouhle, Griffiths, and Wolverton (2004) for discussions of how voluntary programs work and how they are used in the U.S. for environmental policymaking.

- **Economic Efficiency.** Does the policy instrument reach a given environmental goal at the lowest possible cost to firms and consumers? How close does the approach come to the most efficient outcome?
- **Reductions in Administrative, Monitoring, and Enforcement Costs.** Does the government benefit from reductions in costs? How large are these cost savings compared to 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 may 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 abatement techniques that make the cost of compliance with environmental regulations decrease over time?

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

5 Establishing a 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 potential benefits and costs of a proposed regulation. Because the 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 analysis 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 of 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 (e.g., judicial and statutory deadlines, 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.[66] 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, however, 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 by firms and the public to the proposed rule.

On occasion a regulatory program may be set to expire or dramatically change, however, 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, 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 (i.e., the baseline scenario) to the expected state of the world with the proposed policy or regulation in effect (i.e., the policy scenario). Economic and other impacts of policies or regulations are measured as the differences between these two scenarios.

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, however, a few cases in which more than one baseline may be necessary.

[66] A policy action includes both regulations and the issuance of Best Management Practices or guidance documents that do not carry the same force as a regulation, but do affect the decisions of firms and consumers.

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. If, for instance, 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 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. Finally, if the uncertainty surrounding the current level of pollution is so large that a probabilistic analysis becomes difficult, multiple baselines 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. Use of probabilistic tools (e.g., Monte Carlo analyses) may be one way to avoid the need for multiple baselines. 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

To assist analysts in baseline specification, several guiding principles are listed and discussed below. Though they exhibit a common-sense approach to the issue, 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.

Guiding Principles for a Baseline Analysis

1. Clearly specify the current and future state of the economy and 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.

Clearly specify the current and future state of the economy, the environmental problem that the regulation addresses and the regulatory approach being considered. A clear written statement about the current state of the economy and environment will help decision-makers and the general public 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 parties affected.

There should also be a discussion of why a particular regulatory approach was chosen (e.g., best available technology (BAT), performance measures, market incentives, or non-regulatory approaches). Sometimes, the regulatory approach will affect the choice of the baseline. For instance, baselines for rules implementing a BAT may be easier to specify than those for non-regulatory approaches.

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 a novel regulatory approach is being proposed, then a baseline of the alternate regulatory policy might be used as a comparison for the novel approach. 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.

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 may 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 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 is first published in a Notice of Proposed Rule Making, 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 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, then, 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 emissions. In the baseline scenario, the level of greenhouse gases in the atmosphere would steadily increase over time, with a corresponding increase in expected impacts on human health and welfare and ecological outcomes. A benefit-cost analysis limited to the first decade after initiation of the policy 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 also 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.

Alternatively, 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-effective analysis 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.[67]

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 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 also 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

[67] 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.

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 their levels and 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, 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. Therefore, 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 rapidly moving overseas, this 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.[68]

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 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, however, is quite difficult and often controversial. Technological change can be thought of has having at least two components: true technological innovation (such as a new pollution control method) or 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 one reason why regulatory costs are overstated (Harrington, Morgenstern, and Nelson 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 Chlorofluorocarbons 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 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 – a sometimes difficult distinction to make.

Additionally, if technological innovation is included in the policy scenario, then it should be included in the baseline as well. While accepting that innovation will occur in the baseline and policy scenarios, some have argued that the rates across the scenarios may differ because regulation may cause firms to innovate more to reduce their cost of compliance. This is often labeled the “Porter Hypothesis” (Porter and van der Linde 1995; Heyes and Liston-Heyes 1999) which, in its strongest form predicts cost savings

[68] 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 compliment), 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.)

from environmental regulation. 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). As such, analysts should avoid assuming differing rates of technological innovation based on regulatory stringency.

In some cases, however, it may be important to inform decision-makers about the potential impact that technological innovation could have on regulatory costs and benefits. In cases where small changes in technology could 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, at a minimum, are 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.[69] 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, because 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 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.[70] 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 as well as 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, care should also be taken to explain the compliance assumption for the current regulation under consideration. The agency is unlikely to propose a rule that it believes will

[69] See U.S. EPA 1997b and U.S. EPA 2007b

[70] An exception would be if the proposed regulation were designed to correct the under-compliance from the previous rule. This is discussed in the section below on under-compliance.

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, **analysts should develop baseline and policy scenarios that assume full compliance with existing and newly enacted regulations for analyses of 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 may pose some challenges to the analyst, however, when current observed or reported economic behavior indicate otherwise. 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 the EPA.

As a practical matter, before totally rejecting assumptions of "full compliance" for existing and new 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 “current practice” baseline. For a deregulatory rule (e.g., a rule designed to address potential changes in or clarify definitions of regulatory performance that frees entities from enforceable requirements contained in an existing rule), for example, 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 may 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 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 (US EPA 1985).

If under-compliance is assumed in the baseline, then the nature of that non-compliance becomes important. 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, 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, as long as the compliance rate occurs uniformly. 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 with those regulatory options. For example, the expected compliance rate may differ depending on whether entities are regulated using economic incentives or prescribed control technologies.

While a baseline assuming under-compliance may be useful in some cases, it should be executed carefully. 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 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 have been 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., OSHA, DOT, 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.

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 rules together that affect the same industrial category. This was true of the pulp and paper effluent guidelines and NESHAP rules (US 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. ELG and NPDES work together to achieve one objective so 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 may 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 benefit-cost analysis. 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 (NOx), 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. 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 may 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 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, the EPA's Office of Water has calculated the benefits associated with improving river miles for various designated uses (e.g., swimming, fishing, 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 progress to a threshold.[71] Note, however, 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.

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

[71] It should be recognized that 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 rule making 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.

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 scenario 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, however, 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 not to 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 achieve compliance with other rules that are implemented or proposed. In this latter case, the 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 may 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 provided for 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 firm’s 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, et al. 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 will meet the new standard without over complying. This is because 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 firm’s 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 benefit-cost analysis and option selection. However, developing a baseline is not a straightforward process, and many decisions must be made 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.

6 Discounting Future Benefits and Costs

Discounting renders costs and benefits 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 nonmarket goods and services) that will be caused by a policy by a discount factor. At a summary level, discounting reflects the fact that people prefer consumption today over consumption in the future, and the fact 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.

At a more technical level, as detailed later in this chapter, 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.

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. *Private discounting*, on the other hand, is discounting from the specific, limited perspective of private individuals or firms. Implementing this distinction in practice can be complex, as detailed in this chapter, but it is an important distinction to maintain because using a given private discount rate instead of a social discount rate may bias results as part of a benefit-cost analysis.

This chapter addresses discounting over the relatively short term, what has become known as "*intra-generational discounting*" as well as discounting over much longer time horizons, or *inter-generational* discounting. Intra-generational, or *conventional*, discounting applies to those contexts that may well 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. Inter-generational 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 because 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 leading to different recommendations across these two scenarios.

The chapter begins with a description of the mechanics of discounting, followed by overviews of the background and rationale for discounting, and key considerations for discounting in the inter-generational context. The chapter concludes with recommendations and guidance for discounting in EPA benefit-cost analyses.[72]

[72] This chapter is intended to 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 on the topic. 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).

6.1 The Mechanics of Summarizing Present and Future Costs and Benefits

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

The net present value (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:

$$NPV = NB_0 + d_1 NB_1 + d_2 NB_2 + \dots + d_n NB_n \qquad (1)$$

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:

$$d_t = 1 / (1+r)^t \qquad (2)$$

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. It is important that the same discount rate be used for both benefits and costs because 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. The analyst can estimate the present value of costs and benefits separately and then compare them to arrive at net present value.

When estimating the NPV, it is 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 may 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.[73] 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 (i through iv) 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 almost always extremely small in the scope of an entire economic analysis.

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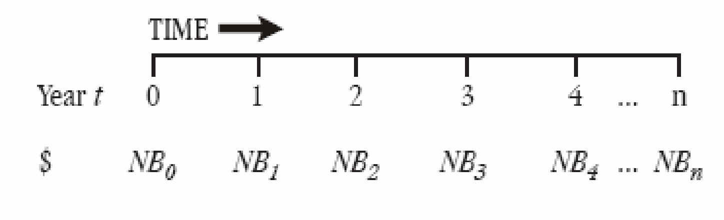
[73] See EPA (1995) for an example in which operating and monitoring costs are assumed to be spread out evenly throughout each year of compliance. While the exponential function above is the most accurate way of modeling the relationship between the present value and a continuous stream of benefits and 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 6.2



Time →

Year t 0 1

$ NB_{0i} NB_{0ii} NB_{0iii} NB_{0iv}

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.[74]

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 in equation 1, above);
r = the discount rate per period; and
n = the duration of the policy.

[74] 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* (US EPA, 2002). [6th Edition, EPA/452/B-02-001, January 2002, OAQPS].

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

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

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

Annualization 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.

6.1.3 Net Future Value

Instead of discounting all future values to the present, it is possible to estimate their 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:

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

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

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

where r is the discount rate.

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. Net present value represents the present value of all costs and benefits, annualization represents the value as spread smoothly through time, and net future value 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 they 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 net present value 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 net present value is positive does not depend on the discount rate, although the discount rate may affect the relative present value if a policy is compared to another policy).

Discounting can, however, substantially affect the net present value 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 net present value.

Text Box 6.1 - Potential Impact of Discounting

Suppose the cost of some environmental policy that is incurred entirely in the present is $1 billion, and that after 30 years a benefit results that is estimated to be worth $5 billion in the future. Without discounting, a policy that offers benefits five times its cost appears to be a very worthwhile social investment. Discounting the $5 billion future benefits, however, can radically alter the economic assessment of the net present value of the policy. $5 billion 30 years in the future discounted at 1% is $3.71 billion, at 3% it is worth $2.06 billion, at 7% it is worth $657 million, and at 10% 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 net present value. For this reason, in this type of scenario, the choice of the discount rate determines whether this policy is considered on economic efficiency grounds to offer society positive or negative net benefits.

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, 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 future values of risk 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 net present value 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 benefit-cost analysis.

For environmental health risks there are at least two kinds of time lags between exposures and effects:

- Latency is the time difference between initial exposure to a contaminant and an increase in health risks, while
- Cessation lag is time difference between *reduction* in exposure and a *reduction* in observed health effects.

Thus, one can consider latency as applying to a newly exposed population and cessation lag applying to a population with prior exposure. These timeframes need not be identical, as noted by EPA's Science Advisory Board:

> A good example is cigarette smoking: the latency between initiation of exposure and an increase in lung cancer risk is approximately 20 years. However, after cessation of exposure, risk of lung cancer begins to decline rather quickly. A benefits analysis of smoking cessation programs based on the observed latency would greatly underestimate the actual benefits.[75]

Both latency and cessation lag should both be accounted for in an economic analysis. Assuming a benefit-transfer approach this can be done by valuing the changes in risk at the time of health effect impacts and not when exposure changes. These values should then be discounted along with other effects in the analysis. Ignoring cessation lag by assuming that benefits accrue immediately upon reduced exposure will produce an upper bound estimate of benefits.

EPA has received recommendations from Science Advisory Board consultations on how to estimate cessation lags, and this information would ideally come from the risk assessment process using consistent models. For carcinogens, the mechanism by which cancer occurs can be informative.[76] However, cessation lag data is often very limited so EPA has been encouraged to pursue other models to examine the influence of the lag. EPA has estimated the benefits of reduced carcinogens in drinking water by modeling cessation lag with available data on contaminant-outcome combinations other than those

[75] *Arsenic Rule Benefits Analysis: A Review* (US EPA 2001c).

[76] See *Arsenic Rule Benefits Analysis: A Review* (US EPA 2001c).

targeted in the regulation, such as smoking and lung cancer.[77] This provides information on alternative cessation lags, but the applicability of these data for any particular contaminant is unknown. Additionally, based on expert recommendations, lags have been implemented in estimating the benefits of particulate matter reductions.[78]

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 benefit-cost analysis 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 $\sum_{t=T}^{\infty}(B_t - C_t)e^{-rt} \leq \varepsilon$, 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:

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

- 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.
- 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, in other words, the tradeoff 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. Tradeoffs (benefits and costs) in this context reflect the preferences of those affected by the policy, and the time dimension of those tradeoffs should reflect the intertemporal preferences of those affected. Thus, social discounting should seek to mimic the discounting practices of the affected individuals.

[77] See EPA's economic analysis for the Final Stage 2 Disinfection and Disinfection Byproducts Rule (US EPA 2005a)

[78] See, for example the recommendations on the distributed lag for particulate matter benefits in EPA-COUNCIL-LTR-05-001 as applied in EPA's Regulatory Impact Analysis for the 2006 Particulate Matter NAAQS www.epa.gov/ttn/ecas/ria.html)

The literature on discounting uses a variety of terms and frameworks, often to describe identical or very similar key concepts. General themes throughout this literature, however, 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 benefit-cost analysis, 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.

Text Box 6.2 - Social Rate and Consumption Rates of Interest

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

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 benefit-cost analysis. 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).

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 tradeoffs. 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, have different levels of assets, and have different opportunities to borrow and invest, the type of interest rate that best reflects marginal time preference will differ among individuals. 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 benefit-cost analysis 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 that later. These factors are combined in the equation:

$$r = \eta g + \rho \qquad (7)$$

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 may be derived from data, the third is unobservable and must be determined.[79] (A more detailed discussion of the Ramsey equation can be found in the inter-generational discounting section of this chapter.)

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-

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[79] The 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. (US EPA 2004c).

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. [80] 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 may 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, equity (e.g., returns to stocks) or some combination of the two. In practice, we typically observe average returns which are likely to be higher than marginal return 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.

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 which represents how society trades and values consumption over time. [81] The adjustment factor is referred to as the "shadow price of capital."[82] Many sources recognize this method as the preferred analytic approach to social discounting for public projects and policies. [83]

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.

[80] 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 this project is that it offers substantial present value net benefits.

[81] 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 (1982) notes, what is really needed is the social rate of time preference, so we use more general terminology. Discounting based on the shadow price of capital is referred to as a “supply side” approach by EPA’s SAB Council. (US EPA, 2004c).

[82] 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.

[83] See OMB Circular A-4, Freeman (2003), and the report of EPA’s Advisory Council on Clean Air Compliance Analysis (US EPA 2004c).

Text Box 6.3 - Shadow Price of Capital

Suppose that the consumption rate of interest is 3%, the pre-tax rate of return on private investments is 5%, 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% 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.

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 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 6.4.[84]

Text Box 6.4 - Shadow Price of Capital Approach

Suppose that the pre-tax rate of return from private investments is 5%, and the post-tax rate is 3%, with the difference attributable to taxation of capital income. Assume as well that increases in government debt displace private investments dollar-for-dollar, and that increased taxes reduce individuals' current consumption also on a one-for-one basis. Finally, assume that the $1 current cost of a public project is financed 75% with government debt and 25% 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% of the $1 current cost (which is the amount of displaced private investment) by the shadow price of capital (assume this is the 1.67 figure from above). This yields $1.2525, to which is added the $.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% consumption rate of interest ($1.5328) minus the $1.5025 social cost.

[84] 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 net present value 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, requiring, among other things, estimates of the social rate of time preference, the social opportunity cost of capital, as well as 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.[85]

Whether or not this adjustment is necessary appears to depend largely on whether the economy in question is assumed to be open or closed, and on the magnitude of the intervention or program considered relative to the flow of investment capital from abroad.[86]

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.[87] 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) and, 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, providing little detailed empirical evidence as to the relationship between the nature and size of projects and capital displacement. Thus, while the approach is often recognized as being technically superior to simpler methods, it is difficult to implement in practice.

Text Box 6.5: Alternative Social Discounting Perspectives

Some of the social discounting 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 some aspects of the future differently from other consequences. The discount rate an individual might apply to a given future benefit or cost, as a result, may not be observable from market prices, interest rates, or other phenomena. This may be the case if the future consequences in question are not tradable commodities. Some evidence from experimental economics also indicates that discount rates appear to be lower the larger the magnitude of the underlying effect being valued is, higher for gains than for losses, and tend 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 discount hyperbolically rather than exponentially, a structure that raises time-consistency concerns. Additional studies have attempted to address the

85 Depending on the magnitudes of the various factors, shadow prices from close to 1 to 3, 20, 100, and infinity can result. 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.

86 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.

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

time-consistency problems that emerge from hyperbolic discounting. 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. [88]

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. For this, historical rates of return, post-tax and after inflation, on "safe" assets, such as U.S. Treasury securities, are normally used, although some may use the return to private savings. Recent studies and reports have generally found government borrowing rates in the range of around 2-4%.[89] Some studies have expanded this portfolio to include other bonds, stocks, and even housing, and this generally raises the range of rates slightly. It should be noted that these rates are realized rates of return, not anticipated, and they are somewhat sensitive to the time periods selected and the classes of assets considered.[90] Studies of the social discount rate for the United Kingdom place the consumption rate of interest at approximately 2% to 4%, with the balance of the evidence pointing toward the lower end of the range.[91]

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. (2008), for example, estimate an intra-generational rate to be 3.5%. 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 inter-generational discounting, which we address in the next section.

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 can be useful for sensitivity analysis and special cases. Estimates of the social opportunity costs of capital are typically in the 4.5% to 7% range depending upon the type of data used.[92]

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[88] 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.

[89] OMB (2003) cites evidence of a 3.1% pre-tax rate for 10-year Treasury notes.).According to the US 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. (2005) suggest 3.71 percent as the real rate of return on 10-year Treasury notes. Newell and Pizer (2003) find rates slightly less than 4% for 30-year Treasury securities. Nordhaus (2008) reports a real rate of return of 2.7% for 20-year Treasury securities. The Congressional Budget Office estimates the cost of government borrowing to be 2%, a value used as the social discount rate in their analyses (US CBO 1998).

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

[91] 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.

[92] OMB (2003) recommends a real, pre-tax opportunity cost of capital of 7% and refers to Circular A-94 (1992) as the basis for this conclusion. Moore, et al. (2004) estimate a rate of 4.5% 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 (US EPA 2004c; US EPA 2007b) recommends using a single central rate of 5% as intermediate between, 3% and 7% rates based generally on the consumption rate of interest and the cost of capital, respectively.

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, then there is probably little displaced investment.[93] Because 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.[94]

6.3 "Inter-generational" 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 inter-generational discounting.

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 intra-generational time horizons, inter-generational investment horizons involve greater uncertainty. Greater uncertainty implies rates lower than those observed in the marketplace, regardless of whether or not the estimated rates are measured in private capital or consumption terms. Policies with very long time horizons often 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 impacts into decision-making. However, there is less 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 inter-generational social discounting. As a starting point, the section first lays out 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 ("inter-generational") time horizons. The need for such modifications arises from several simplifying assumptions behind the conventional discounting procedures described above that 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 the social rate of time preference. Nevertheless, economics can offer important insights concerning discounting over very long time horizons, the implications and consequences of alternative

93 Lind (1990) first suggested this.

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

discounting methods, and some advice on the appropriate and consistent use of the social welfare function approach as a policy evaluation tool in an inter-generational 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 net present value of the flow of utility from consumption over time. [95] 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 it 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 also be equal to the social rate of return on private investments and the consumption rate of interest.

It is worth noting, however, that the optimal growth literature is only one strand of the substantial body of research and writing on inter-temporal 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 an 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):

$$r = \eta g + \rho .$$

The first term reflects the fact that the marginal utility of consumption will change over time as the level of consumption changes. The second term, 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 (e.g., Valdés 1999)—then future generations will be richer than the current generation and therefore increments to consumption will be valued less highly in the future than today due to the diminishing marginal utility of consumption. Thus, 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 in 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 normative (or prescriptive) approach. The descriptive approach attempts to derive likely estimates of the underlying parameters in the Ramsey equation, based on the argument that economic models should be based on actual behavior and that

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

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 may be estimated, allowing ρ to be inferred.

Other economists take what is referred to as a normative approach and assign parameters to the Ramsey equation to match what they believe to be ethically correct.[96] For instance, there has been a long debate on whether the pure rate of time preference should be greater than zero (starting with Ramsey himself). The main responses to the normative approach are (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 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 a very low discount rate advocated by some adherents to the normative approach leads to unethical shortchanging of current and close generations.

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 (1996) 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.

Text Box 6.6 Applying these approaches to the Ramsey Equation

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 DICE model, 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 1994), he estimate the initial return on capital (and social discount) to be 6 percent, the rate of time preference to be 2, 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.

Other analyses have adopted at least aspects of a prescriptive approach. For example, the Stern Review (see Text Box 6.7) 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.

6.3.2 Key Considerations

There are a number of important ways in which inter-generational social discounting differs from intra-generational 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

[96] Arrow, et al. 1996.

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. Hence, while the preferences of each affected individual are knowable (if perhaps unknown in practice) in an intra-generational context, the preferences of future generations in an inter-generational context are essentially unknowable. This is not always a severe problem for practical policymaking, 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 distributed asymmetrically over vast expanses of time. Here the crux of the problem is that future generations are not present to participate in making the relevant social choices. Instead, these decisions will be made only by existing generations. Social discounting in these cases can no longer be thought of as a process of consulting the preferences of all affected parties concerning their valuation today of effects they will experience in different time periods.

Moreover, compounding interest over very long time horizons can have profound impacts on the intergenerational distribution of welfare. An extremely large benefit or cost far into the future has essentially zero present value when discounted at even a low rate. But a modest sum invested today at the same low interest rate 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. [97]

For example, in the climate change context, Pearce et al. (2003) show that decreasing the discount rate from a constant 6% to a constant 4% nearly doubles the estimate of the marginal benefits from CO2 emission reductions. Weitzman (2001) shows that moving from a constant 4% 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 (e.g., a constant discount rate could undervalue net present benefits by 21% to 95% depending on the model of interest rate uncertainty with an initial rate of 7%, and 440% to 700% with an initial rate of 4%.

Using observed market interest rates for inter-generational 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[98] - though not necessarily zero - to (1) correct for market distortions and inefficiencies in inter-generational transfers, and (2) so that generations are treated equally based on ethical principles (Arrow et al., 1996; Weyant and Portney, 1999).

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[97] OMB's Circular A-4 (OMB 2003) requires the use of constant three and seven percent for both intra- and inter-generational discounting for benefit-cost estimation of economically significant rules but allows for lower, positive consumption discount rates if there are important intergenerational benefits/costs.

[98] Another issue is that there are no market rates for intergenerational time periods.

Inter-Generational Transfers

The notion of Pareto compensation attempts to identify the appropriate social discount rate in an inter-generational 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 U.S. presumably, or perhaps the entire world) can transfer wealth across hundreds of years. Some argue that in the U.S. context, a good candidate for this rate is the Federal government's borrowing rate. Some authors also consider the infeasibility of intergenerational transfers to be a fundamental problem for discounting across generations.[99]

Equal Treatment Across Generations

Environmental policies that affect distant future generations can be considered to be altruistic acts.[100] As such, some argue that they should be valued by current generations in exactly the same way as other acts of altruism are valued. For this reason, the relevant discount rate is not that applied to an individual's consumption, but instead that applicable for an individual's valuation of the consumption or welfare of someone else, identified by the analyst through either revealed or stated preference.

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, and decline as the time horizon involved lengthens.

6.3.2.2 Uncertainty

A longer time horizon in an inter-generational 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 what that 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).[101] 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 we can infer a constant, certainty-equivalent discount rate that yields the discount factor (for any given distribution of r). Several methods for accounting for uncertainty into inter-generational discounting are discussed in more detail in the next section.

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[99] See Lind (1990) and a summary by Freeman (2003).

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

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

6.3.3 Evaluating Alternatives

There are a wide range of options available to the analyst for discounting inter-generational costs and benefits. We describe several of these 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, we encourage the analyst to explore a variety of approaches.

6.3.3.1 Constant Discount Rate

One possible approach is to simply make no distinction between inter-generational and intra-generational 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 U.K., the Treasury recommends the use of a 3.5% discount rate for the years 0 - 30, declining to a rate of 3% for years 31 – 75. 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 2002).

6.3.3.3 Declining or Non-constant Discount Rate

Using a constant discount rate in benefit-cost analysis (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.[102] 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 net present value of the policy. Over the long time horizons 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 then dominate the expected net present value calculations for benefits and cost far in the future (Weitzman1999).

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 seven percent could undervalue net present benefits by between 21 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 ignore the effects of uncertainty in economic growth over time. Thus, this approach is not subject to the time inconsistency problems of some of these other approaches.

Text Box 6.7: What's the Big Deal with the Stern Report?

In autumn 2006, the UK government released a detailed report titled *The Economics of Climate Change: The Stern Review*, headed by Nobel Laureate Sir Nicholas Stern (2006). The report drew mainly on published studies to estimate that damages from climate change could result in a 5% to 20% decline in global output by 2100, while costs to mitigate these impacts were significantly less (about 1% 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 Review recommended that policies aimed towards sharp reduction in greenhouse-gas emissions should be enacted immediately.

While lauded for its thoroughness and accurate use of current climate science, the Review drew significant criticism and discussion of how future benefits were calculated, namely Stern's assumptions about the discount rate (Tol and Yohe, 2006; Nordhaus, 2008). The Review used the Ramsey discounting equation (see section 6.3.1), applying rates of 0.1% for the annual pure rate of time preference, 1.3% for the annual growth rate, and a elasticity of

[102] E.g., Nordhaus 2008.

marginal utility of consumption equal to 1. Combining these parameter values reveals an estimated equilibrium real interest rate of 1.4%, 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% or 7% 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 we should save more today to compensate losses incurred by future generations.

Why did Stern use these particular parameter values? First, he argues that we have 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 (2007) 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."

There is little consensus in the economic literature on social discounting for inter-generational policies. In particular, the fundamental choice of what moral perspective should guide inter-generational social discounting - a social planner who weighs the utilities of present and future generations, the preferences of the current generations regarding future generations - cannot be made on economic grounds alone. Additionally, the rule of uncertainty is more important in an inter-generational context, which can have a profound effect on discount rates over the very long run.

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 the need to account for the opportunity costs of capital in practice. Thus, 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.

These recommendations can be used as a starting point for benefit-cost analyses, but if the 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.

- Using 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%.

    - 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 Office of Management and Budget estimates a rate of 7% for the opportunity cost of private capital.

    - EPA intends to review the empirical basis for the consumption discount rate and the rate of return to private capital at regular intervals.

  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 to another -- then also calculate the expected present value of net benefits using the schedule of discount factors estimated by Newell and Pizer (2003) based a stochastic “random walk” model of interest rates (shown in the fourth column of their Table 2; see also Newell and Pizer 2004). This approach is relatively straightforward to implement and accounts for discount rate uncertainty and variability, which are known to have potentially large effects on net present value estimates for policies with long time horizons. EPA will provide an empirical supplement that provides specific rates based on this approach. This does not preclude the use of more detailed approaches that, for example, might construct the discount rate from estimating the individual parameters in the Ramsey equation. However, these 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 time preferences 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).

- Cessation lag and latency should be accounted for in the economic analysis, with the monetary benefits from the expected future impacts--be they changes in human health, environmental conditions, ecosystem services, etc.--discounted at the same rate as other benefits and costs in the analysis.

7 Analyzing Benefits

7.1 Introduction

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.

While the discussion of monetized benefits analysis in this chapter focuses on a "typical" EPA policy, program, or regulation that reduces emissions or discharges of contaminants, the general principles discussed here should apply to other 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.

While this chapter focuses on monetized benefits analysis, it is important to note that there are other methods for evaluating policies. One example is cost-effectiveness analysis, which does not require monetization of benefits but rather divides the costs of a policy by a particular effect (e.g., number of lives saved). Cost-effectiveness analysis can be used to compare proposed policy changes on an effect-by-effect basis, but, unlike benefit-cost analysis, it 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. Methods for evaluating policies (e.g. distributional analyses) are covered in Chapter 9.

Many EPA benefits analyses face several major obstacles. First, a given policy may produce multiple environmental effects, but it is seldom possible to analyze all effects simultaneously in an integrated fashion. In most cases, analysts will have to address each effect individually, and then attempt to aggregate the individual values to generate an estimate of the total benefits of a policy. Although there are exceptions to this "effect-by-effect" approach to benefits analysis, which is described in detail in section 7.3, much of the discussion in this chapter assumes that analysts will need to adopt this approach. A constant challenge in employing an effect-by-effect approach is to balance potential tradeoffs 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 nonuse values, but 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.

A second obstacle analysts often face is the difficulty of conducting original valuation research in support 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 7.4.3, 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 7.4.1 and 7.4.2 include recommendations for evaluating the quality and suitability of published studies for use in benefit transfer.

A third major obstacle sometimes faced in benefits analysis arises from the lack of appropriate analytical tools and/or data with which to apply them. Even though the theory and practice of benefits analysis continue to improve, an off-the-shelf model and data set are usually not available. For this reason, analysts often must either adapt existing tools to the situation using their best professional judgment or simply leave some benefit categories non-monetized.

The rest of the chapter is organized as follows:

- Section 7.2 discusses the economic definition of "value," the major categories of benefits relevant for environmental policies, and some important considerations associated with valuing benefits in each category;
- Section 7.3 describes the main steps in the benefits analysis process;
- Section 7.4 focuses on the final and key step in the process, the economic valuation of environmental changes.

The goals of these sections are to familiarize the reader with the available methods, to provide key references for more detailed information, and to highlight important considerations for judging the quality of studies that use different valuation methods. These considerations will apply whether the study is a new one conducted specifically for the policy being analyzed or a previous study being considered for use in benefit transfer.

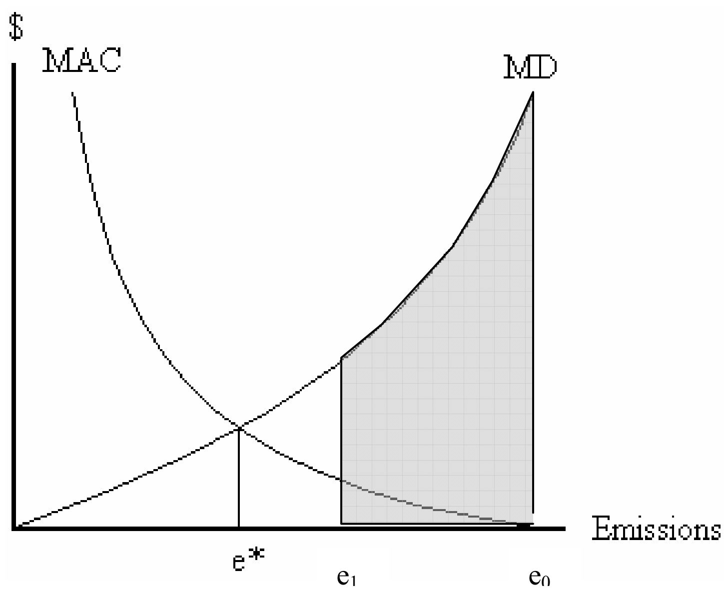
7.2 Economic Value and Types of Benefits

Economic valuation is based on the single, unifying economic theory of human behavior and preferences and focused on measuring the 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, and 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 individuals value and consume. The crucial assumption is that a person can be compensated for the loss of some 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 of one and a little less of the other to achieve a higher level of utility. Thus, market prices can be used to measure the value of market goods and services directly, and the 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 that people choose between on a daily basis.

The benefit of an environmental improvement is 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 curve. Because 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

Figure 7.1



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 7.4 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, but in principle they are all different ways of measuring the same thing, which is utility.

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.[103] 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 benefit-cost analysis 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).

[103] 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.

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.[104] To simplify the presentation, the term WTP is used throughout the remainder of this chapter to refer 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 and WTA are needed for the Kaldor and Hicks potential compensation tests that form the basis of benefit-cost analysis (Boadway and Bruce 1984; Just et al. 1982; Freeman 2003). These tests can be carried out by summing the WTP or WTA for all affected individuals and comparing them 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.[105]

[104] For more information see Appendix A and Hanemann (1991).

[105] See section A.3 of the Appendix for further explanation of Kaldor-Hicks conditions.

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
Nonuse 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.

The types of benefits that may arise from environmental policies can be classified in multiple ways (e.g., 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. The following sections discuss each of the benefit categories listed in Table 7.1 in more detail.

7.2.1 Human Health Improvements

In considering the impact of environmental policy, it is important to note that human health improvements from environmental policies include effects such as reduced mortality rates, decreased incidence of nonfatal 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 valued separately, and are therefore discussed separately in these Guidelines.

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7.2.1.1 Mortality

Some EPA policies will lead to decreases in the risk of premature mortality 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 prematurely 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 $7.0 million (2006$) to value reduced premature mortality for all programs and policies. This value is based on a distribution fitted to twenty-six 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.[106]

Some programs may vary from this default. Recently, for example, air programs have used a $6.6 million (2006$) value originally based on the interquartile range of two published meta-analyses (Viscusi and Aldy 2003; Mrozek and Taylor 2000), and later corroborated by a third meta-analysis (Kochi et al. 2006).[107,108] Any analysis departing from EPA's default VSL should include supporting rationale as well as a clear description of the alternative used and its basis.

At a minimum, 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.

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[106] The studies on which this estimate is based were published between 1974 and 1991, and most are hedonic wage estimates. Although these were the best available data at the time, the Agency is currently considering more recent studies as it evaluates approaches to revise its guidance. See Appendix B for more detail.

[107] See, for example, the economic analysis conducted for the PM NAAQS: http://www.epa.gov/ttn/ecas/ria.html (accesssed May 23, 2008).

[108] Note that the $6.6 million (2006$) is considered an interim value while EPA completes its update of mortality risk valuation estimates and has not been endorsed by either the SAB Council or SAB EEAC.

Valuing mortality risk changes in children is particularly challenging. EPA's *Handbook for Valuing Children's Health Risks* (US EPA 2003b) provides some information on this topic, including key benefit transfer issues when using adult-based studies. *OMB 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 benefit-cost analysis, 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.

Methods for Valuing Mortality Risk Changes

Because individuals appear to make risk-income tradeoffs in a variety of ways, the value of mortality risk changes are estimated using three primary methods. The most common method used is the hedonic wage, or wage-risk, method in which value is inferred from the income-risk tradeoffs made by workers for on-the-job risks. Averting behavior studies have also been used to value risk changes by examining purchases of goods that may affect mortality risk (e.g., bicycle helmets). Finally, stated preference studies have been increasingly used to estimate willingness to pay 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 study may not be fully applicable to the typical environmental policy case.

At one time, reduced mortality risk was valued under a human capital approach that equated the value of 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 willingness to pay 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, et al. (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). For the most part, previous studies have not valued risks in an environmental context, although there are exceptions. EPA applications of VSL are numerous, and include the Clean Air Interstate Rule, the Non-Road Diesel Rule, and the Stage 2 Disinfection By-products Rule (DBP).[109]

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[109] 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);

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

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 effects.

Characterizing and Measuring Mortality Effects

Reduced mortality risks are typically measured in terms of "statistical lives saved." This measure is the aggregation of many small risks over an exposed population. Suppose, for example, that a policy affects 100,000 people and reduces the risk of premature mortality by one in 10,000 for each individual. Summing these individual risk reductions across the entire affected population 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 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 who each had 20 years of remaining life expectancy. The policy would then save 200 statistical life years (10 statistical lives * 20 life years each). In practice, estimating statistical life years saved requires risk information for specific subpopulations (e.g., age groups, health status). It is typical to use statistical life years saved in cost-effectiveness analysis, but valuing a statistical life year remains a subject of debate in the economics literature. Theoretical models show that the relationship between willingness to pay and age, baseline risk and the presence of co-morbidities is ambiguous and empirical findings are generally mixed (US EPA 2006d).

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, 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 (e.g., Jones-Lee, et al. 1993), but others find this effect to be small if it exists at all (e.g., Alberini, et al. 2004). Still others suggest older populations have higher WTP (e.g., Kniesner, et al. 2006). Appendix B contains a more complete discussion of risk and population characteristics and how they may affect WTP.

Timing of Health Effects

Environmental contamination may cause immediate or delayed health effects, and the value of avoiding a given health effect likely depends on whether it occurs now or in the future. Recent empirical research confirms that workers discount future risks of fatal injuries on the job; that is, they are willing to pay less to reduce a future risk than a present risk of equal magnitude (Viscusi and Moore 1989; Cropper et al. 1994). For more information on discounting, see Chapter 6.

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, to fetal loss. 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.

Willingness to pay to reduce the risk of experiencing an illness is the preferred measure of value for morbidity effects (see section 7.2). 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.

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 may 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) and the degree to which they account for all of the components of total WTP. The three primary methods used most often to value morbidity in an environmental context are stated preference (section 7.4.2), averting behavior (section 7.4.1.4), and cost-of-illness (COI) (section 7.4.1.5). Hedonic methods (section 7.4.1.3) have been 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 tradeoffs, for example, do not directly estimate dollar values for risk reductions, but rather, provide rankings of relative risks based on consumer preferences. Risk-risk tradeoffs may be linked to WTP estimates for related risks.[110]

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

[110] EPA analyses have, for example, used risk-risk tradeoffs 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).

and quantity of life lived under various scenarios are often used for cost-effectiveness or cost-utility analyses. These, however, cannot be directly related to WTP estimates as these 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 benefit-cost analyses (Hammitt 2003; IOM 2006). Another commonly suggested alternative is jury awards, but these generally should *not* be used in benefits analysis, for reasons explained in Text Box 7.1.

Previous Studies

A comprehensive summary of existing studies of morbidity values is beyond the scope of these guidelines. Here we provide 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. Tolley et al. (1994) and Johansson (1995) are useful general references for valuing non-fatal health effects. EPA's *Handbook for Non-Cancer Valuation* (US EPA 1999b) 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* (US EPA 2007c) includes estimates for many cancers, developmental illnesses, disabilities, and other conditions. The *Benefits and Costs of the Clean Air Act* (US EPA 1997a) draws upon a number of existing studies to obtain values for reductions of a variety of health effects, describes how the central estimates were derived, and attempts to quantify the uncertainty associated with using the estimates. Other studies may be available through the Environmental Valuation Reference Inventory (EVRI) maintained by Environment Canada and containing over 1,100 studies that can be referenced according to medium, resource, stressor, method, and country.[111]

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.

These two considerations are discussed in the paragraphs that follow.

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.”[112] Severity may also be described in terms of health state indices that combine multiple health dimensions into a single measure.[113] For duration, the primary distinction is between acute effects

[111] See www.evri.ca for more information.

[112] 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.

[113] The difference in the indices is intended to reflect the relative difference in disutility associated with symptoms or illnesses. These indices may be constructed in a number of ways, but consistency with welfare economics requires affected individuals to define these relative tradeoffs for themselves rather than having them

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 frequency of effects also can vary widely across illnesses. Some effects are one-time events, such as a gastrointestinal illness, that are unlikely to recur. Other effects do recur or can be aggravated regularly (e.g. asthma), 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 may 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. (Refer to section 7.4.1.5 and Text Box 7.1 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 part 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 Valuation* (US EPA 1999b).

7.2.2 Ecological Improvements

Environmental policies can lead to ecological improvements that may benefit people in a variety of direct and indirect ways. Ecological improvements can benefit people indirectly by increasing the delivery of “ecosystem services,” which are the end products of ecological functions important to humans (Daily 1997, Balmford et al. 2002, NRC 2005, Banzhaf and Boyd 2005, Boyd and Banzhaf 2006). Such valuable ecological functions include the partial stabilization and moderation of climate conditions, the regulation of water availability and quality, and nutrient retention (Daily 1997). Through their effect on ecosystem services, ecological improvements may lead to improved agricultural yields, recreational opportunities, human health or other types of benefits. For example, protecting wetlands and the natural flow regulation and water purification services they provide may lead to enhanced recreational fishing or swimming opportunities in connected water bodies, reduced flooding in downstream residential areas, or reduced incidences of illness from contaminated drinking water. Ecological improvements may also benefit people directly through aesthetic improvements or through increases in “nonuse” values (e.g., NRC 2005), which can arise from a variety of motivations including an intrinsic concern for the existence of species populations or ecosystems in a relatively undisturbed state or a desire to preserve healthy ecosystems for future generations.

Methods for Valuing Ecological Improvements and Previous Studies

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determined by health experts. Examples of economic analyses that have employed some form of health state index include Desvousges et al. (1998) and Magat et al. (1996).

Economists have used a variety of standard valuation methods for estimating WTP for ecological improvements, many of which are discussed in detail in Section 7.4. Economic methods that have been used to value ecological improvements include production or cost function approaches (e.g., Barbier and Strand 1998, Adams et al. 1997, Acharya 2000, Pattanayak and Kramer 2001), travel cost models (e.g., Herriges and Kling 1999, Haab and McConnell 2002), hedonic property models (e.g., Smith and Huang 1995, Leggett and Bockstael 2000, Irwin 2002; and Thorsnes 2002), and stated preference surveys (e.g., Kopp et al. 1994, Layton and Brown 2000).

Bioeconomic modeling, which involves combining models of species population or ecosystem dynamics with economic models of human behavior, is another approach that can potentially be used to value ecological improvements. Most bioeconomic models have been applied to fishery and forestry management problems, but in many cases these models could be adapted to estimate WTP for environmental improvements that may affect the growth rates or carrying capacities of the focal species. Clark (1990) is the seminal text on bioeconomic modeling, and Conrad (1999) provides a practical introduction.[114]

Important Considerations

The analyst should keep three important considerations in mind when estimating benefits of ecological improvements:

- Defining the commodity;
- The potential for double-counting; and
- Non-economic methods.

These three considerations are discussed in the paragraphs that follow.

Defining the Commodity

Identifying relevant ecological endpoints that can be readily quantified for use in benefits analyses is not always straightforward and may require input from different disciplines beyond economics. A wide variety of ecological endpoints could be taken as relevant for valuation and agreement has not yet emerged as to which should be the focus for benefit-cost analysis. As in the case of human health, endpoints of interest include organism-level effects such as mortality risks or developmental abnormalities, but for a wide-range of non-human species. Other potentially relevant ecological endpoints include population-level effects such as reduced abundances and species ranges, community-level effects such as the reduction of species richness, and ecosystem-level effects such as reductions in the rates of nutrient cycling. Thus, in an ecological benefits assessment, interdisciplinary collaboration is especially important in the "problem formulation" phase when choices about the ecological endpoints to be valued must be made. Analysts are encouraged to seek input from ecologists, risk assessors and other scientists to accomplish this step. Analysts may wish to consult EPA's Ecological Benefits Assessment

[114] See also Crutchfield and Pontecorvo (1969), Hammack and Brown (1974), Freeman (2003), Perrings et al. (1995), Sanchirico and Wilen (1999), and Finnoff and Tschirhart (2003). Other general references related to valuing ecological improvements include *Ecosystem Functions and Human Activities: Reconciling Economics and Ecology* (Simpson and Christensen 1997), *A Framework for the Economic Assessment of Ecological Benefits* (USEPA 2002), *Economics and Ecological Risk Assessment: Applications to Watershed Management* (Bruins and Heberling 2005), *Valuing Ecosystem Services: Toward Better Environmental Decision-Making* (NRC 2005), and the *Ecological Benefits Assessment Strategic Plan* (USEPA 2006a).

Strategic Plan (US EPA 2006a) which has as its goal "enhancing EPA's ability to identify, quantify, and value the ecological benefits of existing and proposed policies."

Potential for Double-counting

Because many ecological functions serve intermediate roles in the production of final goods and services enjoyed by people, it is important to avoid double-counting the value of those functions. See Boyd and Banzhaf (2006) for discussion of this general point and the related distinctions between ecological functions and services.

Non-WTP Approaches

A variety of alternative, non-economic approaches for "valuing" ecological improvements have been used by previous researchers, including approaches based on an ecosystem's "embodied energy" (e.g., Odum 1996; Pimentel et al. 1997) or the replacement cost of entire ecosystems (e.g., Ehrlich and Ehrlich 1997). For example, one high profile study that relied heavily on replacement costs reported an aggregate value in excess of world income (Costanza et al. 1997); however, this estimate cannot be a valid measure of WTP because WTP cannot exceed expendable wealth (Pearce 1998; Bockstael et al. 2000). Furthermore, the only meaningful estimate of WTA in this case would be infinity because what was valued was the totality of all ecosystem functions, without which human life on earth would not be possible (Toman 1998). These and other approaches not based on WTP or WTA are not compatible with standard economic benefit-cost analysis and should not be confused or combined with economic valuation methods (e.g., Shabman and Batie 1978; Dasgupta 2002). Text Box 7.1 provides examples of non-WTP methods.

Text Box 7.1 - Non-WTP Measures

Economic measures of value calculate willingness to pay (WTP) for environmental changes. WTP is a valid measure because it is directly related to utility. WTP is defined as that amount of money which, 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").

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. We consider some examples 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, however, 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 *under*estimate the economic consequences of environmental degradation. If the second condition is not met replacement costs might *over*estimate 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, affected people might be made whole simply by giving them enough money to buy substitutes for the fish they caught 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 (1973) calculated the number of barrels of petroleum that would be required to provide the energy to replace the services of wetland ecosystems. This number is, however, 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 (e.g., Rees 1989). Partha 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 Paul Samuelson's (1951) "non-substitution theorem."

Cost-of-illness. Health effects are often proxied by the "cost of illness" (COI), which are the total costs of treatment and time lost due to illness. Although COI is discussed in greater detail in Section 7.4.1.5, we 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 are the awards made by juries. 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, then, "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. Guilty defendants are made examples of to deter others. For this reason, jury awards may overstate typical damages.

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 Class I 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. For example, 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 will typically begin 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 will be used to relate local concentrations of pollutants to corrosion, soiling, or other physical damages that will affect the production (inputs) or consumption (output) of the material service flows. The market response to these impacts then 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 (US EPA 1997e).

7.3 The Benefits Analysis Process

This section discusses the main steps in the benefits analysis process. The discussion is framed in terms of the general "effect-by-effect" approach to benefits analysis mentioned in section 7.1.[115]

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:

[115] Note that, if original studies are pursued, it may be possible to analyze multiple effects in an integrated fashion.

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 should also 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. 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 policymakers. 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 may have detrimental effects on natural resources (e.g., affecting oxygen availability in surface water or reducing crop yields) as well as direct or indirect effects on human health (e.g., affecting cancer incidence through direct inhalation or through ingestion of contaminated food).

Consider the potential change in these effects as a result of 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:

- Which benefit categories are likely to differ across policy options (including the baseline option)? An assessment of how the physical effects of each policy option will differ and how each physical effect will impact each benefit category should be conducted.
- 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.

- 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[116]

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 (US EPA 2005). 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 a 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.

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

[116] This criterion relates to equity considerations detailed in Chapter 9.

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.

Text Box 7.2 - Integrating Economics and Risk Assessment

Historically, health and ecological risk assessments have been designed to support the setting of standards or to rank the severity of different hazards, and not to support benefits analyses. . As a result, traditional measures of risk often are 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. In addition, these measures often are 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.

However, 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 economist's 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:

- Agree on a set of human health and ecological endpoints that have economic meaning, i.e., endpoints that can be linked directly 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 may also 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 (US EPA 1997e), and the 1997 Guiding Principles for Monte Carlo Analysis (USEPA 1997d).

Step 3: Estimate the values of the effects

The next step is to estimate 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. That is, we calculate an average estimate of WTP for a sample of people in the relevant population and then, assuming that all others in the population hold similar values, we 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 may be applied can be found in section 7.4. 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, however, and any potential overlap should be noted when presenting the results. The discussion of benefit transfer in section 7.4.3 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 also include a quantitative uncertainty assessment when possible. Guiding principles for presenting uncertainty are addressed in Chapter 10.

7.4 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 *nonmarket* inputs, which are, by definition, not traded in markets.[117] 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

[117] 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.

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. We also discuss cost of illness methods in this section, which are sometimes used to value human health effects when estimates of willingness to pay 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.

7.4.1 Revealed Preference Methods

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

- Production or cost functions;
- Travel cost models;
- Hedonic models; and
- Averting behavior models
- Cost of Illness.[118]

7.4.1.1 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.[119] Marginal changes, however, may be evaluated by considering the production side of the market alone.

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 utility obtained from the consumption good in whose production the input is employed by the marginal product of the input. The marginal utility of a consumption good is

[118] Although not a revealed preference method as it does not measure WTP, we discuss COI methods in this section since estimates are based on observable data.

[119] See Appendix A for more detail.

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, however, 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 (e.g., Varian 2005, for an explication of the relationships among these functions).

The value of a marginal change in some environmental condition can, then, 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.[120] It should be noted, however, that problems of data availability and reliability often arise. These problems may motivate the choice among these conceptually equivalent approaches, or in favor of another.

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 would need 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 would need 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.[121]

In the statements above it has been emphasized 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 will require consideration of changes in both producer and consumer surplus. The latter will necessitate application of techniques (e.g., travel cost, hedonics, and stated preference) discussed elsewhere in this chapter.

Before moving on to those topics, there is 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 can also 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

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

[121] Derivation of marginal values often involves an application of the “envelope theorem”: the principle that effects from variables which are already optimized are negligible. So, for example, 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 $\left[p(Q) - \partial C/\partial Q\right]dQ/de - \partial C/\partial e = - \partial C/\partial e$, where the last equality results because competitive firms set price equal to marginal cost. This is the basis for the general proposition that *marginal* values can be estimated by looking solely at the production side of the market.

terms "rent" and "profit" are often used synonymously in economics), and its purchase price 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 7.4.1.3. 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, we might also say that developers "produce" housing services when they build residences. Therefore, hedonic pricing approaches may 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 has also assessed the effects of water quality changes on water supply treatment costs and on the production costs of industry processors, irrigation operations, and commercial fisheries.[122] Production, cost, or profit functions have also found interesting applications to the estimation of some ecological benefits.[123] Probabilistic models of new product discovery from among diverse collections of natural organisms can also be regarded as a type of "production".[124] Finally, work in ecology also points to "productive" relationships among natural systems that may yield insights to economists as well.[125]

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, as data on prices are often more complete than are data on quantities, and

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

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

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

125 For example, see e.g., Tilman, et al. (2005).

because 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 may be regarded as second-order approximations to any production technology. The translog and generalized Leontief specifications are examples. Estimation will often 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 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 that originated with Buchanan (1969), analysts should exercise caution and restraint in attempting to correct for departures from competitive behavior. 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 will typically be split between private producers and the general public through tax collection. The issues here can also 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.

7.4.1.2 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. For example, 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 a traveler makes for recreational purposes. In most cases, 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 and time (opportunity) costs. 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 time cost is the income forgone in order to take the time to recreate. A variety of approaches have been used in the literature to estimate the opportunity cost of time, and to date no single approach is widely accepted. Researchers have used anywhere from one third to one hundred percent of a person's hourly wage as their hourly opportunity cost of time depending on assumptions about how freely individuals are able to substitute labor and leisure.[126] 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.

People are assumed to take a trip as long as their expected utility gained from taking a trip to a given site is greater than the cost to them. Following the law of demand, as the cost of a trip increases the quantity of trips demanded generally falls, all else equal. In practice 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 (e.g., parking, restrooms, boat ramps), participant characteristics (e.g., age, income, experience, work status), and environmental quality can also 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.

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 a number of 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) for a detailed discussion of 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 willingness to pay 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 (i.e., 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, for example, as is often the case, a policy affects several recreation sites in a region, traditional single site models are required for each site. Each single site model also needs to include the price of all relevant substitute sites as explanatory variables (or risk biasing estimates). In cases with large numbers of sites, defining the appropriate substitute sites for each participant and estimating individual models for each site may impose overwhelming data collection and computational costs. Because of these difficulties, most

[126] See discussion below on "Considerations in Evaluating and Understanding Recreation Demand Studies" as well as Text Box 7.4 for more information.

researchers have opted to refrain from using the single site models when examining situations with large numbers of substitute sites.[127]

Multiple Site Models. The most common multiple site models are random utility maximization (RUM) travel cost models. RUM's model a recreator's choice of which site to visit from a set of available sites on a given choice occasion. Each site in the recreator's choice set is assumed to provide the recreator with a given level of utility, and the recreator is assumed to choose to visit the site that provides the highest level of utility. The characteristics of each of the available sites – such as the amenities available at each site, including environmental quality, and the travel costs to and from the site – are assumed to affect the utility of visiting each site. Because people generally do not choose to recreate at every opportunity, a non-participation option is also often included.[128] By examining how recreators trade off the differing levels of each site characteristic and travel costs, it is possible to place a per trip dollar value on each of the characteristics and on the site as a whole.

Due to the discrete nature of the data (visit or no visit for each site), RUM models are often estimated using logit models.[129] Using data on the characteristics of each site and participant, and data on participants' actual choices, the RUM model predicts the probability that a recreator would choose to visit a site on any given choice occasion. Desirable characteristics, such as good environmental quality or low travel costs, should increase the probability of a visit. The estimated probabilities may be translated into participants' "maximum expected utility" from a trip. To simulate the welfare effects of a change in site quality or access, a participant's maximum expected utility is calculated separately under the baseline and changed access or quality conditions. The difference in the expected utilities between the two situations is then monetized by dividing it by the estimated marginal utility of income (i.e., the travel cost coefficient) to produce the change in participant welfare.

Compared to the single site model, the strength of the RUM model is its ability to account for the availability and characteristics of substitute sites when estimating welfare changes. Using a RUM model, it is possible to estimate the welfare effects of changes in access or site quality at single site or at multiple sites simultaneously. However, because the RUM model estimates 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 (2003) for a detailed discussion of methods of incorporating seasonal participation estimates into the RUM framework.

[127] 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, with the number of trips to a given site used as a function of the cost of visiting that site as well as the cost of visiting other available sites. See Bockstael, et al. (1991) and Shonkwiler (1999) for more discussion and examples of extensions of the single site model.

[128] 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.

[129] See Parsons (2003) and Haab and McConnell (2003) for a detailed discussion of the logit model in this context.

While RUM logit models are by far the most common multi-site models in the literature, several alternative models have recently gained acceptance. One of the most versatile, the Kuhn-Tucker model, estimates recreators' *demand for a set of sites over a season*, (rather than modeling a recreator's choice of which site to visit from a set of available sites on a given choice occasion like a RUM model). The model is built on the theory that people maximize their utility subject to their budget constraint by purchasing those recreation and other goods that give them the greatest utility. In this fashion, the model simultaneously estimates both the sites a person visits (like a multi-site model), and how many times the person will visit each site (like a single site model). While recent applications have shown that the Kuhn-Tucker model is capable of accommodating a large number of substitute sites, the model is computationally intensive compared to traditional models. For examples of the Kuhn-Tucker model see Herriges et al. (2000) and von Haefen and Phaneuf (2004).

Considerations in Evaluating and Understanding Recreation Demand Studies

Definition of a site. Ideally, one could estimate a recreation demand model in which sites are defined as specific points (such as exact fishing location, campsites, etc) because the more exact the site definition, the more exact the measure of travel costs, and therefore WTP, that can be calculated. However, the data requirements of detailed models are large and may be cost and time prohibitive. Similarly, for a given site the range of alternative sites may vary by individual. Ultimately, every recreation demand study strikes a compromise in defining sites, balancing data needs and availability, costs, and time.

Opportunity cost of time. Defining the value of time is an important component of the travel cost models. If individuals have a flexible time schedule and are able to freely substitute labor and leisure, then their opportunity cost of time is equivalent to their wage rate. However, given that many people are constrained in these choices, researchers often assume that the opportunity cost of time is less than the full wage rate. See Feather and Shaw (1999) and Freeman (2003) for further discussion. It is important to understand the choices made in defining the opportunity cost of time. See Freeman (2003) for a detailed discussion of the major issues. Text Box 7.5 also provides additional information.

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.[130] See Parsons (2003) for further discussion.

Day trips versus multi-day trips. The recreation demand literature has almost exclusively focused on modeling 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, 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 bullet). 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

[130] Excluding any type or class of trip (like multi-site or multi-purpose) 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.

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.

7.4.1.3 Hedonics

Hedonic pricing models use statistical methods to measure the contribution of a good's characteristics to its price. Many economic analyses assume that goods traded in markets are homogeneous; however, this is not always the case. 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 their attributes (e.g., risk levels, supervisory nature, etc.) that should be reflected in wages. Hedonic pricing models are commonly used to value the characteristics of properties or jobs using variations in property prices or wages. 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, for example, a cleaner neighborhood, are observable, then that price difference can be interpreted as the willingness to pay 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.[131] 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 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.

[131] Michaels and Smith (1990) offer guidance for defining the extent of the market.

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 (e.g. Leggett and Bockstael 2000), natural amenities (e.g. Irwin and Bockstael 2002, Thorsnes 2002), land contamination (e.g., 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 "value of a statistical life." Even if the risks workers face on the job are not caused by environmental factors, estimates of the wages they would forego to escape such risks can be used to value their aversion to potentially life-threatening environmental circumstances.

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 tradeoffs between wages and occupational risks of death or injury. Most analysts believe 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, et al. 1988). A thorough treatment of the hedonic wage model that includes many of these considerations can be found in Viscusi and Aldy (2003). Black and Kneiser (2003), however, question the ability of hedonic wage studies to measure job risks in general due to measurement error and bias problems. Further, while estimates from the hedonic wage literature have been relatively consistent over the years, questions persist about their applicability to environmental benefits assessment.[132] Hedonic wage studies have been used most frequently in benefits assessments to estimate the value of fatal risks. That is, 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 preferred VSL estimate, see section 7.2.1 and Appendix C.[133] The value of a statistical life determined by a hedonic wage study, for example, typically relates willingness to accept higher wages in exchange for the increased likelihood of accidental death during a person's working years. However, care should be taken 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. 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) data. The NIOSH data are state level data of fatalities by occupation or industry, while the BLS data provide a finer resolution of occupation or industry fatalities, but do not vary by location.

Hedonic property value studies measure the different contributions of various characteristics to the value of property. These have typically been conducted using residential housing data, but they have also been applied to commercial and industrial property, agricultural land, and vacant land.[134] Bartik (1988) and Palmquist (1988, 1991) provide detailed discussions of benefits assessment using hedonic methods.

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[132] For example, EPA's Science Advisory Board 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 2000b)

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

[134] See Xu, et al. (1993) and Palmquist and Danielson (1989) for hedonic values of agricultural land; Ihlanfeldt and Taylor (2004) for commercial property; Dale, et al. (1999) and McCluskey and Rausser (2003) for residential property; and Clapp (1990) and Thorsnes (2002) for vacant land.

Property value studies require large amounts of disaggregated data. Market transaction prices on individual parcels or housing units are preferred to aggregate data such as census tract information on average housing units to avoid aggregation problems. Problems may 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 Box 7.3 on Spatial Correlation for more information).

Text Box 7.3 - 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 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 recently as the capability for assessing such locational relationships within hedonic property data has improved, 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, 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).

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 (1996) use a hedonic analysis to determine the tradeoffs between automobile price and safety features to infer the value of a statistical life, and Ashenfelter and Greenstone (2004) relate legislated changes in driving speed limits to a public expression of the value of statistical life.

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 risks. 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. If in relating the wage premium paid on dangerous jobs to the statistics on premature mortality we fail to distinguish between causes of death—between on-the-job accidents and environmentally induced conditions acquired at home, for example—analysts might underestimate the wage premium demanded on the job. Conversely, if the same job poses multiple risks – say the risk of both accidental death and serious, but non-fatal injury were higher on a particular job – the wage premium the job offers would overstate willingness to pay for reductions in mortality risks if the injury risks were not properly controlled for in the analysis.

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 of the hedonic price function can be interpreted as willingness to pay 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, et al. (2004) for recent treatments.

7.4.1.4 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 – the method is sometimes referred to as the "defensive behavior method" – may 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 defensive behaviors economists can attempt to estimate the value individuals place on small changes in risk (Shogren and Crocker 1991, 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 (i.e., defensive purchases) (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, 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 (e.g., 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, 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 in order to determine the benefits individuals perceive they are receiving when engaging in defensive activities. These perceived benefits can then be used as the object of the valuation estimates. For example, if perceived risks are found to 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 (e.g., age, health status, 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 (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.

7.4.1.5 Cost of Illness

A frequently encountered alternative to willingness-to-pay 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 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 cost-of-illness estimates in support of a specific policy.

Economic Foundations of Cost of Illness 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 (e.g., 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), whereas these costs may be overlooked in WTP estimates -- especially when derived from consumers or employees covered by sick leave.

Available comparisons of cost-of-illness 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 Cost of Illness 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 willingness-to-pay 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.

Text Box 7.4 - 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 who 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 than the case of employed victims. 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, OMB guidance (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.

Most existing cost-of-illness studies estimate indirect costs based on the typical hours lost from a work schedule or home production, evaluated at an average hourly wage. The direct medical costs of illness are generally derived in one of two ways. The empirical approach 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. The EPA's *Cost of Illness Handbook* (U.S. EPA, forthcoming) provides estimates for many cancers, developmental illnesses and disabilities, and other illnesses.

Considerations in Evaluating and Understanding Cost-of-Illness 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 cost-of-illness studies, the analyst should be sure to research whether and how the generally accepted treatment has changed from the time of the study.

Measuring the value of lost productivity. 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 included in most cost-of-illness studies. (See Box 7.5 for a discussion of the value of time.)

7.4.2 Stated Preference[135]

The distinguishing feature of stated preference (SP) methods compared to revealed preference (RP) methods is that SP methods rely on data drawn from people's responses to hypothetical questions while RP methods rely on observations of actual choices. SP 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 SP methods include their ability to estimate nonuse values and to incorporate hypothetical scenarios that closely correspond to a policy case. The main disadvantage of SP methods is that they may be subject to systematic biases that are difficult to test for and correct.

The Report of the NOAA Panel on Contingent Valuation (Arrow et al. 1993) 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. While still useful, the NOAA panel recommendations do not completely reflect nor address current stated preference issues.

7.4.2.1 Economic Foundation of Stated Preference Methods

The responses elicited from SP surveys, if truthful, are either direct expressions of willingness to pay or can be used to estimate willingness to pay for the good in question with minimal additional assumptions. However, the "if truthful" caveat is paramount. While many environmental economists believe that respondents can provide truthful answers to hypothetical questions and therefore view SP methods as

[135] This section based in part on Stratus (2000).

useful and reliable if conducted properly, a non-trivial fraction of economists are more skeptical of the results elicited from SP surveys. Due to this skepticism, it is important to employ validity and reliability tests of SP results when applying them to policy decisions.

If the analyst decides to conduct an SP survey or use SP results in a benefit transfer exercise, then a number of survey design issues should be considered. SP researchers have attempted to develop methods to make individuals' choices in SP 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 SP value estimates. Three components of market transactions need to be constructed in SP surveys: the good, the payment, and the marketplace (Fischoff and Furby 1988).

SP studies thus need to carefully define the commodity to be valued, including the 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.[136]

Respondents also must be informed about the transaction context, including the method, timing, and duration of payment; the transaction must be uncoerced; 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 also be described so as to minimize potential strategic behavior such as "free-riding" or "overpledging." In the former case, 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 latter case, respondents pledge amounts greater than their true WTP with the expectation that they will not be made to pay for the good but believe their response could influence whether or not the good will be provided.

It is recognized in both the experimental economics literature and the survey methodology 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.

7.4.2.2 General Application by Type of Stated Preference Study

Two main types of SP 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 SP survey formats is described in detail below.

[136] Depending on the scenario, the description of the source may produce strong reactions in respondents and could introduce bias. In these cases, the detail with which the source of the change is specified will need to be balanced against the ultimate goals of the survey. Regardless the source will need to be specified with enough detail to make the scenario credible.

Goals that should guide selection of the survey format include the minimization of survey costs, nonresponse, unexplained variance, and complications associated with WTP estimation. For example, open-ended questions are simpler to analyze than other methods of asking the valuation question and require smaller sample sizes. These advantages could lead to significant cost reductions. However, these advantages may be mitigated by higher nonresponse 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 WTP Questions

Direct/open-ended WTP questions ask respondents their maximum WTP for the good or service that has been described to them, including specific quantity or quality changes. An important advantage of open-ended SP questions is that the answers provide direct, individual-specific estimates of WTP. Although this is the measure that economists want to estimate, early SP studies found that some respondents had difficulty answering open-ended WTP questions and nonresponse 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 SP 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). 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) discuss concerns that the range and intervals of the dollar amounts used in payment card methods may influence respondents’ WTP answers.

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. (Thus, when analyzing the data the dependent variable will be continuous for open-ended WTP formats and discrete for stated choice formats.)[137] 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, "live in area A or area B); two alternatives with an 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, SP 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, SP researchers are able to extract WTP estimates for incremental changes in the attributes of the good, based on the preferences expressed by the respondents. Adamowicz et al. (1998b) 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. Because less information is obtained for each respondent than with direct/open-ended question formats (i.e., only an interval

[137] 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 we use the term "stated preference" to refer to all valuation studies based on hypothetical choices (including open-ended WTP and stated choice formats), as distinguished from "revealed preference."

containing WTP is known, not WTP), significantly larger sample sizes are needed for dichotomous choice questions to obtain the same degree of statistical efficiency in the sample means (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.[138] 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 Quiggen 1994, 1998; 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 (e.g., Johnson et al. 1995; 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 it would be paid need to be explained clearly and plausible, as with any payment mechanism in a SP study.

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, 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.[139] Asking repeated choices also 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.

Some applications of multi-attribute survey formats include Opaluch et al. (1993), Adamowicz et al. (1994), Viscusi et al. (1991), Adamowicz et al. (1997), Morey et al. (2002), Adamowicz et al. (1998a), Layton and Brown (2000), Johnson and Desvousges (1997), 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).

[138] Alberini (1995a) 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.

[139] Yea-saying refers to the behavior of respondents when they overstate their true willingness to pay in order to show support for a situation described in survey questions.

7.4.2.3 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 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 answer questions over the phone. The same may 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 U.S. 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 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 also 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 willingness to pay 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; Brown et al. 1996; Balistreri et al. 2001). It has not been determined whether yea-saying may 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. SP 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 thing, 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 SP 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 SP 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 measure 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.

Validity tests seek to assess whether WTP estimates from SP 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 measure 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 measure 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. 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 (see Bishop et al. 1997).

- **Criterion Validity.** Criterion validity assesses whether SP results relate to other measures that are considered to be closer to the concept being assessed (WTP). Ideally, one would compare results from a SP 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 an SP survey. Another approach would be to estimate a sample of individuals' WTP for a commodity using an SP 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. 1987; Kealy et al. 1990; Brown et al. 1996; and Champ et al. 1997 for examples).

- **Convergent Validity.** Convergent validity examines the relationship between different measures of a concept.[140] 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 their 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 SP results are found to be larger than RP results for the same good, it is often presumed that the difference is the result of hypothetical bias because RP results are based on actual behavior. However, there can be many other sources of bias and error in both SP and RP results that cause them to differ from one another and "true" WTP.

  Empirical convergent validity tests use comparisons of SP results with RP or experimental results that are thought to be free of hypothetical bias.[141] 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 Text Box 7.7 for a discussion on combining RP and SP data.

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

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[140] 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.

[141] Some analysts include the comparisons of SP 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.

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. (See McClelland et al. 1991)

- **Survey Non-response Bias** is created by those who refuse to take the survey. 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.[142] 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 most common is to examine the relevant measurable characteristics of the respondent group, such as income, resource use, gender, age, etc., and 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 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).

[142] 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" (page 60-70).

Text Box 7.1 - Combining Revealed and Stated Preference

In some cases RP and SP data can be combined. This can allow analysts to generate more efficient estimates of preference parameters than could be estimated by either approach alone, and to estimate other preference parameters that cannot be estimated by RP alone (e.g., those that pertain to nonuse values specifically). At the same time it provides information to evaluate the convergent validity of SP and RP methods.

Morikawa et al. (2002) elaborate on the advantages of combining SP and RP data:

> "Since SP data are collected in a fully controlled 'experimental' environment, such data have the following advantages in contrast with RP data that are generated in natural experiments: 1) they can elicit preferences for nonexisting alternatives; 2) the choice set is prespecified; 3) multicollinearity among attributes can be avoided; and 4) the range of attribute values can be extended."

Further, because SP data allow the researcher to control more variables and because there are more unknowns influencing the decisions in RP data, the SP data often contain less noise and measurement error (Louviere 1996). Collecting RP data in a survey along with SP data also can help respondents prepare for answering the SP questions by having first reviewed their actual choices and reasons for them.

There are some analytical challenges that need to be addressed when combining different types of data because different data sources may mean the data are not directly comparable. The ensuing issues, and how to address them, are beyond the scope of these guidelines. However, they have been discussed and examined in several studies that have combined these data including Ben-Akiva and Morikawa (1990), Cameron (1992), Hensher and Bradley (1993), Adamowicz et al. (1994, 1997), Ben-Akiva et al. (1994), Swait et al. (1994), Louviere (1996), and Kling (1997). Whitehead et al. (2008) provide review the literature on combining RP and SP data. The literature on benefit-transfer, discussed below, has also examined issues associated with combining the results of SP and RP studies.

7.4.3 Benefit Transfer

Benefit transfer refers to the use of estimated nonmarket 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 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 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. Still, 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. In doing so, 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.[143] 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.

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 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, older adults across the U.S.) if possible. 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.”[144] Including studies from the gray literature may also help mitigate “publication bias” that results from researchers being more likely to submit and/or editors being more likely to publish studies that demonstrate “strong” results.[145] Online searchable databases summarizing valuation research may be especially helpful at this stage.[146]

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. The commodity definitions, the baseline and extent of the change, and the affected populations for the study and policy cases should be

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[143] 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.

[144] Peer review of benefit transfer studies using gray literature is highly advisable.

[145] 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.

[146] For example, the Environmental Valuation Reference Inventory (EVRI) is maintained by Environment Canada and managed by a working group that also 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.

comparable. The analyst should determine whether adjustments should and can be made for important differences between each study and policy case. It often will be useful for the analyst to discuss her interpretation and intended use of the study case with the original authors. See Desvousges et al. (1992) for additional information on criteria used to determine quality and applicability. For more information on applicability as related to specific benefit categories, see Desvousges et al. (1998), the draft *Handbook for Non-Cancer Valuation* (US EPA 1999b), and the *Children's Health Valuation Handbook* (US EPA 2003b).

No single study needs to match perfectly with the policy case. Rather, each study case should inform at least some aspect of the policy decision. Of course, results from study cases ought to be valid as well as relevant. That said, analysts should avoid using benefit transfer in cases where the policy case or the study case are focused on a "good" with unique attributes or where the magnitude of the change or improvement across the two cases differs substantially (OMB 2003). 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.

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 (e.g. Hunter and Schmidt 1990; Stanley 2001), so some otherwise suitable studies may be rejected as "outliers."

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.

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 and apply it directly to the policy case. 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 may be useful for developing preliminary, order-of-magnitude estimates of benefits, but it should be possible to base final benefit estimates on more 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 may yield inaccurate results.

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 by transferring the estimated function upon which the value estimate in the study case is based to the policy case. 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.

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.

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. If the choice of the functional form for the transfer function to be estimated is based mainly on statistical and qualitative economic considerations, then the meta-analysis could be considered non-structural. If the functional form were derived analytically from a structural model of household utility, then the meta-analysis would be considered structural. The advantage of non-structural transfer functions is that they are generally easier to estimate while controlling for a relatively large number of confounding variables. The advantages of structural transfer functions are that they can accommodate different types of economic value measures (e.g., WTP, WTA, consumer surplus) and can be constructed in such a way that certain theoretical consistency conditions (e.g., WTP bounded by income) can be satisfied. To date, most transfer functions estimated using meta-analysis have employed a non-structural approach (e.g., Poe et al. 2001, Shrestha and Loomis 2003a and 2003b, Rosenberger and Loomis 2000, and Bateman and Jones 2003). The few structural transfer functions that have been estimated to date have used very few study cases and a calibration approach to estimate the underlying preference parameters (e.g. Smith and Wilen 2003; Smith and Pattanayak 2002).

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) and Moher (1999) for more information. In general, in 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 US EPA (2006d) for a detailed discussion of meta-analysis as applied to value of statistical life (VSL) estimates.

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 also be quantified and reported when possible. (See Chapter 10 on presenting uncertainty).

7.5 Accommodating Non-monetised 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. This section discussed what analysts can do to incorporate these endpoints 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 10 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, may 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 benefit-cost analysis in their ability to identify an economically efficient policy. This and other short-comings should be discussed when presenting results from these analyses to decision-makers.

7.5.2.1 Break Even Analysis

Breakeven analysis is one alternative that can be used when either risk data or valuation data are lacking.[147] Analysts who have per unit estimates of economic value but lack risk estimates, cannot quantify net benefits. They may, 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.[148] For example, 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 willingness to pay 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 “breakeven.” This estimate can then be assessed for plausibility either quantitatively or qualitatively. Policy makers will need to determine if the breakeven value is acceptable or reasonable.

The same sort of analysis may be performed when analysts lack valuation estimates, producing a breakeven 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 willingness to pay to avoid a case of this endpoint. In this case, the policy can be considered to “breakeven” if willingness to pay is at least $200.

One way to assess the credibility of economic breakeven values is to compare them to risk values for effects that are more or less severe than the endpoint being evaluated. For the breakeven 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 willingness to pay to avoid a case of a more serious effect was only $100, the above “breakeven” point may not be considered plausible.

Breakeven 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

[147] Boardman et al. (1996) describes determining breakeven points under the general subject of sensitivity analysis and includes empirical examples.

[148] Circular A-4 (OMB 2003) refers to these values as “switch points” in its discussion of sensitivity analysis.

they will need to consider a “breakeven 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.

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. While often portrayed as being relatively straightforward – particularly compared to the estimation of benefits – cost estimation presents a number of analytical challenges.

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 benefit-cost analysis, social cost is the correct measure to use. Social cost represents the total burden that a regulation will impose on the economy, and is defined as the sum of all opportunity costs incurred as a result of the regulation. 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 the 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 a large number of sectors are expected to be impacted and the effects will be more broadly spread throughout the economy.

The third challenge is choosing one or more models to use in an analysis. Factors that may be considered 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.

In the next section, social cost and its underlying economic theory are discussed. In the third section, several alternative concepts of cost are presented. The fourth section discusses several additional issues in cost estimation. The final section presents a number of the models that may 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 benefit-cost analysis – is "social cost." Social cost represents the total burden a regulation will impose on the economy; it may 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.[149]

[149] This section discusses the prospective estimation of social cost, i.e. 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).

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.[150] 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 other broad measure of economic activity, that may result from its imposition. While 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 may be 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 will be more appropriate to use general equilibrium analysis to estimate social cost. The use of these two analytical paradigms is explored in the following sections.

8.1.1 Partial Equilibrium Analysis

When the effects of a regulation are expected to be confined primarily to a single market or a small number of markets, the estimation of social cost can be performed using partial equilibrium analysis. 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. In this section, some simple diagrams are presented 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 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.[151]

In this market, the imposition of a new environmental regulation raises firm's production costs. Each unit of output is now more costly to produce because of expenditures made 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

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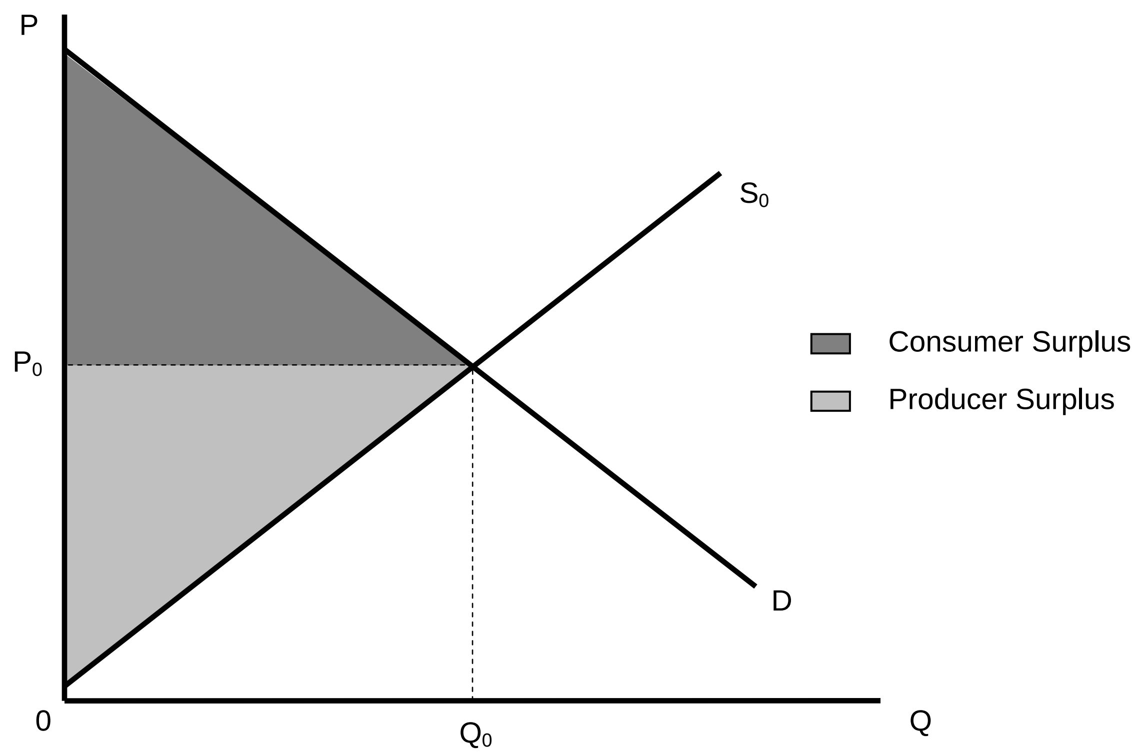
[150] For example, comparing the social cost of different regulations may provide some sense of the relative burden they impose on the economy, but this exercise alone would not indicate which, if any, of the regulations may be worthwhile from a public policy standpoint. However, the accurate measurement of social cost would be an essential component in attempting to make such a determination.

[151] It should be noted that total welfare as depicted ignores the negative pollution externality arising in this market that the environmental regulation is designed to correct. The appendix presents a graphical description of how to account for this externality, the reduction of which 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 the appendix.

equilibrium output to Q_1, holding all else constant. As can be seen by comparing Figures 8.1 and 8.2, the overall effect on welfare is a decline in both producer and consumer surplus.[152]

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.[153] 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, it can be seen that when consumers are price sensitive – as reflected in the downward sloping demand curve – a higher price causes them to reduce their consumption of the good. If costs are estimated *ex ante* and this price sensitive behavior is not taken into account – so that the estimate is based on the original level of output, Q_0 – compliance costs will be overstated. This can be seen by extending the vertical dotted line in Figure 8.2 from the original equilibrium to the new supply curve, S_1.[154]

Figure 8.1: Competitive Market Before Regulation

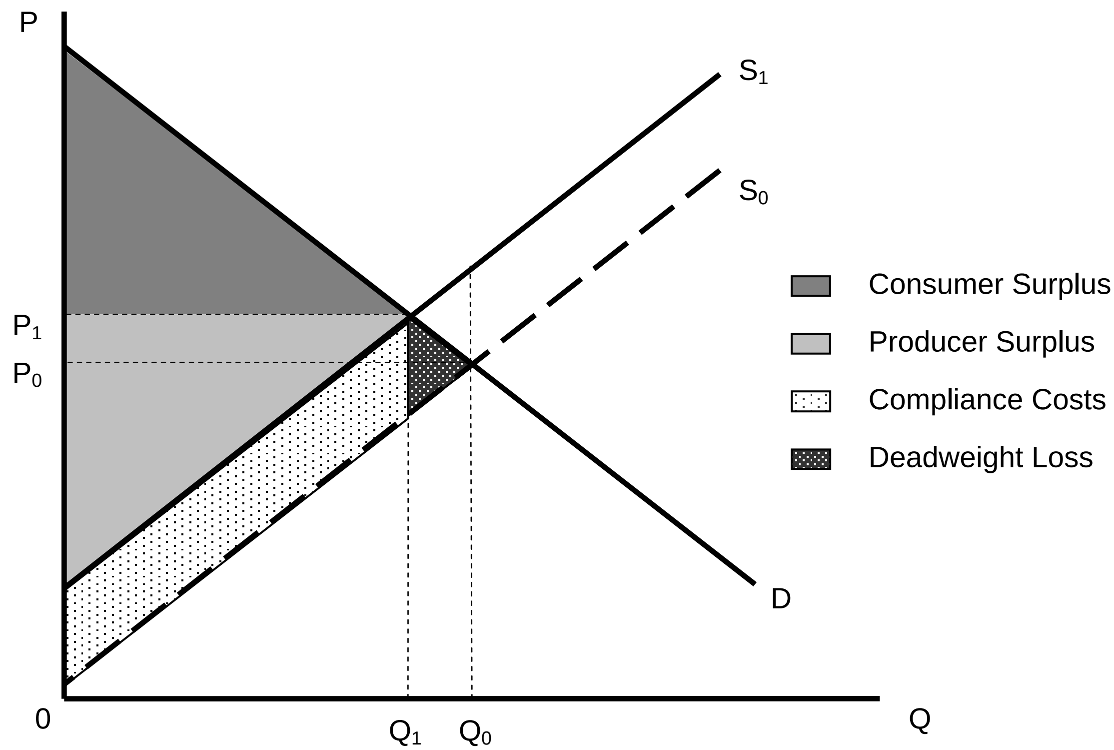


[152] The figure depicts an equal distribution of welfare between consumers and producers, in both the old and new equilibria, but this need not be the case depending on the elasticities of supply and demand. These elasticities will determine the magnitude of the price and quantity changes induced by the cost increase.

[153] Here we abstract from distinctions between the fixed and variable costs of abatement and assume that all of the costs are represented in the movement of the supply curve. See Tietenberg (2002).

[154] 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.

Figure 8.2: Competitive Market After Regulation



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" shown in Figure 8.2 is an additional, real cost arising from the regulation, reflecting the foregone net benefit due to the reduction in output.[155] While in reality at least part of the compliance cost will likely be spent on abatement-related purchases from other industries – and is thus not necessarily a loss to society – in this market, the deadweight loss resulting from the regulation is lost completely. Moreover, unlike many one-time compliance costs, it 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 benefit-cost analysis 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.

[155] Typically, in a market already distorted with pollution externalities, the deadweight loss triangle shown in Figure 8.2 will serve to offset (at least in part) the existing deadweight loss 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.

The preceding discussion describes the use of partial equilibrium analysis when the regulated 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.[156] 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 may still 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, such as 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. These 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 may be analyzed in a similar fashion.

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. Instead, 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 many 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, can also be calculated in the model.

The previous section showed how the social cost of a regulation can be estimated in a single market using partial equilibrium analysis. The regulation caused a deadweight loss in that market, reflecting a decline in economic welfare as measured by consumer and producer surplus. In reality, deadweight losses

[156] 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.

already exist 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 has caused 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 deadweight loss. 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 – with an upward sloping labor supply curve – the amount of labor supplied.[157] This is shown in the figure as a decrease in the net wage to W_1^n and the amount of labor supplied to L_1.

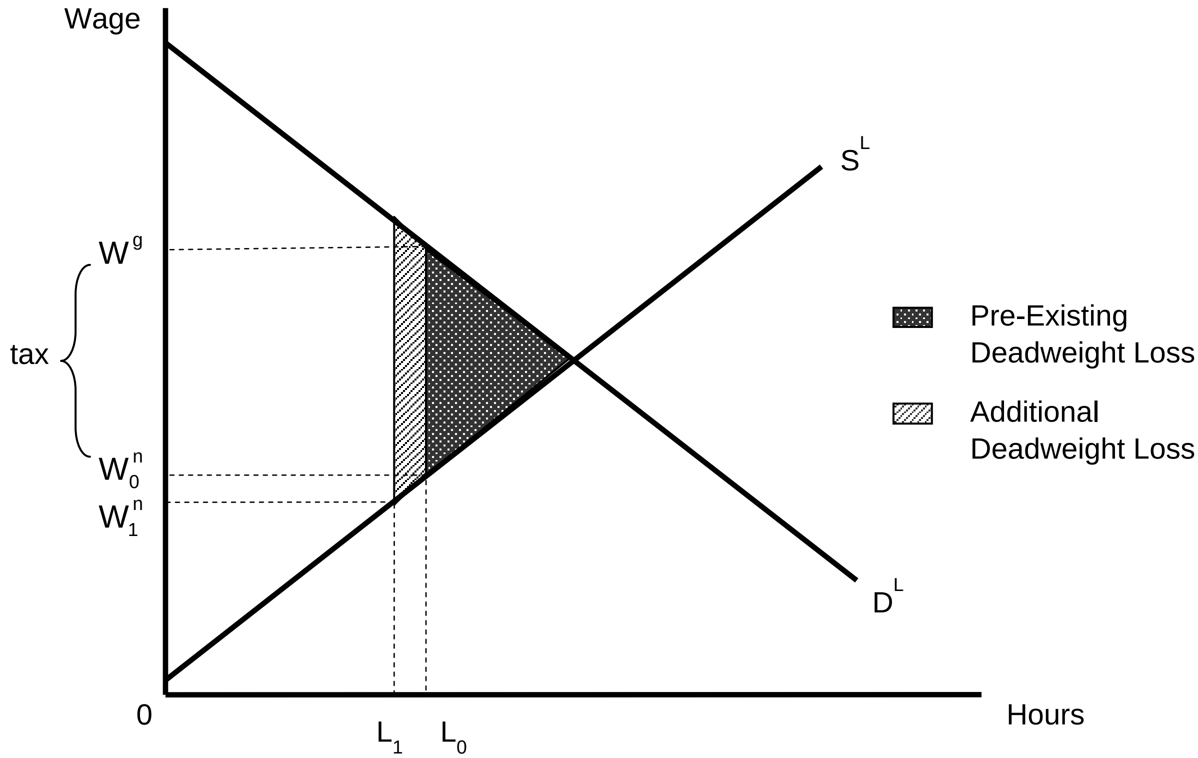
The interaction between new and pre-existing distortions is especially pronounced in the labor market because pre-existing distortions there are large. As shown in Figure 8.3, even a small reduction in the amount of labor supplied will result in a large increase in deadweight loss.[158] Similar interactions are likely to occur in other markets with pre-existing distortions. In cases where they are likely to have a significant impact, these distortions should be incorporated into models used to estimate social cost.[159]

[157] 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.

[158] 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).

[159] 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 Mooij (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).

Figure 8.3: Labor Market with Pre-Existing Distortions



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 may 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.[160]

Even in a general equilibrium analysis, care must be taken 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.[161]

[160] 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, 1997).

[161] Final demand consists of household purchases, investment, government spending, and net exports (exports minus imports).

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 and compensating variations. 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 equivalent and compensating variation 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.

When conducting a benefit-cost analysis in which the intertemporal effects of a regulation are expected 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, an estimation of social cost can be made 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.[162] 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.

8.2 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 most appropriate measure to use in benefit-cost analysis. In addition to social cost, a number of other concepts of cost exist and may be used to describe the effects of a regulation. This section discusses these alternative concepts and introduces a number of additional terms. Measures that define temporary costs or define how costs are distributed across different entities are also discussed.

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

[162] 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.

these concepts are comprehensive in nature. However, an important distinction is that while social cost is a measure derived from economic theory, these three concepts are, in general, only descriptive.[163]

Consideration of these alternative concepts may 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. In addition, several executive and legislative mandates require that a number of different types of costs be included in a Regulatory Impact Analysis (RIA).[164]

8.2.1.1 Explicit and Implicit Costs

The total costs of a regulation may include both explicit and implicit costs.[165] 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 would normally include the costs of purchase and operation of pollution control equipment. This would include 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 would be included in explicit costs. Implicit costs, on the other hand, are costs for which monetary values do not readily exist and which are thus likely to be 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 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 may 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.

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[163] In certain cases, a single component, such as direct cost, may provide a reasonable estimate of social cost.

[164] Executive Order 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 Unfunded Mandates Reform Act of 1995 requires that cost estimates take into account both indirect and implicit costs on state and local governments.

[165] 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.

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 care must be taken to avoid double counting.[166]

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.

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 and operating costs. Compliance costs may 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 benefit-cost analysis, capital costs are usually "annualized" over the period of the useful life of the equipment.

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[166] References which provide definitions of cost terminology include Congressional Budget Office (1988) and Callan and Thomas (1999).

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 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 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.[167]

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 benefit-cost analysis is by definition concerned only with

[167] Government entities may themselves be polluters and therefore subject to regulation. Compliance costs under this scenario would be captured as such.

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 benefit-cost analysis. Distributional costs are covered in detail in Chapter 9.

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: those that arise when estimating costs over time and 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 by their imposition. When measuring costs over time, assumptions related to the time horizon of the analysis, the use of a static vs. 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, great care must be taken to ensure that the year(s) selected are truly representative, that no important transitional costs have been effectively dismissed by assumption, and that no one-time costs have been 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 contrast, in the long term, 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, all of which 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.[168]

Discounting does have one application that is unique to cost analysis. In calculating firms' private costs, e.g. the internal cost of capital used for pollution abatement, analysts should use a discount rate that reflects the industry's cost of capital. The analyst should use these private costs in determining the responses of regulated entities, but discount social cost according to a social discount rate discussed in Chapter 6.

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).

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), it 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.[169] 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 over 100 studies including some dealing with the energy and manufacturing sectors. It should be noted 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).[170]

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[168] In a cost-effectiveness analysis, it is equally important to properly discount cost estimates of different regulatory approaches to facilitate valid comparisons.

[169] 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 that should be considered include those resulting from a shift to regulatory performance standards and incentive based policies.

[170] 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.

8.3.2 Difficulties in Valuing Social Cost

Difficulties in measuring social cost generally fall into two categories: difficulties in developing a numeric value for some social cost categories; and, 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 benefit-cost analysis. Some alternative techniques for measuring and presenting these effects to policymakers 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. Therefore, the conclusions drawn in the social cost analysis will be sensitive to the degree of uncertainty regarding the assumptions that were made. 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. Section 5.5 outlines a process for analyzing and presenting uncertainty.

Text Box 8.1 - The SO_2 Cap and Trade Program – A Case Study[171]

Under Title IV of the 1990 Clean Air Act Amendments (CAAA), coal fired power plants are required to hold one 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, uncertainties) can make to actual cost estimates.

Estimates of Title IV's compliance cots have declined over time, particularly 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; Burtraw and Palmer 2004). Furthermore Popp (2003) concluded that Title IV induced R&D led to technological innovations which 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 Ellerman et al., 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 (Burtraw and Palmer 2004, p. 54)."

[171] This example is taken from Burtraw and Palmer (2004).

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, while others may be complex systems of thousands of equations that require highly specialized software.[172]

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, while in other cases, the use of more than one model will be required. For example, a compliance cost model may be used to estimate the direct costs of a regulation in the affected sector. These direct cost estimates may 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.

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				✓	✓	✓
Can capture feedback and interaction effects						✓

Selecting the most appropriate model (or models) to use in an analysis can be difficult. However, a number of factors can be identified that may be helpful in making a choice:[173]

- **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.

[172] 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.

[173] The following draws on Industrial Economics, Inc. (2005). Proprietary models discussed in this section are examples only and no endorsement by EPA is given or implied.

- **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 the direct costs to an industry 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 are often 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 may be used as inputs into other models in estimating 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, VOC, PM_{10}, $PM_{2.5}$, NH_3) as well as CO and 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 in its operation.

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 can then be used to estimate the change in market price and output. Changes in producer and consumer surplus reflect the social cost of the regulation. In addition, the relative changes between producer and consumer surplus provides 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 either be estimated from industry and consumer data or taken from previous studies.[174]

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;

[174] Because of their widespread use, the Air Benefit and Cost (ABC) Group in EPA's Office of Air and Radiation maintains an elasticity database. 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. An online submittal form allows you to provide elasticity estimates for consideration as part of this databank. Available online at http://www.epa.gov/ttn/ecas/Elasticity.htm.

- 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.

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 may be employed in a variety of applications, their use in the analysis of EPA regulations occurs most frequently in the estimation of compliance costs.[175] 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 also 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

[175] 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.)

electrical generating units are mapped to approximately 1,700 representative plants and the 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 mercury.

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 and this permits an existing model to be used in a range of applications.

Limitations:

- Linear programming models do not normally estimate costs beyond a single sector and are thus unable to estimate indirect or distributional costs.
- 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.[176]

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. 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.

[176] Miller and Blair (1985) is a standard reference on input-output analysis.

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 10-sector table.)

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 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.[177] 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.

[177] An off-the-shelf input-output model often used in the analysis of the impacts of environmental regulation is IMPLAN. IMPLAN is based on data for the United States which covers over 500 sectors and can be disaggregated down to the county level.

- 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 relationships that more accurately account for these conditions. However, unlike standard macroeconometric 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, making 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 REMI Policy Insight.[178] The standard REMI model includes 70 production sectors and 25 final demand sectors and can provide output on changes in income and consumption for over 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.

[178] REMI stands for Regional Economic Models, Inc.

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 (CGE) 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, the impacts of domestic and international policies for greenhouse gas 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 may 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. In some models, agents may also 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 then 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 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.

An example of the use of a CGE model at EPA is the retrospective benefit-cost analysis of the Clean Air Act (CAA), which used a dynamic CGE model to compute the costs of CAA compliance over the period 1970 to 1990 (US 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* these long-term capital and operating costs from the industries that incurred them. A comparison was then made between the simulated path of the economy without these abatement expenditures and the actual path of the economy, which included them. Changes in both long-run GDP and equivalent variation were computed, as well as impacts on investment, household consumption, and sectoral prices, output, and employment.

Text Box 8.2 - The Pollution Abatement Costs and Expenditures (PACE) 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 the 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, except 1999. 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.

CGE models have also been used extensively in estimating the costs of greenhouse gas 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 greenhouse gases 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.

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 not appropriate for analyzing short-run transitional costs.
- 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.

9 Distributional Analyses: Economic Impact Analyses and Equity Assessment

The detailed study of regulatory consequences allows policymakers 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: first, the regulation's efficiency, and second, its distributional effects. In principle, both of these consequences could be estimated simultaneously by a general equilibrium model. In practice, however, they are usually estimated separately, for the reasons discussed in Section 1.3.

This chapter discusses how the distributional effects of environmental regulations can be examined through economic impact analysis (EIA) and equity assessment. An EIA focuses on traditional classifications of affected entities, such as industrial sector classifications.[179] In contrast, an equity assessment addresses the distribution of impacts across subpopulations, and may also address broader concerns such as changes in the national distribution of income or wealth. Disadvantaged or vulnerable groups (e.g., small businesses, low income households, racial or ethnic minorities, and young children) may be of particular concern. Together, EIAs and equity assessments are referred to as distributional analyses.

9.1 Introduction to Economic Impact Analyses

An EIA identifies the specific groups that benefit from or are harmed by a policy, and then estimates the magnitude of their gains and losses. These estimates are derived from a study of the economic changes that occur across broadly defined economic sectors of society, including industry, government, not-for-profit organizations, and consumers. An EIA also examines more narrowly defined sectors within these broad categories, such as the solid waste industry or even an individual solid waste company. Therefore, EIAs may measure a broad variety of impacts, such as direct impacts on private business - including individual plants, whole firms, and industrial sectors - and indirect impacts on consumers and suppliers. The term "impacts" includes changes in profitability, employment, prices, government revenues or expenditures, and trade balances.

For any regulation, it is essential to ensure consistency between the EIA and the Benefit Cost Analysis. If a benefit-cost analysis is conducted, the corresponding EIA must be conducted within the same set of analytical bounds. To the extent possible, the EIA should adopt the same set of assumptions used by the BCA. 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.

[179] The term "affected" is used throughout this chapter as a general economic 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, 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 Unfunded Mandates Reform Act (UMRA) and Executive Order 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. See Chapter 8 for more details.

9.2 Introduction to Equity Assessments

An equity assessment examines the distribution of a regulation's costs, benefits, and other economic impacts on specific sub-populations (for an example, see Shadbegian, et al. 2005). An equity assessment may also analyze a regulation's impact on the distribution of national income or wealth.

An equity assessment is generally more concerned with sub-populations that experience net costs or other negative impacts than with those that experience net benefits or positive impacts. Whereas an EIA focuses on traditional classifications of affected entities (e.g., industrial classifications), an equity assessment often focuses on disadvantaged or vulnerable sub-populations, such as low-income households or children. An equity assessment may also focus on the average income of households or other relevant sub-groups (e.g., upper income households).

While an equity assessment may consider a regulation's effects on any sub-population and often starts with an evaluation of the impacts across all groups, it should always consider the effects on disadvantaged or vulnerable groups. Specifically, an equity assessment should examine sub-populations that are physically susceptible to environmental contamination, are less than fully capable of representing their own interests, and/or are economically disadvantaged or vulnerable. Groups such as children, low-income, or minority or ethnic populations, or populations with limited English proficiency, and small businesses, small governments, and small not-for-profit organizations are often included in equity assessments. EPA is frequently required by statute or policy to examine the effects of a rule on one or more of these groups when they are expected to experience a disproportionate, significant, and substantial impact. Finally, as with an EIA, an equity assessment should adopt the same assumptions included in the BCA.

9.3 Statutes and Policies

The following major statutes and executive orders directly concern distributional issues:

- 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);
- E.O. 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations";
- E.O. 13045, "Protection of Children From Environmental Health Risks and Safety Risks";
- E.O. 13132, "Federalism"; and
- E.O. 13175, "Consultation and Coordination with Indian Tribal Governments"; and
- E.O. 13211, "Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution, or Use."

Two additional executive orders ask agencies to consider distributional effects under special circumstances. E.O. 12866, "Regulatory Planning and Review," has multiple objectives, including a specific directive for agencies to consider distributive impacts and equity when designing regulations.[180] E.O. 13211, "Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution, or

[180] EPA's Regulatory Management Division's Action Development Process Library (http://intranet.epa.gov/adplibrary/) is a resource for those who wish to access relevant statutes, executive orders, or Agency policy and guidance documents in their entirety. Accessed 7/14/2004.

Use," requires agencies to provide a statement of energy effects for any significant energy action where adverse impacts on energy supply and prices are expected. Finally, OMB Circular A-4 provides guidance on preparing regulatory impact analyses, including a discussion of distributional effects (OMB 2003).[181]

Table 9.1 lists dimensions that may be relevant in an EIA or equity assessment and links each dimension to a statute, order, or directive; population or entity; and sub-population.

Table 9.1 - Potentially Relevant Dimensions to Distributional Analyses[182]

Dimension	Statute, Order, or Directive	Population or Entity	Sub-Population
Sector	UMRA; E.O. 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
Minority status, income	E.O. 12898; OMB Circular A-4; E.O. 13175	Individuals or households	Racial or ethnic populations, low-income populations, Tribal populations
Age	E.O. 13045; OMB Circular A-4	Individuals or households	Children or elderly
Gender	OMB Circular A-4	Individuals	Male or female
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	E.O. 13211	Entities that use, distribute, or generate energy	Energy sector

9.4 Chapter Summary

The remainder of this chapter is organized as follows. Sections 9.5 and 9.7 discuss economic impact analyses and equity assessments, respectively, including details regarding the components of each analysis, screening tools, data sources, and relevant dimensions to consider. Sections 9.6 and 9.8 discuss potential modeling approaches and frameworks for each type of analysis.

It is important to note that the analyses described in this chapter could potentially be incorporated into a single social welfare function; however, such an approach is rigorous, and often infeasible due to lack of data or other limitations. Text Box 9.1 discusses this approach in more detail.

[181] See Chapter 2 for a brief description of these statutes and executive orders.

[182] Some environmental statutes may also identify sub-populations that merit additional consideration. This document is limited to those statutes with broad coverage.

Text Box 9.1 - Using a Social Welfare Function to Evaluate Efficiency-Equity Tradeoffs

It is possible to combine the efficiency and distributional effects of a regulation into a single social welfare function. A social welfare function establishes criteria under which efficiency and equity choices are transformed into a single metric, making them directly comparable. The 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, is better or worse than a less efficient alternative that exhibits more equitable distributional consequences. See Sen (1970) and Arrow (1977) for a theoretical discussion of social welfare functions, and Norland and Ninassi (1998) for an application to energy markets.

In practice, a social welfare function requires explicit decisions about society's preferences regarding the distribution of resources. As a consequence, no functions exist that are universally or even widely accepted. As such, their use to evaluate efficiency-equity tradeoffs remains controversial. Nonetheless, future research may result in a set of feasible and practical options for social welfare functions (see Farrow (1998) for a description of potential alternatives). These guidelines do not suggest a particular social welfare function, or even recommend that analysts attempt to use such a function, but the approach merits further consideration as additional research and applications are developed.

9.5 Conducting an Economic Impact Analysis

There are three important issues to consider when conducting an EIA.[183] First, total cost is no longer of primary importance in an EIA. Rather, the main focus is on the components and distribution of the total social cost.

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 costs of a regulation, an EIA focuses on the private costs associated with compliance responses. The same basic engineering or direct compliance cost estimates may be used as a starting point for developing both social and private cost estimates, and are adjusted according to their current purpose.[184]

Finally, there is a greater need for disaggregation in economic impact analyses than in benefit-cost analyses. Results may be presented for specific counties or other geographic units, specific demographic groups, or types of entities, as appropriate, placing heavy demands on the modeling framework.

9.5.1 Screening for Potentially Significant Impacts

A comprehensive analysis of all aspects of economic costs 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 may be thought of as a "mini-EIA" consisting of a rough examination of the data to identify sectors that may warrant further analysis.[185] Screening is

[183] 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 sub-populations. While this chapter discusses costs only, EIAs can also focus on benefits.

[184] For example, the tax status of a required piece of equipment is considered in private costs, but not in social costs.

[185] 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 9.4.1.1).

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, 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 weeds out unfounded impacts. In this way, the screening analysis will eventually balance the risk of identifying "false positives" and "false negatives."

9.5.2 Profile of Affected Industry 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.

9.5.2.1 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.[186] This includes an understanding of the condition of the industry in terms of its finances and structure in the absence of the rule, also called a baseline. 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 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.[187]
- **Industry summary statistics.** Summary statistics of total employment, revenue, number of establishments, number of firms, and size of firms are available from U.S. Department of Commerce Economic Census or the Small Business Administration.[188]
- **Baseline industry structure.** Industry-level impacts depend on the competitive structure and organization of the industry and the industry's relationship to other economic entities. In addition, the number and size distribution of firms/facilities and the degree of vertical integration

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[186] 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.

[187] For more information see www.census.gov/epcd/www/naics.html, which includes a NAICS/SIC correspondence.

[188] See www.sba.gov/advo/stats/data.html for more information.

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 9.2.3 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.

9.5.2.2 Profile of Government Entities, Not-for-profit Organizations, and Households

Analysts should carefully consider whether a particular rule will directly affect government entities, not-for-profit organizations, or households. [189] 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; and 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;
- 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.

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. Depending on

[189] Government entities that may be affected include states, cities, counties, towns, 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.

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.[190]

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/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.

Relevant features of *households* include standard socioeconomic and demographic characteristics, such as:

- Income level;
- Household size;
- Number of children and elderly;
- Education level; and
- Ethnic composition.

Section 9.2.4 discusses the unique characteristics that should be considered when estimating impacts on government and not-for-profit entities. Section 9.4 discusses households.

9.5.2.3 Data Sources for Profiles

Profiles generally rely on information from the following sources: economic journals, working papers, dissertations, and industry trade publications as well as quantitative data describing the characteristics of the industry.[191] Relevant literature can be useful in characterizing industry activities and markets as well as regulations that already affect the industry and 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 over 800 databases covering business, economic, and scientific topic areas. Table 9.2 describes some commonly used data sources for retrieving quantitative data.

The industry profile may also identify situations where insufficient data are available through standard sources. These situations often arise when the affected industry is one of many product lines or activities of the identified facilities. In addition, for some industries, identification of the appropriate NAICS code for all the firms or facilities included in the industry may be difficult if the industry can be categorized in multiple ways. In these cases, and particularly if facility-level data are required to estimate economic

[190] 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).

[191] Academic literature may or may not contain quantitative data.

impacts, a survey of either a statistical sample or a census of affected facilities may be required to provide sufficient data for analysis.

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/econ/census02/)	Sales, receipts, value of shipments, payroll, number of employees, number of establishments, value added, cost of materials, capital expenditures by sector
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
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

9.5.3 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.

9.5.3.1 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. In addition, 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.[192] At the other extreme, firms may be able to raise prices enough to recover costs fully (depending on the elasticity of demand). In this case, there is no impact on the profitability of the directly-affected firms but their customers bear the burden of increased prices.

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. The methods used to conduct the analysis of the directly-affected markets depends on the availability of appropriate estimates of supply and demand elasticities.[193] 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. For example, an examination of the number of firms, quantity of a product produced, and industry size will provide basic information about supply and demand.

9.5.3.2 Impacts on Production and Employment

As noted above, regulations may sufficiently increase the cost of doing business to make some or all production unprofitable, or may reduce the quantity of product demanded as producers raise their prices to maintain profitability. The associated reductions in output may result from lower operating rates at existing plants, closure of some plants, or reduced future growth in production relative to what would have occurred in the absence of regulation. Losses in employment are typically associated with these reductions. However, regulations may cause a rise in output and employment in other sectors. When determining the distribution of impacts of a regulation, both contractions and expansions should be noted.

EPA uses a variety of methods to assess changes in production and employment. In some cases, demand and supply elasticities are used directly to calculate changes in output and prices that would result from a supply curve shift associated with new compliance costs. Because the shape of the supply curve is often not known, assumptions are made about its shape in the relevant region. General or partial equilibrium models can also be used, as discussed in Chapter 8.

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.[194]

Therefore, total employment and production may not decline, but instead shift from higher cost plants to more efficient competitors. Where appropriate, such changes in output and employment are addressed in

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[192] For example, regulations limiting sulfur 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.

[193] See Appendix A for a more complete discussion of elasticity.

[194] 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.

the EIA. This is especially the case for rules that may have a strong regional impact.[195] Data on the ratio of production or sales to employment can help predict the number of jobs lost as a result of reductions in production. The regional distribution of job losses (and gains) can be calculated based on plant locations, if they are known.

9.5.3.3 Impacts on Profitability and Plant Closures

The availability of financial information used to assess profitability varies greatly, depending on the industry in question and the extent to which EPA is able to collect new information by surveying affected entities. With limited exceptions, detailed financial information is generally unavailable from published sources for individual plants or for privately-held companies. Financial data for publicly-held companies may be too aggregated to allow analysis of the specific business practices affected by the rule. In the absence of a new data collection by EPA, analysts may need to rely on financial profiles constructed for model plants, or on industry-average data provided by the Census Bureau and other sources.[196] In some cases, financial profiles used in the analysis of a previous rule-making might be adapted and updated to analyze the impacts of the rule in question.

The severity of financial impacts to firms from a rule can range from no impact (if all costs are recovered through price increases, for example) to a modest reduction in profits (closure of a production line or plant) to bankruptcy of the firm. Criteria for assessing the degree of financial distress and for predicting when a production line or plant would be shut down are not clear-cut.[197] If detailed financial profiles can be developed, including revenues, costs, income statements, and balance sheets, a variety of financial tests can be used to assess the likelihood of financial distress or closure. These tests address the following issues:

- Do the costs of the regulation result in a negative discounted after-tax cash flow?198
- Does the facility or firm’s profitability fall below acceptable levels (as measured by some industry standard)?
- Is the facility’s or firm’s ability to finance its operations and pay its obligations jeopardized?

Plants or firms that fail these tests are potentially at risk for closure.[199] A variety of considerations affect a firm’s decision to close a production line or a plant. These include the following:[200]

[195] UMRA Section 202 requires regional impact analysis as an element of the UMRA cost analysis when “accurate estimates are reasonably feasible.”

[196] Some sources of financial data are listed in Table 9.2.

[197] As stated above, most analyses assume a perfectly competitive market, which in practice may not correctly characterize the market structure. In these cases, analyses should be adapted to the relevant market structure.

[198] If after-tax cash flow is negative under baseline conditions (before considering compliance costs), the facility is a likely candidate for closure even in the absence of additional compliance costs. These closures should not be attributed to the rule, but rather should be classified as baseline closures.

[199] If it is possible to estimate plant liquidation values, another test can be added under the assumption that plants may close if the value of continuing to operate is less than the liquidation value.

[200] Micro-economic principles dictate shutdown when price is less than marginal cost at the point where marginal cost is increasing. It is unlikely that analysts will have the data necessary to conduct this type of analysis, but it is important to note.

- **The profitability of the plant** itself provides insight into whether the operation will be continued if the plant represents a stand-alone business. This also assumes that it is possible to construct a financial profile of that business.
- **The role the plant plays in a larger operation** may influence closure decisions. For example, some plants may be part of a vertically or horizontally integrated operation. Such plants might not be viable as stand-alone operations but may continue to operate based on their contribution to the business line as a whole. It is therefore important to analyze impacts on the firm as a whole and not just on individual plants.
- **A negative discounted cash flow** indicates that returns are below the rate required to provide an adequate return on equity and payment of interest. Closures in the short-run are likely to occur if earnings do not cover variable costs plus the amortized cost of compliance. Divestment and closures occur over the longer term if earnings are not sufficient to justify investment in the plant and equipment.

Where closures and reduced production are likely for some but not all plants, firms may face complex decisions about which plants to close. These decisions reflect relative operating costs, age of equipment, tax and other incentives offered by local communities and states to retain business, and logistical considerations. It is important to note that analyses of plant closures identify candidates for closure, rather than provide reliable predictions of which specific plants will close. The available information on plant-level operating costs and contributions to earnings is generally too uncertain to allow a more precise prediction of plant closures.

Short of closure, financial distress may occur. Financial distress measures a continuum from mild to severe financial weakness, and may result in difficulty obtaining financing and attracting capital.[201] Although in practice analysts may use a variety of measures of financial distress, using financial ratios has the advantage that it mirrors analyses of investment and lending institutions. Particular measures include impacts on profitability (such as return on equity or assets) and measure of liquidity (such as interest coverage ratio).[202]

9.5.3.4 Impacts on Related Industries and Households

The economic and financial impacts of regulatory actions spread to industries and communities that are linked to the regulated industries, resulting in indirect business impacts. These indirect impacts may include employment and income losses, as well as changes in the competitiveness and efficiency of related markets. Compliance-related industries may also yield offsetting gains in employment and income when a regulated industry purchases equipment, facilities, or labor to comply with a regulation.

Although in principle every economic entity can be thought of as having a connection with every other entity, practical considerations usually require an analysis of indirect impacts for a manageable subset of

[201] Researchers have developed various composite measures that are designed to assess the potential for bankruptcy. The most commonly cited is the ZETA model or "Z-score" developed by Altman (1993). This model uses a weighted average of five variables to predict potential for bankruptcy. The ratios include working capital/total assets, retained earnings/total assets, earnings before interest and taxes (EBIT)/total assets, market value of equity/par value of debt, and sales/total assets. The model includes levels for this composite score that represent clear potential for bankruptcy, low or no potential for bankruptcy, and an uncertain grey area.

[202] The interest coverage ratio is a ratio of earnings before interest and taxes (EBIT) divided by interest. It measures the number of times a company can make its interest payments with EBIT. The lower the ratio the higher the debt burden of the company.

economic entities that are most strongly linked to the regulated entity. In addition to considering major customers and specialized suppliers of the affected industry, it is also important to consider less obvious but potentially significant links, such as basic suppliers like electricity generators.

Features of households relevant to economic impacts include standard socioeconomic and demographic characteristics such as income and education level, household size and age distribution. These factors play a role in determining the ultimate effect of changes in employment and income generated from a regulatory action.

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.[203]

9.5.3.5 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 gross domestic product, 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.

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 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.

9.5.3.6 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.[204] 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.[205] 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.

[203] Approaches to measuring these effects are discussed in Section 9.3.

[204] See Jaffe, et al. (1995) for an overview.

[205] Sunk costs are sometimes referred to as exit barriers.

- **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.

9.5.3.7 Impacts on Energy Supply, Distribution, or Use

Executive Order 13211 requires agencies to prepare a Statement of Energy for "significant energy actions,"[206] which are defined as significant regulatory actions (under Executive Order 12866) that also are "likely to have a significant adverse effect on the supply, distribution, or use of energy." These significant adverse effects are defined as

1. Reductions in crude oil supply in excess of 10,000 barrels per day;
2. Reductions in fuel production in excess of 4,000 barrels per day;
3. Reductions in coal production in excess of 5 million tons per year;
4. Reductions in natural gas production in excess of 25 million mcf per year;
5. Reductions in electricity production in excess of 1 billion kilowatt-hours per year or in excess of 500 megawatts of installed capacity;
6. Increases in energy use required by the regulatory action that exceed any of the thresholds above;
7. Increases in the cost of energy production in excess of one percent;
8. Increases in the cost of energy distribution in excess of one percent; or
9. 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.

9.5.4 Detailing Impacts on Governments and Not-for-Profit Organizations

Section 9.2.3 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?

[206] See section 2.1.6 for EPA and OMB's guidance on EO 13211.

- To what extent does the regulation increase operating costs?
- To what extend 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 may affect governments and not-for-profit organizations in at least three significant ways. First, they may directly impose requirements on the entity, such as water pollution requirements for publicly-owned wastewater treatment works or air pollution restrictions that affect municipal bus systems or power plants. Second, they may impose costs on government agencies associated with implementing and enforcing regulations. Finally, they may impose indirect costs, such as increased unemployment (and thus less tax revenues) in a community because a regulation has resulted in reduced production (or even closure) at a plant in the community.

9.5.4.1 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 various 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 the rule; all involve calculating the annualized costs of complying with the regulation. For *government entities*, the three approaches are:

- **Annualized compliance costs as a percentage of annual costs for the 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 one 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).[207] 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, however, some or all of the costs are allocated to users (e.g., fares paid by bus riders or fees paid by users for sewer, water, or electricity supplied by municipal utilities), then this needs to be taken into account and a more narrow measure may be appropriate. In addition, if some of the costs are borne by local firms, then that portion of the costs should be handled separately.

There are two commonly used impact measures for *not-for-profit entities*: annualized compliance costs as a percentage of annual operating costs, and 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 (see Table 9.3), bond ratings evaluate a community's credit capacity and thus reflect the current financial conditions of the government body.[208] 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.209 See Table 9.3 for interpretations of the values for these measures.

[207] 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 over 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.

[208] The indicators and benchmark values in Table 9.3 are drawn from a document, "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 1997). These are general benchmarks that may prove useful in assessing financial stability in an EIA.

[209] 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.

- **Neither of these two debt measures is 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. Also, if the government entity does not rely on property taxes, as may be the case for a state government or an enterprise district, then the ratio of debt to full market value 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. Two commonly used measures are the unemployment rate and median household income. Unemployment rates are available from the Bureau of Labor Statistics. Median household income is available from the U.S. Census Bureau; some states maintain more up-to-date databases on income levels. 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 revenues as a percentage of the full market value of taxable property indicates the burden that property taxes place on the community. 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.

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.

Measuring the financial strength of *not-for-profit* entities includes assessing:

- How many reserves the entity has;
- 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.

9.5.4.2 Administrative, Enforcement, and Monitoring Burdens on Governments

Many EPA programs require effort on the part of 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.

9.5.4.3 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 familiar multiplier effect, in which loss of income in one household results in less spending by that household and therefore less income in households and firms associated with goods previously purchased by the first household.

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.

9.6 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. As noted above, the analytic methods used for the distributional analysis of a particular regulation should be consistent with those used for the corresponding BCA.

9.6.1 Direct Compliance Costs

The simplest approach to measuring the distribution of 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 entities. 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, but 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 checked to determine if all elements are included and the costs are consistent within a given base year. To ensure consistency across years, either a general inflation factor, such as the Gross Domestic Product (GDP) implicit price deflator, or various cost indices specific to the type of project should be used.[210] 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 (e.g., programs involving recycling of household wastes). The cost of acquiring a permit is not so much the permit fee as it is the lost opportunities during the approval process. Likewise, the cost of having a car's emissions inspected is not so much the fee as it is the value of registrants' time.

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.[211] 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.[212]
- **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.213 In contrast to social costs and benefits, which are discounted using a social discount rate, private

[210] The GDP implicit price deflator is reported by the U.S. Department of Commerce, Bureau of Economic Analysis in its *Survey of Current Business* (www.bea.gov/bea/pubs.htm). The annual *Economic Report of the President*, Executive Office of the President, is another convenient source for the GDP deflator (www.gpoaccess.gov/eop/).

[211] Current federal and state income tax rates can be obtained from the Federation of Tax Administrators, *State Tax Rates & Structure*, available from http://www.taxadmin.org/fta/rate/default.html. Accessed 5/12/2004.

[212] These transfers cancel out in a BCA. In an EIA the distribution of results is important, therefore the transfers are included.

[213] The present value of costs may then be annualized to provide an annual equivalent of the uneven compliance cost stream. Annualized costs are also discussed in Chapter 6.

costs are discounted using a rate that reflects the regulated entity's cost of capital.214 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.215 For firms, the cost of capital may also be determined by their ability to deduct debt from their tax liability.

- **Annualized Costs.** Annualizing costs involves calculating the annualized equivalent of the stream of cash flows associated with compliance 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.216 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.

9.6.2 Partial Equilibrium Models

A partial equilibrium framework limits a distributional analysis to impacts on entities associated with a few directly and indirectly affected output markets only. 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 distributional outcomes, then it may be possible to adopt further assumptions and acquire additional data to approximate distributional consequences of concern. This may include deriving ratios to aggregate changes in order to distribute 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 computable general equilibrium models). These models are best used when potential distributional effects on related markets might be considerable, but more complete modeling using a computable general equilibrium (CGE) framework may not be available or

[214] 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.

[215] Risk adjusted rates for different industries can be obtained from Ibbotson Associates (2004).

[216] Economies of scale characterize costs that decline on a per unit basis as the scale of an operation increases.

practical. Partial equilibrium models may also be more appropriate for regionally-based or resource specific regulations which are too specific for more aggregated CGE models.[217]

9.6.3 Computable General Equilibrium Models

CGE models are particularly effective in assessing resource allocation and welfare distribution 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. By design, the basic capacity to describe and evaluate these sorts of distributional 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 may 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 may be disaggregated into dozens or even hundreds of sectors, each producing a unique commodity. The government, in addition to implementing a variety of taxes and other policy instruments, may provide a public good or run a deficit. CGE models may be international in scope, consisting of many countries or regions linked by international flows of goods and capital. The behavioral equations that characterize economic decisions may take on simple or complex functional forms. The model may be solved dynamically over a long time horizon, incorporating inter-temporal decision-making on the part of consumers or firms. 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 distribution analysis 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.[218]

9.7 Conducting an Equity Assessment

This section discusses various subpopulations or other groups that may be disproportionately affected by the benefits or costs of a policy and addresses how to consider these groups in an analysis. Note that the term “subpopulations” refers to portions of the population, such as small businesses or socially or economically disadvantaged people, as well as populations at different ages, such as children or elderly.

9.7.1 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

[217] See the discussion of multi-market modeling in Chapter 8 and Just, Hueth, and Schmitz (1982).

[218] For a discussion of CGE analysis see Chapter 8 and Dixon, et al. (1992).

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.

9.7.1.1 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.[219] 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. The impacts on small businesses are generally assessed by estimating their 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 discussed in section 9.2.3 (specifically in relation to small businesses) may provide additional information for decision-makers.[220]

9.7.1.2 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 if required, the 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 factors indicating 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

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[219] The current version of SBA's size standards can be found at http://www.sba.gov/size. Accessed 5/18/2004.

[220] See Agency guidance (U.S. EPA 1999) for details on complying with the RFA.

consultants for implementation, or requirements to purchase and operate expensive or sophisticated equipment.[221] See Section 9.2.4 for information on measures of impacts to governments in general.

9.7.1.3 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 9.2.4 for more information on measuring impacts on not-for-profit organizations in general.

9.7.2 Socially and Economically Disadvantaged Populations

The impact of a rule or regulation on the human health or the environment of socially or economically disadvantaged groups (including racial or ethnic minorities or low-income populations) raises issues of environmental justice.

Executive Order 12898 "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" and its accompanying memorandum[222] have the primary purpose of ensuring that "each Federal agency ...make[s] achieving environmental justice part of its mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low-income populations …" [223] The Executive Order also applies equally to Native American programs.[224]

For current information on EPA's environmental justice guidance, guidelines, policies, practices and other resources, see EPA's Office of Environmental Justice homepage: http://www.epa.gov/compliance/environmentaljustice/index.html.

There are many different types of impacts that may be considered when examining the effects of a regulation, and many factors that can contribute to these impacts. Among the factors to be considered are (1) proximity and exposure to environmental hazards;[225] (2) the existence of susceptible populations, [226]

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[221] Guidance on complying with Section 203 of UMRA, "Interim Small Government Agency Plan," is available on EPA's intranet site, "Action Development Process Library" at http://intranet.epa.gov/adplibrary/statutes.htm. Accessed 6/3/04.

[222] Memorandum, Executive Order on Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations, http://www.epa.gov/compliance/resources/policies/ej/clinton_memo_12898.pdf (accessed June 24, 2008).

[223] Executive Order 12898, Executive Order on Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations, http://www.epa.gov/compliance/resources/policies/ej/exec_order_12898.pdf (accessed June 24, 2008)

[224] Id. at Sec. 6-606. In addition, E.O. 13084, *Consultation and Coordination with Indian Tribal Governments*, requires regulations that "significantly or uniquely" affect the communities of Indian tribal governments and that impose substantial direct compliance costs on such communities to either refund the direct costs incurred, or consult with elected officials and other representatives of the Indian tribal governments and provide a description of the consultation and/or communication to the Office of Management and Budget.

[225] Adverse human health and environmental impacts may occur from proximity and exposure in lead, air pollution, including toxic air emissions, groundwater contamination; mining waste; uncontrolled leachate or stormwater

(3) unique exposure pathways,[227] (4) multiple and cumulative effects;[228] (5) inability to participate in the decision-making process;[229] and (6) vulnerable infrastructure.[230] A disproportionately high and adverse human health or environmental effect may result from a combination of several of the above factors. In some circumstances, even one or two of these factors could, in and of themselves, disproportionately expose a population to environmental harms and risks.[231]

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Many studies exist in the academic literature that attempt to identify and characterize human health or environmental inequities. The approach to be used in analyzing whether a disproportionately high and adverse human health or environmental impacts currently exist, or could be impacted by a proposed action, depends upon the circumstances (see Textbox 2 for examples of approaches that may be useful, although not exclusively.) E.O. 12898 does not identify particular approaches that could be used for the identification of potentially disproportionate impacts. To properly screen and conduct more targeted analyses to evaluate such impacts, the analyst may consider a number of methodological issues, including:

- Measurement issues (e.g., how to properly define the impacted neighborhood or sub-population; the threshold for a disproportionate impact; and the proper comparison population);

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runoff from municipal landfills or abandoned toxic waste sites; vehicle emissions from adjacent transportation thoroughfares and ports; agricultural chemicals and pesticides; contaminated fish and shellfish; and off-site migration of hazardous wastes from Superfund sites.

[226] Certain conditions could render different groups less able to resist, or tolerate, an environmental stressor. These susceptibility factors may be intrinsic to the group, based on age, sex or other factors. Or, they may be acquired, such as chronic medical conditions, health-care access, nutrition, fitness, other pollutant exposures, and drug and alcohol use. An evaluation of the susceptibility of a subpopulation should include an assessment of pre-existing health conditions, lack of access to health care, psychosocial stress, and/or lack of social capital

[227] Some communities sustain unique environmental exposures because of practices linked to their cultural background or socioeconomic status. For example, some indigenous peoples and immigrant populations rely on subsistence fishing. This subsistence fishing diet could present a different exposure scenario to consider when proposing regulations. Alternatively, economic deprivation, rather than cultural factors, may result in a subsistence diet or other pathways for increased exposure. For examples, pica is a habit among malnourished young children of eating dirt or paint chips, thus increasing exposures to lead.

[228] Disadvantaged and vulnerable populations may suffer a wide range of environmental burdens. The chemical-specific focus to assessing environmental risks may not always account for the multitude of contaminants.

[229] Conditions which could contribute to the ability of a community to participate fully in the decision-making process include, among others, (1) lack of trust; (2) lack of information; (3) language barriers; (4) socio-cultural issues; (5) inability to access traditional communication and information exchange channels; and (6) limited capacity to access scientific, technical, and legal resources.

[230] The physical infrastructure in a community, such as poor housing or poorly maintained public buildings (e.g. schools), is a significant factor that may contribute to make a community more vulnerable to environmental hazards.

[231] Likely to be of particular interest or concern are exposures to toxins and adverse health impacts, such as asthma and blood lead poisoning. Impacts could also be cultural, such as degraded ecosystems that prevent people from engaging in subsistence hunting and fishing. It is important to identify the features of the population of concern that may be adversely impacted. For example, if individuals live in primarily urban areas then they may be adversely exposed to air pollution and hence disproportionately suffer adverse health impacts

- How to determine the extent and nature of potential impacts (e.g., how does a Regional or Program Office determine the appropriate level of analysis that is warranted);
- How to properly frame the question (e.g., whether the key concern is facility location, multiple or cumulative risk, or a specific health impact, for example, cancer, asthma, and developmental or neurological disorders);
- How to incorporate measures of population or community vulnerability;
- How to incorporate measures of health or environmental impacts (e.g., differential exposure, differential preparedness or ability to recover);
- What methodologies can be used for evaluation (e.g., comparison of summary statistics, regression analysis, or multi-level analysis); and
- How to select the appropriate methodology.

One example of an approach for considering potential disproportionate impacts of federal agency actions under the National Environmental Policy Act (NEPA) is found in *Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance Analyses* (US EPA, 1998a)[232] According to the guidance, the environmental impact assessment should include an analysis of "interrelated social and economic effects, …scaled according to the severity of the impacts." It is recommended that potentially disproportionate impacts be identified through a screening exercise designed to address two basic questions:

- To what extent does the potentially affected population include racial or ethnic minorities and/or low-income residents?
- Are the environmental impacts likely to fall disproportionately on racial or ethnic minority and/or low-income members of the population and/or tribal resources?

In the event that no disproportionate impacts on minority or low-income populations are identified, the guidance suggests that the steps and results of the scoping process be properly documented and included in the environmental impact assessment.

While this chapter does not recommend a particular approach, the analyst should keep these issues in mind and carefully document the way in which each is addressed throughout the course of an equity analysis.[233]

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[232] U.S. EPA, *Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance Analyses* (1998), http://www.epa.gov/compliance/resources/policies/ej/ej_guidance_nepa_epa0498.pdf

[233] For EPA's most current environmental justice guidance, guidelines, policies, practices and other resources, *see* EPA's Office of Environmental Justice homepage: http://www.epa.gov/compliance/environmentaljustice/index.html

Text Box 9.2 - Environmental Justice: Examples of Environmental Equity Assessment

The approaches that have been used by both EPA and entities outside of EPA to assess disproportionate impacts are as varied as environmental exposures and risks to consider. For example, some approaches use simple screening tools and descriptive statistics. Other approaches involve multiple linear regression modeling. Below is a discussion of three types of disproportionality analyses that have commonly been used, listed here in order of increasing complexity: (1) proximity to hazards; (2) cancer risk/cumulative risk analysis; and (3) epidemiological or health outcome analysis.

Proximity-to-hazards studies evaluate how the distribution and proximity of hazards (e.g., Superfund sites, toxic emissions, existing waste facilities) relate to population demographics. Residential proximity to a waste site or other hazard is often used as a surrogate measure for exposure to contaminants found at those sites (see Anderton, et al. 1997; Davidson and Anderton 2000; Mantaay 2001; Perlin, et al. 2001; Wilson, et al. 2002).

Cancer risk/cumulative risk analysis uses exposure (e.g., model estimates of ambient levels of pollutants) combined with toxicity information to derive estimates of health risks following EPA's standard risk assessment methodology (see Morello-Frosch, et al. 2002). If modeling coefficients are provided by race, ethnicity, and/or income, or allow inferences to be made about such subpopulations, equity analyses are then conducted.

Finally, epidemiological or health outcome analysis uses individual level data on health status or health effects that may be related to or exacerbated by environmental exposures/conditions to look for differences by income and racial/ethnicity (see Gwyn and Thurston 2001). Regression techniques are sometimes employed.

9.7.2.1 Socially Disadvantaged Populations

Socially disadvantaged populations are classified by the Office of Management and Budget's *Statistical Policy Directive* 15 (Federal Register 1997). This directive defines five categories for race: American Indian or Alaska Native, Asian, Black or African American, Native Hawaiian or Other Pacific Islander, and White. In addition there are two categories for ethnicity: Hispanic or Latino and Not Hispanic or Latino. Traditionally, a socially disadvantaged population is identified where either: (1) the population of concern in the affected area exceeds 50 percent or (2) the percentage of the population of concerns in the affected area is meaningfully greater than its corresponding percentage in the general population (or other appropriate unit of geographic analysis). A socially disadvantaged population also exists if there is more than one group of concern present and the percentage calculated by aggregating all socially disadvantaged persons meets one of these thresholds. In identifying socially disadvantaged populations the Agency may consider as a population either: (1) a group of individuals living in geographic proximity to one another or (2) a geographically dispersed/transient set of individuals (such as migrant workers, children), where either type of group experiences common conditions of environmental exposure or effect. The selection of the appropriate unit of geographic analysis may be a governing body's jurisdiction, a neighborhood, census tract, or other similar unit that is to be chosen so as to not artificially dilute or inflate the affected socially disadvantaged population. The selection of the appropriate unit of geographic analysis may also be influenced by the accuracy and precision of environmental quality models (see Text Box 3).[234]

[234] For EPA's most current environmental justice guidance, guidelines, policies, practices and other resources, *see* EPA's Office of Environmental Justice homepage: http://www.epa.gov/compliance/environmentaljustice/index.html

9.7.2.2 Economically Disadvantaged Populations

Low income or economically disadvantaged populations in an affected area can be identified with the annual statistical poverty thresholds defined by the Census Bureau, *Current Population Reports, Series P-60 on Income and Poverty*.[235] In conjunction with Census data, the analysis should also consider state and regional low-income and poverty definitions as appropriate. As with minority population, in identifying low-income populations, the Agency may consider as a population either a group of individuals living in geographic proximity to one another, or a geographically dispersed/transient set of individuals (such as migrant workers children), where either type of group experiences common conditions of environmental exposure or effects.[236]

Text Box 9.3 - Environmental Justice: Methods for Consideration in Defining an Impacted Population

EPA's *Final Guidance for Incorporating Environmental Justice Concerns in EPA's Compliance Analyses* (U.S. EPA 1998a) recognizes that "environmental effects are often realized in inverse proportion to the distance from the location or site of the proposed action."

The analyst's definition of the neighborhood may affect the results of a study. Broad neighborhood definitions may hide important relationships between environmental impacts and socioeconomic status, while narrow definitions may exclude areas that should be included (see Anderton, et al. (1994)). The ideal way to define the neighborhood is based on the exact potential exposure of the surrounding population to pollution from the plant or site.[237] It is nearly impossible to acquire such precise data, however, and so many have alternative definitions of the affected population.. Early environmental justice work used fairly broad neighborhood proxies in the form of counties or zip codes (see UCC (1987) or GAO(1983)). More recent studies have defined the relevant neighborhood more precisely based on concentric circles surrounding a site, distance to the site, or the census tract.

The use of circles approximates the distance at which a resident is concerned about the location of a hazardous facility. Concentric circles may be preferred to a census-based neighborhood definition for several reasons. First, a census-based definition ignores how households or different socioeconomic groups are distributed within the neighborhood; this is problematic when assuming broad neighborhood definitions. Second, a census-based definition often reflects topographical features such as rivers and highways that may exclude a large portion of those who, although separated by some physical feature, receive a large portion of the negative externalities or the potential effect of a regulation from the site or plant.

One reason for not employing the concentric circle technique is the arbitrary choice of a radius: the circles drawn are unlikely to reflect population-defined borders between neighborhoods. Most studies in the literature that use these techniques select a range of radii to examine how sensitive results are to alternative definitions. Also, the technique of drawing circles around a site usually involves the use of the Geographic Information System (GIS), which is difficult to apply to anything other than the most recent census.[238]

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[235] *Current Population Reports*, Series P-60 are available on line at: http://www.census.gov/prod/www/abs/income.html

[236] For EPA's most current environmental justice guidance, guidelines, policies, practices and other resources, *see* EPA's Office of Environmental Justice homepage: http://www.epa.gov/compliance/environmentaljustice/index.html

[237] RSEI considers the following information in calculating risk-weighted air and water releases: the amount of chemical released, the location of the release, the toxicity of the chemical, its fate and transport through the environment, the route and extent of human exposure, and the number of people affected. More information on RSEI is available at http://www.epa.gov/opptintr/rsei/. Accessed 5/18/04.

9.7.3 Children and the Elderly

Two subpopulations that may be disproportionately impacted by a rule are children and the elderly. Both groups may experience differential exposures to environmental contaminants compared to the rest of the population due to either developing (in the case of children) or weakened (in the case of elderly) biological systems and different activity patterns.

9.7.3.1 Children

E.O. 13045, *Protection of Children From Environmental Health Risks and Safety Risks,* 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 order also states that each "covered regulatory action"[239] submitted to OMB, unless prohibited by law, should be accompanied by ". . . an evaluation of the environmental health or safety effects of the planned regulation on children." The term "children" is not defined. The EPA's Office of Children's Health Protection, established in response to this Executive Order, does not use a single definition of child, but instead suggests that the definition will vary depending upon the issue(s) of concern. The Children's Health Valuation Handbook (U.S. EPA 2003b) defines a child as any person less than 18 years of age. This definition should be applied when conducting an equity assessment.

9.7.3.2 Elderly

In October 2002, EPA announced plans for an *Aging Initiative*. One goal of this initiative is to study the relationship between the elderly and environmental health hazards. As part of this announcement EPA also proposed to develop a *National Agenda for the Environment and the Aging*, including public participation in the process.[240] There are three components to the National Agenda:

- Identify research gaps in environmental health hazards to older persons;
- Prepare for an aging society in a smart growth context; and
- Encourage older persons to become involved in communities to reduce environmental hazards and protect the environment.

While there are no standard procedures for including the elderly in an impact analysis, EPA stresses the importance of addressing environmental issues that may adversely impact the elderly.

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[238] The Neighborhood Change Database (NCDB) provides Census data over time using a common tract definition. See http://www.geolytics.com/USCensus,Neighborhood-Change-Database-1970-2000,Products.asp. Accessed 12/2/04.

[239] A "covered regulatory action" is any substantive action in a rule making that is likely to result in a rule that may be economically significant (have an annual effect on the economy of $100 million or more or would adversely affect in a material way the economy, a sector of the economy, or the environment) and concern an environmental health risk that an agency has reason to believe may disproportionately affect children.

[240] See www.epa.gov/aging/agenda/index.htm#naea for more information on the National Agenda Accessed May 2, 2008.

9.7.4 Disproportionately High and Adverse Exposure or Human Health Effects

When determining whether human health effects are disproportionately high and adverse for a minority or low-income population, consider the following factors:[241]

- What is the proximity to the environmental or health hazard(s)?
- Are there unique exposure pathways?
- Are specific subpopulations highly susceptible and/or highly exposed?
- Are exposures known to be associated with an adverse health effect?
- Have the synergistic effects of multiple or cumulative exposures been assessed and addressed by a reasonable margin of safety?
- Are the potential health effects upon the population significant, unacceptable, or above generally accepted norms?

The definition of adverse health effects, risk, or rate of hazard exposure contained in specific environmental statutes under whose authority a regulation is being developed may also guide consideration in the equity assessment.

9.8 Framework for Assessing Equity Considerations

The following is a very general framework to guide analysts conducting equity assessments. For each component, choosing to measure the equity-related consequences of a regulation involves balancing costs of data acquisition and analysis against the value of improved accuracy. The framework attempts to conserve resources by screening out situations for which any of the equity impacts are unlikely. This permits analytical and empirical efforts to focus on circumstances with a higher probability of creating significant equity-related effects. The three components should not necessarily be viewed as sequential steps. Instead, at the outset of a particular regulatory analysis, all aspects of the suggested approach should be considered. This will ensure that the data gathered and the analyses performed are well suited to measuring the equity impacts of concern.

9.8.1 Equity Screening Analysis

An equity screening analysis consists of several tasks described below in the order in which they would be implemented.

- **Determine which sub-populations are within the scope of the analysis.** In certain cases, some sub-populations may not be connected closely enough to the regulation to be meaningfully affected. For example, governmental entities might not be involved in activities affected by the regulation. If so, then no further analysis is necessary for these groups.
- **Determine whether the rule or regulatory alternative imposes costs, offers benefits, or results in other economic effects too small to warrant further analysis.**[242] When considering costs it might be possible to argue that incremental unemployment or impacts on small businesses, for example, resulting from even small regulatory costs cannot be distinguished from

[241] For EPA's most current environmental justice guidance, guidelines, policies, practices and other resources, *see* EPA's Office of Environmental Justice homepage: http://www.epa.gov/compliance/environmentaljustice/index.html

[242] An economic impact analysis is inapplicable in screening actions that are health, rather than technology, based.

the underlying economic variability inherent in these activities. This also applies when a regulation imposes one burden on an entity, but reduces another on the same entity, so that the net effect is small. While some equity impacts are dismissed on the basis of this screening analysis, others may require further analysis.

- **Identify which equity dimensions require further analysis.** Negative impacts on small entities, low income or minority populations, and children are important to consider in all cases because of statutory and other mandates. In addition to equity dimensions that must always be considered for distributional analysis, efforts should be made to determine whether other dimensions listed in Table 9.1 are relevant. Analysts should collect readily accessible information on the characteristics of affected entities and populations to make this determination, paying special attention to who is expected to receive the benefits of the regulation as well as who will pay the costs.
- **Prioritize relevant equity dimensions.** Assuming there is more than one relevant equity dimension, each should be prioritized according to which dimension seems to warrant greatest concern. The level of concern should be determined by how strongly analysts expect a regulation to affect a particular sub-population.

9.8.2 Distributional Variables

The next step in assessing impacts on certain subpopulations is to define distributional variables associated with the equity dimensions identified from the screening step (section 9.5.1). For example, if a regulation's potential impact on poor neighborhoods is of concern, then a classification system for "poor" versus "not-poor" neighborhoods should be developed. The established definitions reviewed in Section 9.4.2 may be used, or appropriate alternatives may be developed. Sensitivity to a specific definition might be examined. In either event, the methods employed and their rationale should be well documented.

9.8.3 Measuring Equity Consequences

Finally, specific economic effects must be measured across the carefully defined subpopulations from the screening step. In some cases, estimating the equity-related effects of a regulation will involve disaggregating costs and benefits across these subpopulations and tabulating or otherwise accounting for their distribution. In other cases, new impacts may need to be estimated. This process should be consistent with the assumptions made in the BCA.

9.8.4 Data

The U.S. Census Bureau collects household data in forms that may be useful for environmental justice matters. Data are available on population distributions by race/ethnicity, age, household income, education,, language (to determine populations with limited English proficiency) at the state, county, metropolitan statistical area, or census tract level. An additional Census web site allows one to view a map of any part of the U.S., at the desired scale, that shows data on population distributions by family income, or a specified race (e.g., percent white, or percent black).[243] Income data collected by the Internal Revenue Service and made available in aggregated form on the Internet may be useful for some analyses.[244]

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[243] The address for this site is http://www.census.gov/geo/www/tiger/index.html. Accessed May 2, 2008.

[244] The address of the website providing these data is http://trac.syr.edu/tracirs/. Note that a user ID and password are necessary to access the data. Registration at http://trac.syr.edu/register/registration.html. Accessed May 2, 2008.

In addition, EPA has identified five federally-recognized or managed databases to aid in the identification of areas with potentially disproportionately high and adverse environmental and public health burdens. These include the following environmental databases: (1) National Air Toxics Assessment (NATA) cancer risk; (2) NATA noncancer neurological and respiratory hazard index; (3) NATA noncancer diesel particulate matter (PM); (4) toxic chemical emissions and transfers from industrial facilities, as modeled using the Risk-Screening Environmental Indicators (RSEI) tool; 5) population weighted ozone monitoring data; and 6) population weighted PM 2.5 monitoring data.[245]

10 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 policymakers.

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 policymaking process.

This chapter begins by discussing the components, or inputs, of an economic analysis, and how they can best be presented. Following this is general guidance on how to present the results of economic analyses, after which the impact of uncertainty on both the inputs and results of an economic analysis is discussed. Some brief comments on the relationship between economic analyses and environmental policy making conclude the chapter.

10.1 Communicating Data, Methods, Assumptions, and 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.

10.1.1 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 an Agency Guidance, *Guidance for Quality Assurance Project Plans,* that includes procedures for documenting secondary data (U.S. EPA 2002b)

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. In addition, when data are available from more than one source, a rationale for choosing the source of the data should be provided.

10.1.2 Methods

The economic impact of an environmental policy is often determined by many variables that have complicated relationships with one another. It is frequently impossible, without the benefit of a formal theoretical model, to keep track of the relationships between these variables and the policy in question. Economic models are tools that allow analysts to make informed judgments about the economic effects of different policy options. It would be prohibitively difficult to develop a reliable estimate of a given policy's impact on compliance costs, affected products' prices, and other important economic dimensions without using a quantitative economic model.

Modern economies are highly complex systems, composed of many discrete but interacting processes. Some of these processes may be significantly affected by environmental regulations; others will be largely undisturbed. The key to selecting the right tool for any given economic analysis is ensuring that key processes are illuminated while avoiding irrelevant details.

Quantitative economic models are, in essence, economic theories that have been numerically represented. As such, quantitative economic modelers must decide which aspects of an economic system to include, and which to omit. Some regulations may have significant impacts on commodity markets (e.g., mobile source emission reductions which motivated the use of platinum-based catalysts.) In such a case, the analyst must choose a tool that captures the upstream general equilibrium effects in the platinum market.

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.

Finally, 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.

10.1.3 Assumptions

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 also 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 the key assumptions and the potential magnitude and direction of likely errors in assumptions in the summary of results.

10.1.4 Addressing Uncertainty Driven by Assumptions and Model Choice

The analysis should address uncertainties resulting from the choices the analyst has made. 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 benefit-cost analysis 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 making this choice 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.

10.2 Presenting Results of Economic Analyses—General Issues

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 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.

Economic analysis is most useful when all results are reported in a common metric—specifically, monetary units. Whenever possible, benefits and costs should be reported in monetary 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). Agencies are required to provide OMB with an accounting statement reporting benefit and cost estimates when sending over each economically significant rule. These *Guidelines for Preparing Economic Analyses* and *Circular A-4* should be relied upon for developing these estimates. Figure 10.1, drawn from OMB's *Circular A-4*, is OMB's suggested accounting statement.[246]

10.2.1 Providing Appropriate Context for Analytical Results

Economic analysis is often motivated by a desire to find an optimal outcome—e.g., 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 (e.g. best available control technology, lowest achievable emission rate, etc.). Policy makers rely on quantitative analysis to promulgate these approaches, in particular on analyses that delineate the costs, benefits or other impacts of a wide range of control options. Unless the effects of "too much" and "not enough" regulation are assessed, it will be difficult to know what the "just right" level of control is, even under command and control approaches.

Economic analyses of potential policy options should present estimates of the costs and benefits of the preferred option, as well as for at least two other options. OMB's *Circular A-4* required that at least one option be more stringent and at least one option less stringent than the chosen or preferred option. Each option should be considered reasonable (e.g., cost effective options that achieve the same environmental outcome). If the options can be rank-ordered by level of stringency, the incremental costs and benefits of each option should be reported.

10.2.2 Reporting the Effects of Uncertainty on Results of Analysis

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

[246] Pages 44-46 of US OMB (2003) provide further discussion of the information to be provided in the accounting statement.

(see U.S. EPA, 1997). Delphic panels, or expert elicitation techniques, can help close knowledge gaps surrounding key relationships (see IEC, 2004).

Many economic analyses performed at EPA include assessments of economic impacts 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

Table 10.1 - OMB Accounting Statement (adapted from Circular A-4, p. 47)

Category	**Primary Estimate**	**Minimum Estimate**	**Maximum Estimate**	**Source Citation (RIA, Preamble, etc.)**
Benefits				
Monetized benefits				
Annualized Quantified, But Unmonetized, Benefits				
(Unquantified) Benefits				
Costs				
Annualized monetized costs				
Annualized Quantified, But Unmonetized, Costs				
Qualitative (Unquantified) Costs				
Transfers				
Annualized Monetized Transfers: "On Budget"				
From Whom To Whom?				
Category		**Effects**		**Source Citation (RIA, Preamble, Etc.)**
Effects on State, Local, and/or Tribal Governments				
Effects on Small Businesses				
Effects on Wages				
Effects on Growth				

and reported when possible, to allow 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.)

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.[247]

Discounting

Generally, benefits and costs that accrue sooner are worth more than benefits and costs that accrue later. Chapter 6 deals extensively with issues surrounding appropriate discounting of streams of costs and benefits. When an economic analysis estimates different time streams of costs and benefits, these should be presented for the reader, expressed in constant (inflation-adjusted) dollars and undiscounted. Results should also be presented as discounted or annualized results, with appropriate discount rates used (and reported). 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.

10.3 Reporting the Results of Specific Types of Analyses

The results of economic analyses of environmental policies should generally be presented in three sections.

- **Results from Benefit-Cost Analysis.** Estimates of the net social benefits should be presented based on the benefits and costs expressed in monetary terms. A discussion of non-monetized and unquantifiable benefits and costs should also be provided.
- **Results from Cost-Effectiveness Analysis.** Under OMB Circular A-4, cost-effectiveness analysis 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.[248]
- **Results from Economic Impact Analysis and Equity Assessment.** Results of the economic impact analysis and equity assessments should be reported, including predicted effects on prices, profits, plant closures, employment, and other effects, such as human health. 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.

[247] A forthcoming chapter in the Guidelines will more fully address uncertainty issues.

[248] The Institute of Medicine (IOM 2006) recently issued recommendations to regulatory agencies on how to perform health-based cost effectiveness analyses. Guidance on performing cost effectiveness analysis of environmental policies is currently under development. Recent examples of cost effectiveness analyses 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) and the Ground Water Rule (see Appendix H listed at http://www.epa.gov/safewater/disinfection/gwr/regulation.html).

10.3.1 Presenting the Results of Benefit-Cost Analyses

When presenting the results of a benefit-cost analysis, 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 Chapter 6, appropriately discounted benefits and costs should be reported as well.

Ideally, all benefits and costs of a regulation will be expressed in monetary terms, which greatly ease comparisons across options. However, an important benefit or cost category should not be excluded from a benefit-cost analysis if it cannot be monetized. 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 nonmonetized 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/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.[249] In this example, the cost side of the equation includes general equilibrium feedback effects while the benefit side does not and this difference should be clearly presented and explained.

10.3.2 Presenting the Results of Cost-Effectiveness Analyses

When benefit-cost analysis is not possible, cost-effectiveness analysis may be the best available option. The cost-effectiveness of a policy 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 Regulatory Impact Analyses, or other analyses of specific regulatory or policy options, cost-effectiveness analysis 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

[249] 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, Jena et al. 2008)

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 cost-effectiveness analysis 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 cost-effectiveness analysis will be. Consider the following steps that a typical environmental economic assessment follows:

- Changes in emissions are estimated (e.g., tons of emissions), then
- 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 cost effectiveness analyses.

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 the value of the analysis to decision makers (and the public).

When presenting the results of a cost effectiveness analysis, 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.

Cost-effectiveness analysis 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 cost-effectiveness analysis. 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.) (US EPA 2003b).

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, cost-effectiveness analysis can indicate which control measures or policies are inferior options.

10.3.3 Presenting the Results of Economic Impact Analysis and Equity Assessments

Economic impact analyses and equity assessments focus on distributional outcomes. Therefore, the presentation of these results should focus on disaggregating effects to show impacts separately for the groups and sectors of interest. If costs and/or benefits vary significantly among the sectors affected by the policy, then both costs and benefits should be shown separately for the different sectors. Presenting results in disaggregated form will provide important information to policy makers that may help them tailor the rule to improve its efficiency and equity outcomes.

The results of the economic impact analyses 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 equity assessments may include the distribution of benefits, costs, or both for specific subpopulations including those highlighted in the various mandates. These include minorities, low-income populations, small businesses, governments, non-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.

10.4 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, however, 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. Thus, 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 tradeoffs among them. While ultimately statutory requirements may dictate if and how the analytic results are used in standard setting, these results, nevertheless, serve as important inputs for this broader policy-making process along with other analyses and considerations.

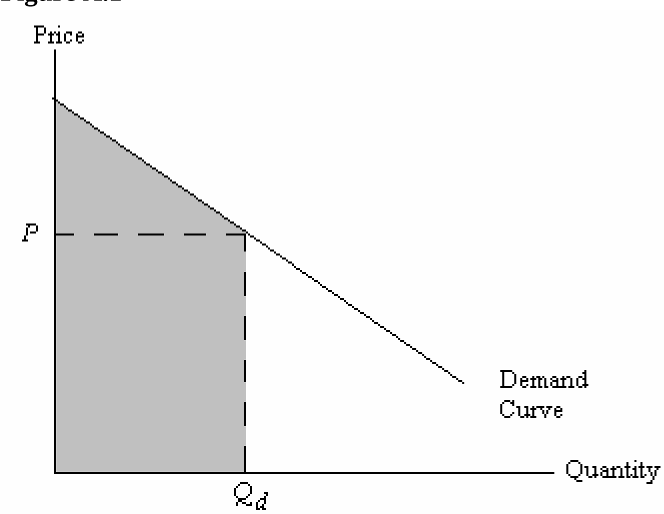
A. Economic Theory

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.[250] The usefulness of benefit-cost analysis 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, and 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[251] – of their choices, given their own resource constraints. 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.

Figure A.1



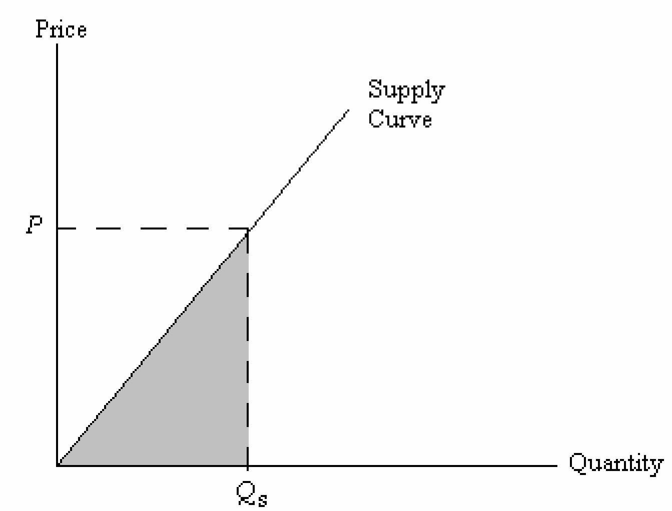
[250] EPA's mandates frequently rely on criteria other than economic efficiency as well, so policies are sometimes adopted that are not justified by the lack of efficiency.

[251] *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 (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 curve indicates the maximum price, y, an individual with x 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 afforded by the previous unit so 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 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 willingness to pay is one of the fundamental concepts used in economic analyses, and it is important to distinguish between total and marginal willingness to pay (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 gives the marginal WTP for the Q_d -th unit. The total WTP is equal to the marginal WTP for each unit up to Q_d – i.e., the shaded area under the demand curve from the origin up to Q_d.

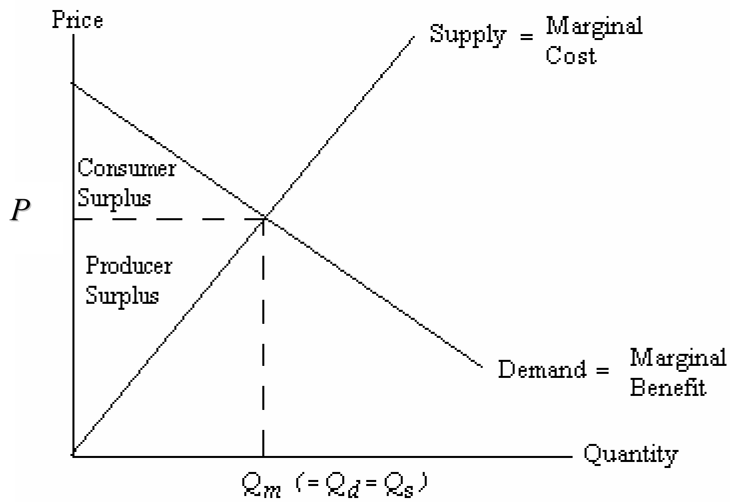
Figure A.2



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, and so 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 *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.[252] The market supply curve is simply the horizontal summation of the individual producers' marginal cost curves for the good or service in question.

[252] This is actually the long run total cost. In the short run there would be fixed costs as well.

Figure A.3



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 willingness to pay 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 price P_m and 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 may 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 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.[253]

[253] Section A.4.2 provides a more technical discussion of how consumer surplus serves as a measure of benefits.

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 willingness to pay 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 + producer surplus) is maximized and 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 occurs automatically.[254] 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.

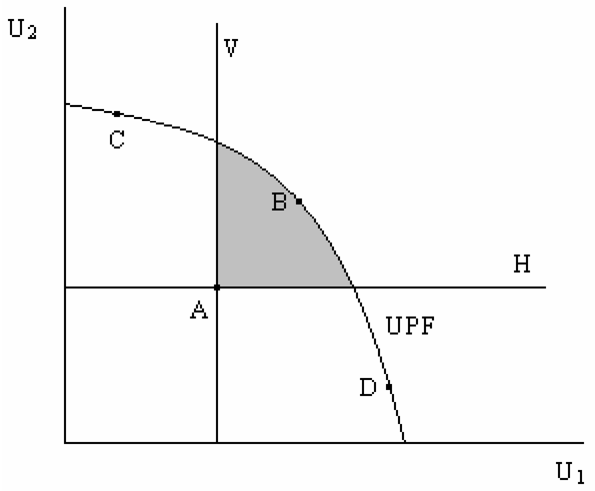
One graphical representation of these results is given in Figure A.4, which shows utility (welfare) levels in a two-person economy.[255] The curve shown is the utility possibility frontier (UPF) curve; the area within it represents the set of all possible welfare outcomes. Each point on the negatively 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

[254] 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.)

[255] Another, perhaps more commonly used, graphical tool to explain the First and Second Welfare Theorems is an Edgeworth box. See Varian (1993) or other basic economic textbook for a detailed discussion.

enhancing) outcomes include all points in the shaded area, bordered by *H*, *V*, and the UPF curve.[256] 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.[257]

Figure A.4



A.2 Reasons for Market or Institutional Failure

If the market supply and demand curves reflect society's true marginal social cost and willingness-to-pay, 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.[258] 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, machinery, energy – and 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

[256] Note that efficiency could be obtained by moving along the vertical line V, which keeps utility of person 1 constant while increasing utility of person 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.

[257] Note that outcomes on the frontier such as C and D, although efficient, may not be desired on equity, or fairness, grounds.

[258] 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.

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.[259] 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.[260] 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. These additional social costs are not included in firms' profit maximization decisions and so are not considered when firms decide how much pollution to emit. Thus, the lack of a market for clean air causes problems and provides the impetus for government intervention in markets involving polluting industries.

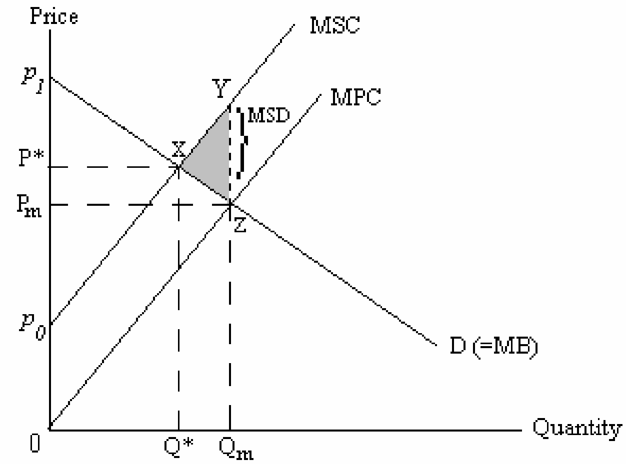
Figure A.5 illustrates a negative externality associated with the production of a good. For example, a firm producing some product might also be generating pollution as a by-product. The pollution may impose significant costs – in the form of adverse health effects, for example – on households living downwind or downstream of the firm, but 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:

- 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).

[259] 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).)

[260] Often these are goods that exhibit public good characteristics. Pure public goods are those which 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



In an incomplete market, producers pay no attention to external costs, and production occurs where market demand 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 willingness to pay minus total social costs) is equal to the area of the triangle p_0p_1X less the area of triangle XYZ.[261] 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. 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.[262] The shaded triangle (*XYZ*), referred to as the *deadweight loss*, 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.[263] Figure A.5 neatly summarized 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 Benefit-Cost Analysis (BCA).

BCA can be thought of as an accounting framework of the overall social welfare of a program, which illuminates the tradeoffs involved in making different social investments (Arrow et al., 1996). It is used

[261] Recall from Section A.1 that total willingness to pay 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 marginal social cost curve (MSC) from the origin to the point of production ($0p_0YQ_m$).

[262] Similarly, the private market will undersupply goods for which there are positive externalities, such as parks and open space.

[263] Chapter 4 discusses the various regulatory techniques and some non-regulatory means of achieving pollution control.

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.

Benefit-cost analysis can also be see 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 willingness to pay for the environmental commodity. As with private markets, the efficient outcome is the option that maximizes net benefits.

As mentioned above, the key to performing BCA lies in the ability to measure both benefits and costs in monetary terms so that they are comparable. The 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 the 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 has been calculated, economists use a number of techniques to estimate the value that society places on these health improvements.[264] 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.

[264] Chapter 7 discusses a variety of methods economists use to value environmental improvements.

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.[265] There are also practical limitations to BCA. Most importantly, it requires assigning monetized values to non-market benefits and costs. In practice, it may be very difficult or impossible to quantify gains and losses in monetary terms (e.g., the loss of a species, intangible effects).[266] 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

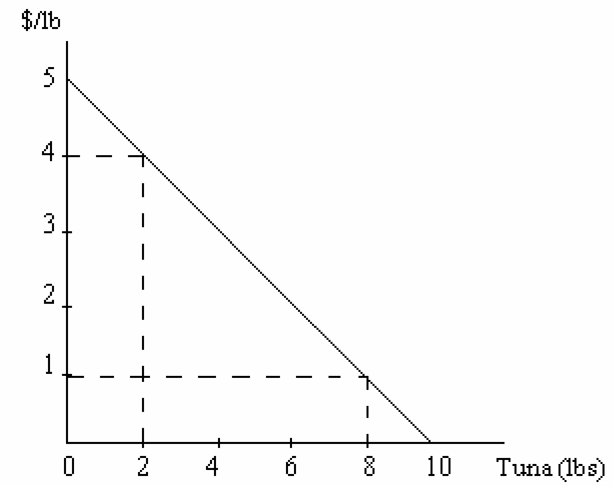
The net change in social welfare brought about by a new environmental regulation is the sum of the negative effects (i.e., loss of producer and consumer surplus) and the positive effects (or social benefits) of the improved environmental quality. This is shown graphically for a single market in Figure 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 responsive producers and consumers' decisions are to a change in price. Economists 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 one 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% increase in the price of tuna results in a 1% 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% increase in price would reduce quantity demanded by 2.5% (from 8 lbs to 7.8 lbs). At a price of $4 per pound, a 10% increase in price would result in a 40% 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. Therefore, when calculating elasticities it is important to state the price or quantity of the good demanded (or supplied).

[265] Chapter 9 addresses equity assessment and describes the methods available for examining the distributional effects of a regulation.

[266] Kelman (1981) argues that it is even unethical to try to assign quantitative values to non-marketed benefits.

Figure A. 6



Elasticities are important in measuring economic impacts because they determine how much of a 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).[267] 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. Supply characteristics in the industries affected by a regulation can be as important as demand characteristics in determining the economic impacts of a rule. For highly elastic supply curves 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, 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.

[267] 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".

Whether the affected product represents a substantial or necessary portion of customers' costs or budgets is another factor that affects demand elasticities. 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 than for basic requirements. If the good is a necessity item, the quantity demanded is unlikely to change drastically for a given change in price and 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, as customers begin to find substitutes or as new substitutes are developed. However, temporary price changes may affect consumers' decisions differently than permanent ones. The response of quantity demanded during a 1-day sale, for example, will be much greater than the response of quantity demanded when prices are expected to decrease permanently. Finally, it also 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 costs per unit 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 also 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.[268] When

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[268] Another useful source of elasticity estimates is the recently developed EPA Elasticity Databank. 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 (EIA) studies conducted by the Agency since

such information is unavailable, as is often the case for intermediate goods, elasticities may be quantitatively or qualitatively assessed.[269] 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 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.[270] 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 benefit-cost analysis. 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 movements along the Hicksian demand function can be used to determine the monetary change that would compensate the consumer for the price change, considering the substitution effect alone.

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.

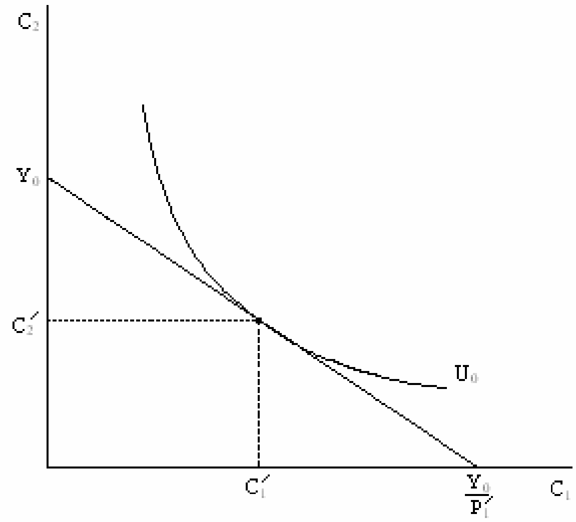
1990 as well as estimates found in the economics literature. It may be accessed from the Technology Transfer Network Economics & Cost Analysis Support website: http://www.epa.gov/ttnecas1/Elasticity.htm.

[269] 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).

[270] See Perman et al. (2003), Just et al. (2005), or any graduate level text for a more thorough exposition of this issue.

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.[271] This can be shown with indifference curves and a budget line, as seen in Figure A.7.

Figure A. 7



Assume that the consumer is considering the tradeoff 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 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.[272] A utility-maximizing consumer will choose quantities C_1' and C_2', the point where the indifference curve is tangent to the budget constraint.[273]

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[274] 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*.

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[271] The notation and discussion in this section follow Chapter 12 of Perman et al. (2003).

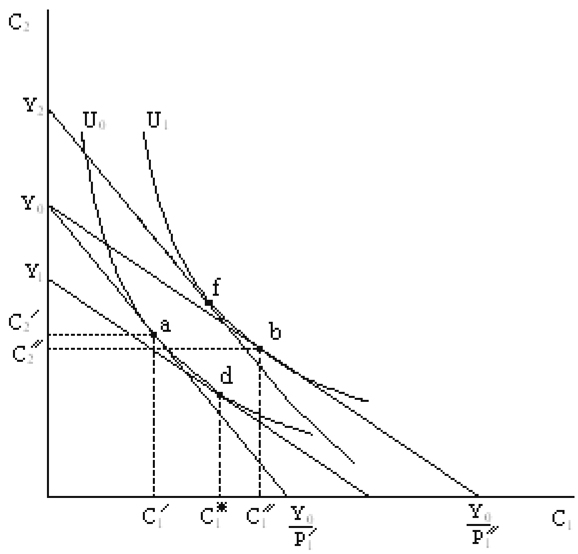
[272] In Figure A.7, C_2 is considered the numeraire good (i.e., prices are adjusted so that P_2' is equal to 1).

[273] For a review of the utility maximizing behavior of consumers, see any general microeconomics textbook.

[274] 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*.

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.[275] Notice that in the case of a price decrease, the CV measures how much the consumer would be willing to pay (WTP) to receive the price reduction and EV measures how much the consumer would be willing 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.

Figure A.8



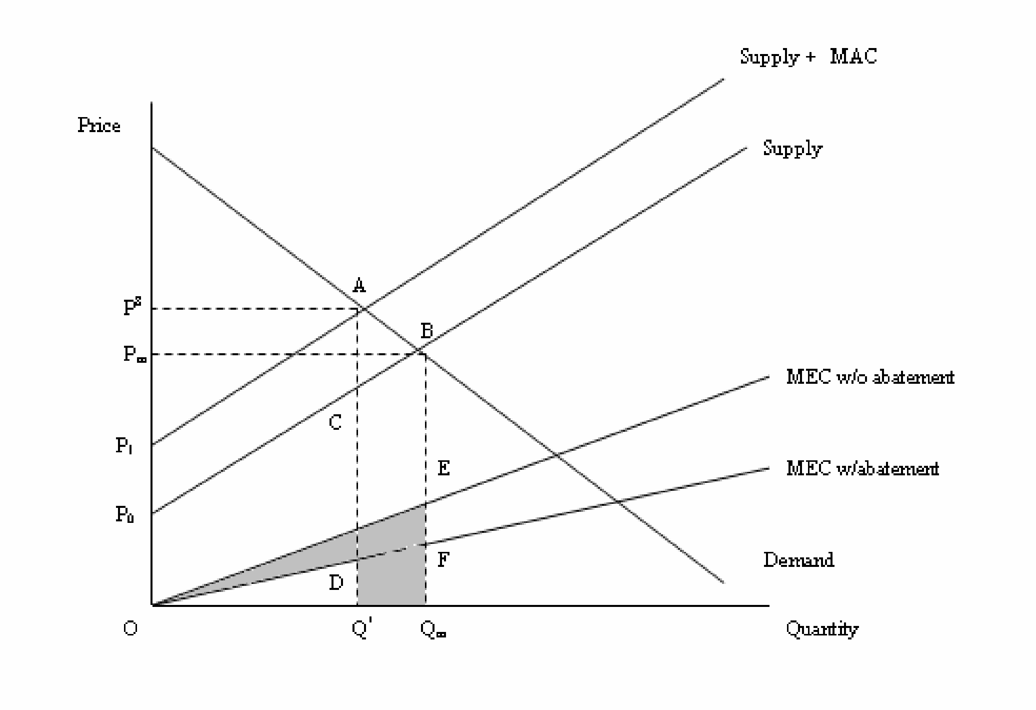
In order to examine the implications of these measures for valuing changes in environmental conditions, one can think of C_1 in the above discussion as an environmental commodity, henceforth denoted by *E*. Then an improvement in environmental quality (or an increase in an environmental public good) 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 (WTP) 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 (WTA) to not experience the gain in

[275] 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.

environmental quality. If valuing an environmental degradation, then CS measures the WTA and ES measures WTP.

Figure A.9



Whereas statements can be made about the relative size of CV, EV, and MCS for price changes of normal goods,[276] it is not possible to make similar statements about CS, ES, and MCS for a change in environmental quality (Bockstael and McConnell, 1993). 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 can be illustrated in Figure A.9 where the market

[276] Willig (1976) shows that ordinary, or Marshallian, demand curves may 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% (see Perman et al., 2003).

produces the quantity Q_m in equilibrium without intervention. The external costs of production are shown by the marginal external costs (MEC) curve without 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_1. 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 also 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 was 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.[277]

In some cases, a regulation may have such a significant impact on the economy that a general equilibrium modeling framework is required.[278] 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

[277] 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 (EPA, 1989).

[278] *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.

linkages between all sectors of the economy and all feedback effects, and can measure total costs comprehensively.[279]

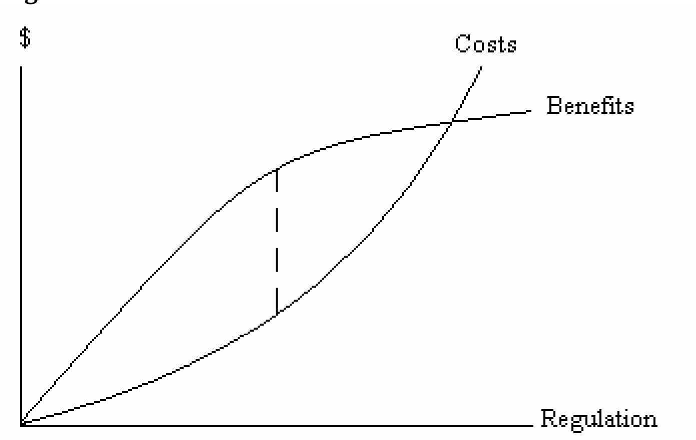
A.5 Optimal Level of Regulation

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.

Note that this is *not* necessarily the point at which:

- Benefits are maximized,
- Costs are minimized,
- Total benefits = total costs (i.e., benefits/costs ratio = 1),
- Benefits/costs ratio is the largest, or
- The policy is most cost-effective.

Figure A.10



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 benefits/costs 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 benefits/costs ratio is not optimal either. Two policy options could yield equivalent benefits/costs 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.[280] Finally, finding the most cost-effective policy has similar

[279] Chapter 8 provides a more detailed discussion of partial equilibrium, multi-market, and general equilibrium analysis.

[280] However, 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

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)[281] is equal to the cost of abating an additional unit (i.e., marginal abatement cost).[282] If the marginal 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 it provides in benefits, and will make it 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.*[283]

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

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$750. If options are selected according to the net benefits criterion, the first option would 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 would 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 would yield $1,000 in total net benefits, while choosing options by the benefit-cost ratio criterion would 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 (e.g., 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 cost-effectiveness analysis (e.g., Hyman and Leibowitz (2000)), while the net-benefit criterion is not.

[281] 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.

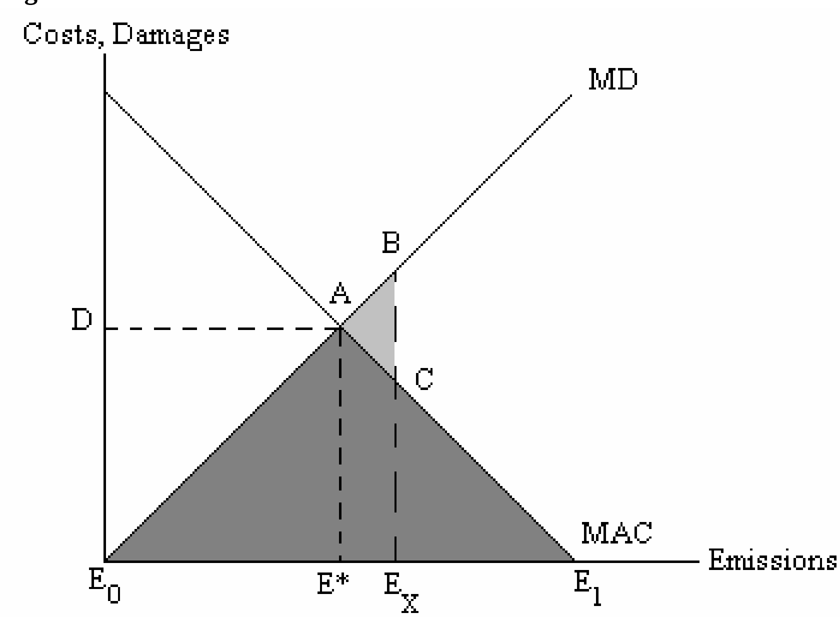
[282] The idea that a given level of abatement is efficient – as opposed to abating until pollution is equal to zero – is based on the economic concept of diminishing returns. For each additional unit of abatement, marginal social benefits decrease while marginal social costs of that abatement increase. 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 willingness to pay for pollution equals the marginal social costs of polluting.)

[283] 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.

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.[284]

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.[285] Emissions are drawn on the horizontal axis and increase from left to right.. The damages from emissions are represented by the marginal damage curve (MD). 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



[284] 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 (2002), p. 105, or any other environmental economics text for a detailed explanation and example.)

[285] 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)).

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 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^*. In this case, 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 marginal damage (MD) curve) and the marginal abatement cost (MAC) curve intersect – is economically efficient because it imposes the least net cost on (i.e., yields the highest net benefits for) society. That is, the triangle E_0AE_1 is the smallest shaded region that can be obtained.

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 (2002) 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 (2000) and Portney and Stavins (2000). Finally, general microeconomics textbooks (Mankiw (2004), Varian (2005) at the undergraduate level, and Mas-Colell et al. (1995), Kreps (1990), Varian (1992) at the graduate level), and applied welfare economics textbooks (Just et al., 2005) are useful references as well.

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 benefit-cost analyses, 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" to express these benefits in monetary terms.[286, 287]

The risk reductions themselves can generally be classified according to the characteristics of the risk in question (e.g., voluntariness, controllability) and the characteristics of the affected population (e.g., age, 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 4.3 and will not be repeated here.

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 $7.0 million ($2006) with a standard deviation of $4.7 million.[288] [289] EPA recommends that the central

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[286] Suppose a policy affects 100,000 people and reduces the risk of premature death by 1 in 10,000 for each individual. Summing these individual risks across the entire affected population results in 10 statistical lives "saved" if the policy is implemented. Suppose, too, that individuals are willing to pay $500 each for the risk reduction. Aggregating the individual values to get the value of $500 statistical life would yield a VSL of $5 million (=$500* 1/(risk reduction)). The total mortality benefits for the policy would be $50 million.

[287] It is important to emphasize that the Value of Statistical life does not represent the value of an identifiable person. Rather it is the value associated with a *small* change in the probability of dying, aggregated up to the value of a statistical life.

[288] 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

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.[290] It remains EPA's default guidance for valuing mortality risk changes although the Agency has considered and presented alternatives.[291]

Other VSL Information

For most of mortality risk reductions EPA uniformly applies the VSL estimate discussed above. Beginning in 2003, the Office of Air and Radiation began using an estimate of $5.5 million (1999 dollars; $6.6 million in 2006 dollars) in its analysis of mortality risk reductions from air regulations. This estimate reflects a more recent review of the literature. The estimate is derived from the range of values estimated in recent meta-analyses of the hedonic wage literatures. Mrozek and Taylor (2000) estimate a lower bound of $1 million in a meta-analysis of the hedonic wage literature, while Viscusi and Aldy (2003) estimate $10 million as the upper end of the inter-quartile range in their meta-analysis of the same literature (though using a different set of studies). The central tendancy in the range produced by these two studies is $5.5 million and has been applied consistently to mortality risk reductions from air regulations since 2003.

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 USEPA 1997 for more details.

[289] This VSL estimate was produced using the Gross Domestic Product (GDP) Deflator inflation index. Some economists prefer using the Consumer Price Index (CPI) in some applications. The key issue for EPA analysts is that the chosen index be used consistently throughout the analysis.

[290] The studies listed in Table B-1 were published between 1974 and 1991, and most are hedonic wage estimates. Although these were the best available data at the time, the Agency is currently considering more recent studies as it evaluates approaches to revise its guidance.

[291] 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, 2006). The Agency is committed to using the best available science in its analyses and will revise this guidance in response to SAB recommendations (see USEPA 2007 for recent SAB recommendations).

Table B-1: VALUE OF STATISTICAL LIFE ESTIMATES
(mean values in millions of 2006 dollars)

Study	Method	Value of Statistical Life
Kneisner 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
Kneisner 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, 1979)	Labor Market	$6.07
R.S. Smith (1976)	Labor Market	$6.80
V.K. Smith (1976)	Labor Market	$6.92
Olson (1981)	Labor Market	$7.65
Viscusi (1981)	Labor Market	$9.60
RS.Smith (1974)	Labor Market	$10.57
Moore and Viscusi (1988)	Labor Market	$10.69
Kneisner 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.

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.

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:[292]

- 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.[293] 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.

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 Science Advisory Board (SAB) reviewed an Agency summary of the available empirical literature on the effects of risk and population characteristics on willingness to pay (WTP) for mortality risk reductions (US EPA 2000c). 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 willingness-to-pay estimates to account for higher future income levels, though not for cross-sectional differences in income; (2) discounting risk reductions that are brought about in the future by

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

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

current policy initiatives (that is, after a cessation lag), using the same rates used to discount other future benefits and costs.[294] 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.*[295] Recent EPA analyses have assumed a triangular distribution from these values and used the results in a probabilistic assessment of benefits.[296]

- **Cessation lags:** 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.[297] This delay between the change in exposure and manifestation 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.

  A simple *ad hoc* approach to adjusting existing VSL estimates is to apply the social discount rate over the expected cessation lag. However, defining cessation lags with existing risk assessment methods may be difficult and empirical estimates highly uncertain.[298] Further, the underlying assumptions supporting this procedure may oversimplify how individuals appear to consider delayed health effects.[299]

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 Environmental Economics Advisory Committee on issues related to valuing changes in life expectancy for which age and baseline health status are close

[294] The results of the review can be found in "An SAB Report on EPA's White Paper *Valuing the Benefits of Fatal Cancer Risk Reductions*," EPA-SAB-EEAC-00-013, July 27, 2000 (website address http://www.epa.gov/sab/eeacf013.pdf , accessed 8/28/00).

[295] For details see Kleckner, N. and J. Neuman. 2000. Update to Recommended Approach to Adjusting WTP Estimates to Reflect Changes in Real Income. Industrial Economics, Incorporated. Memorandum to Jim DeMocker. U.S. Environmental Protection Agency. September 30, 2000.

[296] See, for example, page 6-84 of the *Final Economic Analysis for the Stage 2 DBPR,*(USEPA,2005).

[297] 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.

[298] See Chao and Gibb (2003) for an example.

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

correlates.[300] 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 willingness to pay 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 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.[301] In general this literature has shown that the relationship between age and willingness to pay for mortality risk changes is ambiguous, requiring strong assumptions to even sign the relationship.[302] Empirical evidence is also mixed. A number of empirical studies (mostly hedonic wage studies) suggest that the value of a statistical life follows a consistent "inverted-U" life-cycle, peaking in the region of mean age.[303] Others find no such statistically significant relationship and still others show willingness to pay increasing with age.[304] 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.[305]

  In spite of the ambiguous relationship between age and WTP, two alternative adjustment techniques have been derived from this literature. The first, *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 value of statistical life equals the sum of discounted values for each life year, and 2) each life year has the same value. This method was applied as an alternative case in an effort to evaluate the sensitivity of the benefits estimates prepared for EPA's retrospective and prospective studies of the costs and benefits of the Clean Air Act (US EPA 1997a and US EPA 1999a).

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[300] USEPA (2006) summarizes much of the literature related to the effects of age and health status on willingness to pay for changes in mortality risk and includes the charge questions put to the SAB-EEAC on these issues.

[301] See, for example, Shepard and Zeckhauser (1984), Rosen (1988), Cropper and Sussman (1988, 1990) and Johannson (2002).

[302] See Evans and Smith (2006) for a recent summary.

[303] See Jones-Lee et al. (1985), Aldy and Viscusi (2006) and Viscusi and Aldy (2007), Kniesner, Viscusi and Ziliak (2006).

[304] 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.

[305] 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 a 20 percent reduction for those aged 70 and over; however, this difference was not statistically significant.

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.[306]

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, however, especially when children are found to be a significant portion of the affected population.[307] 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 studies showing no statistically significant effects (Krupnick et al. 2002).[308]

Application of existing value-of-statistical-life-year 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.[309]

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

[307] See USEPA (2003) 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.

[308] 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).

[309] It should be noted that many observers have expressed reservations over adjusting the value of mortality risk reduction on the basis of population characteristics such as age. One of the ethical bases for these reservations is a concern that adjustments for population characteristics may imply support for variation in protection from

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, 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 Science Advisory Board as guidance recommendations are revised.

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environmental risks. Another consideration is that existing economic methods may not capture social willingness to pay to reduce health risks. Chapter 9 details how some these considerations may be informed by a separate assessment of equity.

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D. Subject Index