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Guidelines for Preparing Economic Analyses

Third Edition

Office of Policy
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www.epa.gov/economics

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Chapter 1 - Introduction

The *Guidelines for Preparing Economic Analyses* are part of the U.S. Environmental Protection Agency's (the EPA's) commitment to improve the preparation and use of sound science in economic analysis to inform decision making. Written primarily for the economic analyst, the main purpose of this document is to define and describe best practices for economic analysis grounded in the economics literature. It also describes Executive Orders (EOs) and other documents that impose analytic requirements and provides detailed information on selected important topics for economic analyses.

1.1 Background

Thorough and careful economic analysis is an important component for informing and developing sound environmental policies. High-quality economic analyses can greatly enhance the effectiveness of environmental policy decisions by providing policy makers and the public with data-driven information needed to systematically assess the consequences of various actions or options.[1] An economic analysis of a rulemaking is a positive exercise, as opposed to a normative one, that provides information on the potential economic efficiency of policy alternatives and assesses the magnitude and distribution of an array of impacts through careful investigation. Economic analysis also serves as a mechanism for organizing information carefully, identifying the kinds of impacts associated with stated policy alternatives, and projecting who will be affected. Ultimately, economic analysis based on sound science should lead to better-informed regulatory and policy decisions.

The *Guidelines for Preparing Economic Analyses,* hereafter *Guidelines,* focus on the conduct of economic analysis to inform policy decisions and to meet requirements described by related statutes, Executive Orders (EOs), and associated implementing guidance of those EOs.[2] Based on the state of science and economics at the time of its writing, this document is intended to ensure high-quality analyses and consistency in how these economic analyses are prepared, performed and reported. In so doing, the *Guidelines* elevate the quality of information shaping environmental policy decisions and EPA-issued guidance. The *Guidelines* also describe an interactive development process between analysts and decision makers; reviews and summarizes environmental economics theory and the practice of benefit-cost analysis; and emphasizes issues in practical applications.

1 It is important to note that economic analysis is but one component in the decision-making process. Depending on the statutory context, all or certain components of the economic analysis may not be used by or required for the legal rationale for the regulation. Other factors that may influence decision makers include statutory requirements, health risks, distributional considerations, enforceability, technical feasibility, policy priorities and ethics.

2 Chapter 2 describes many of these statues, EOs and the analytic and/or procedural requirements they impose, as well as associated guidance materials.

1.2 The Scope of the *Guidelines*

The *Guidelines* apply to economic analyses conducted for environmental policies using both regulatory and non-regulatory management strategies (e.g., support for voluntary programs) as well as Agency-issued guidance. Separate EPA guidance documents exist for related analyses, such as risk assessments, which can be inputs to economic analyses. No attempt is made here to summarize such guidance materials. Instead, their existence and content are noted in the appropriate sections.

The *Guidelines* assume the reader has some background in microeconomics as applied to environmental and natural resource policies. To fully understand and apply the approaches and recommendations presented in the *Guidelines,* readers should be familiar with basic applied microeconomic analysis, the concepts and measurement of consumer and producer surplus, and the economic foundations of benefit-cost analysis. Appendix A provides a brief review of economic foundations, and the Glossary defines selected key terms.

The *Guidelines* are designed to assist staff with the preparation of economic analyses but are not a rigid blueprint nor a detailed set of step-by-step directions for all economic analyses. The most productive and illuminating technical approaches for an analysis will depend on case-specific factors and will require professional judgment. The *Guidelines* are a summary of analytical methodologies, empirical techniques, best practices, and data sources that can assist in identifying and implementing those approaches.

Finally, it is important to note that while the *Guidelines* apply to all types of economic analysis, the focus is on benefit-cost analysis and economic impact analysis -- two mainstays of the EPA's economic analyses. Typically, these economic analyses are not independent from other analyses. Assessing the effects of environmental policy is an inherently complex process in which results from various disciplines are integrated and inform one another. Taken together, they are used to predict environmental and behavioral outcomes and their economic consequences.

1.3 Economic Framework for Analysis

Conceptually, the ideal economic framework for assessing the effects of policy actions is one of general equilibrium that defines the allocation of resources and interrelationships for an entire economy with all its diverse components (e.g., households, firms, government). Potential regulatory alternatives are then modeled as economic changes that move the economy from a state of equilibrium absent the regulation (the baseline), to a new state of equilibrium with the regulation in effect. The differences between the old and new states are measured as changes in prices, quantities of goods, services and factors produced and consumed, including environmental quality, as well as wealth, income, and other economic metrics. These measurements may then be used to characterize the net welfare change for each affected group to inform questions of efficiency and distribution, based on individuals' expected changes in their own welfare.

Questions about efficiency focus on aggregate changes in welfare. Economists generally define benefits as positive changes in welfare and costs as the opportunities foregone, or reductions in welfare. To assess efficiency under this scenario, we add these changes in welfare measured in monetary terms across all affected individuals. In the ideal, general equilibrium framework, we can estimate and sum all benefits and costs; so, a policy is a movement toward efficiency if the sum is positive and a movement away from efficiency if the sum is negative. The policy that maximizes this sum, i.e., net benefits, is considered economically efficient.[3]

Questions about the distribution of benefits and costs examine how specific groups of households and industries are affected by the policy. The ideal framework would answer questions framed in terms of welfare changes for groups of individuals (e.g., is the policy welfare-improving for a specific group?) or in terms of specific economic factors (e.g., how much will prices change for some goods?). These assessments of distributional outcomes are often important, apart from analysis of benefits and costs (i.e., economic efficiency).

In practice, of course, capturing this idealized framework empirically can be difficult, if not impossible, due to data availability and in most cases, it is not possible to monetize all benefits and costs. No single modeling tool allows us to answer all policy-relevant questions about efficiency and distributional effects.[4] As a practical matter, most economic analyses assemble a set of models to address these issues separately, but, even then, not all effects can be monetized. If limitations are appropriately described, however, it is still informative to present comparisons of benefits and costs that can be monetized and qualitatively characterized, as well as evaluations of effects on specific groups.

As detailed more fully in Chapter 2, economic analysis of benefits, costs and distributional impacts are required by EO 12866 for economically significant rules. Although EO 12291 in 1981 was the first to require an economic assessment of significant regulatory actions in a regulatory impact analysis (RIA), these analyses were not as extensive as the economic analyses required now by EO 12866. A complete economic analysis today, though it may still at times be labeled as an RIA, consists of a benefit-cost analysis and any related cost-effectiveness analyses and assessments of economic and distributional impacts. The Office of Management and Budget (OMB) has a useful checklist (shown in adapted form in Text Box 1.1) for all components of an economic analysis conducted under EO 12866 (OMB 2010).[5]

1.3.1 Assessing Economic Efficiency with Benefit-Cost Analysis (BCA)

Benefit-cost analyses assess economic efficiency using the *Potential Pareto* criterion: is it theoretically possible for those who gain from the policy to fully compensate those who lose, and remain better off? When the answer to this question is "yes," then net benefits (benefits minus costs) are positive and the policy is a movement toward economic efficiency.[6]

While conceptually identical, benefits and costs are often evaluated separately due to practical considerations. The benefits of reduced pollution are often attributable to changes in outcomes not

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3 Appendix A provides a conceptual overview of the economic theory of welfare changes and benefit-cost analysis.

4 As discussed in Chapter 8, computable general equilibrium (CGE) models capture most, or all, modeled market benefits and costs, but may not include non-market benefits. In practice, CGE models may be unable to analyze relatively small sectors of the economy. See Chapter 8, Section 4.6.

5 The questions in Text Box 1.1 have been reproduced with minor modification from the OMB checklist without the extensive footnotes. The footnotes and other details about the checklist can be found at https://www.whitehouse.gov/wp-content/uploads/legacy_drupal_files/omb/inforeg/inforeg/regpol/RIA_Checklist.pdf.

6 Appendix A describes the underlying economic theory in greater detail.

exchanged in markets, such as improvements in public health. In contrast, the costs are generally measured through changes in outcomes that are exchanged in markets, such as pollution control equipment. As a result, different techniques are often used to estimate benefits and costs.[7]

Social benefits analyses evaluate the total expected welfare gains individuals experience resulting from the regulation or policy action. From the perspective of an action that reduces pollution or environmental contaminants, many of these benefits come from improvements in environmental quality. Once the changes in pollution levels or other environmental effects resulting from a policy are estimated, these changes are translated into health outcomes or other relevant outcomes using information provided by risk assessment and other disciplines. Benefits analyses then apply a variety of economic methodologies to estimate the value of these anticipated health improvements and other types of environmental benefits, but it is important to note that even those benefits that cannot be quantified or put into dollar terms should be described in a benefits analysis. Chapter 7 provides details on methods for estimating social benefits. Within a benefits assessment, pollution exposure may increase for some, e.g., emissions of a pollutant other than the one being regulated may increase, or when the policy is deregulatory. Such costs may be presented as negative benefits and may be described as disbenefits or foregone benefits provided that the analysis is internally consistent.

Social cost analyses evaluate the total expected welfare losses experienced by individuals resulting from the regulation or policy action. In most instances, these costs are measured by higher prices for goods and services for consumers and lower earnings for producers and factors of production. Sometimes one modeling effort can be used to estimate both social costs and inputs for benefits analyses, such as predicted changes in pollution from regulated sources. Chapter 8 provides detailed information on methods for estimating social costs. As with benefits, costs that cannot be quantified or put into dollar terms should be described. Also, some costs may decrease due to the regulation. For example, profits may increase for certain related entities or when the action is deregulatory. These outcomes may be presented as negative costs and may be described as avoided costs, again, provided that the analysis is internally consistent. Ultimately, from the perspective of economic theory, the treatment of disbenefits and avoided costs in the analysis is primarily a communications issue and should not affect efficiency analysis and whether net benefits are positive or negative.

7 These Guidelines are organized from the perspective of an action that is designed to achieve health and environmental protection benefits, albeit at some cost. Chapter 7 (Estimating Benefits) therefore focuses primarily on how to evaluate improvements in health and environmental quality, while Chapter 8 (Social Costs) focuses on evaluating the costs associated with actions to achieve those benefits. However, the methods described in these chapters are equally applicable to evaluating decrements in health or environmental quality, and for cost savings if that is appropriate for the policy being evaluated (e.g., for deregulatory actions).

Text Box 1.1 - Agency Checklist for Regulatory Impact Analysis

Does the RIA include a reasonably detailed description of the need for regulatory action?

Does the RIA include an explanation of how the regulatory action will meet that need?

Does the RIA use an appropriate baseline (i.e., best assessment of how the world would look in the absence of the proposed action)?

Is the information in the RIA based on the best reasonably obtainable scientific, technical, and economic information and is it presented in an accurate, clear, complete, and unbiased manner?

Are the data, sources, and methods used in the RIA provided to the public on the internet so that a qualified person can reproduce the analysis?

To the extent feasible, does the RIA quantify and monetize the anticipated benefits from the regulatory action?

To the extent feasible, does the RIA quantify and monetize the anticipated costs?

Does the RIA explain and support a reasoned determination that the benefits of the intended regulation justify its costs (recognizing that some benefits and costs are difficult to quantify)?

Does the RIA assess the potentially effective and reasonably feasible alternatives? Does the RIA assess different regulatory provisions separately if included in the rule?

Does the RIA assess at least one alternative that achieves additional benefits and at least one alternative that costs less?

Does the RIA consider setting different requirements for large and small firms?

Does the selected/finalized option have the highest net benefits (including potential economic, environmental, public health and safety, and other advantages; distributive impacts; and equity), unless a stature requires a different approach?

Does the RIA include an explanation of why the planned regulatory action is preferable to the identified potential alternatives?

Does the RIA use appropriate discount rates for benefits and costs that are expected to occur in the future?

Does the RIA include, if and where relevant, an appropriate uncertainty analysis?

Does the RIA include, if and where relevant, a separate description of distributive impacts and equity?

Does the RIA provide a description/accounting of transfer payments?

Does the RIA analyze relevant effects on disadvantaged or vulnerable populations (e.g., persons with disabilities and low-income groups)?

Does the analysis include a clear, plain language executive summary, including an accounting statement that summarizes the benefit and cost estimates for the regulatory action under consideration, including qualitative and non-monetized benefits and costs?

Does the analysis include a clear and transparent table presenting (to the extent feasible) anticipated benefits and costs (quantitative and qualitative)?

Adapted from OMB’s Agency Checklist: Regulatory Impact Analysis (2010).

1.3.2 Assessing Economic and Distributional Impacts

The assumptions and modeling framework developed for the BCA often do not include or allow for detailed examination of impacts on specific groups. Understanding the nature and magnitude of policy impacts and who will gain or lose from a regulation can be important to policy evaluation, and this requires analyses to supplement BCA.

The EPA addresses economic and distributional impacts of environmental policy through two sets of analyses:

- *Economic Impact Analyses* (EIAs) provide insight into how compliance costs, transfers and other policy outcomes are distributed across groups. EIAs describe and often quantify outcomes such as changes in employment, plant closures or local government tax revenues that provide insight into the economic consequences of regulation. Economic impacts may fall on groups such as industry sectors, small businesses, state or local governments, consumers or workers that may benefit or be harmed by a policy. Chapter 9 provides information on analyzing economic impacts.
- Other analyses evaluate the distribution of changes in environmental risks or health outcomes due to regulation from environmental justice (i.e., on minority, low-income or Indigenous populations) and life stage (i.e., on children, the elderly) perspectives. Consideration of costs may also be relevant in such analyses. Chapter 10 provides information on how to analyze impacts from these perspectives.

1.4 Principles for Conducting Economic Analysis

Many aspects of an economic analysis will vary depending on the purpose, area of focus, available data, and needed level of detail for the analysis. That said, the following are core principles that apply to all economic analyses:

- **Economic analyses should be based on sound economics and science.** Economic analyses should be grounded in well-established economic methods, theory, and principles. The effects considered in BCA, for example, should follow from economic principles and are independent of what is considered in legal or policy analyses, or what may be defined by science policy in other disciplines. Economic analysis should also be flexible enough to incorporate new information and advances in theory and the practice of economics. Economic analyses often rely upon or draw from the tools and results of other scientific analyses. These analyses should also be grounded in the principles, theories, and methods appropriate to their discipline.
- **Economic analyses should be objective and avoid bias.** The goal of the economic analysis is to provide objective information about the consequences of policy decisions. Professional judgments and assumptions are generally required for economic analyses, but these judgments and assumptions should not be based on the preferences of the analyst or policy maker. Economic analyses should seek to capture the expected behavioral responses of households, firms, and governments to incentives and options created by the actual requirements of the regulation or other context being analyzed as accurately as possible. Analyses should be unbiased and should not be framed or performed in a manner to obtain predetermined results or to defend a particular policy decision. In addition, judgments or assumptions should not be made to favor one conclusion over another. For instance, sensitivity analysis can be used to explore a range of possible outcomes but should examine both higher and lower values rather than only one or the other.

- **Economic analyses should be transparent and replicable.** Economic analysis requires choices about data sources, methods, models, and assumptions. The reasons for these choices should be presented explicitly and clearly, along with appropriate justification. Economic analysis should also explicitly acknowledge and characterize important uncertainties in the analysis, state the judgments and decisions associated with these uncertainties, and should identify the implications of these choices. Specific references should be made to all data sources and models, and publicly available data and models should be used to the maximum extent possible. The analysis should provide enough information for readers to clearly see how final empirical estimates and conclusions were reached.

Key Best Practices Covered in the *Guidelines*

Key best practices that apply to all or most economic analyses are also covered in these *Guidelines*. These are listed below along with the chapter in which they are covered:

- Economic analyses produced by the EPA should be responsive to directives from applicable statutes and executive orders (Chapter 2).
- Analyses should describe the economic basis for the policy action and evaluate multiple options to arrive at the most desirable decision (Chapter 3).
- Economics and economic analysis can also inform the consequences of different regulatory designs under consideration, identifying those that are likely to be most cost-effective (Chapter 4).
- The economic impact and consequences of policy must be evaluated relative to some alternative setting, generally one without the policy action. This alternative setting is called the analytic baseline. Specifying a baseline can sometimes be challenging, but it is essential for sound and informative economic analysis. The scope of the analysis should also be clearly defined, and uncertainties in the analysis should be evaluated and characterized (Chapter 5).
- The economic effects of policies typically occur over several years. As such, consistent application of discounting is needed to make these effects comparable (Chapter 6).
- Analysis of benefits and costs should be grounded in sound, well-established economic principles and approaches, should capture all relevant outcomes to the extent possible, and should incorporate advances in the field where warranted (Chapter 7 and Chapter 8).
- Analysis of the distribution of impacts associated with policy decisions should adhere to the same high standards of an economic analysis, should start with the same baselines as the economic analysis, and should provide a balanced accounting of who gains and who loses as a result the policy action (Chapter 9 and 10).
- Finally, an economic analysis must be clearly and effectively communicated for it to be valuable for decision-making (Chapter 11).

Chapter 1 References

OMB. 2023. Circular A-4, Regulatory Analysis, November 9, 2023. Available at: https://www.whitehouse.gov/wp-content/uploads/2023/11/CircularA-4.pdf (accessed June 13, 2024).

OMB. 2010. Agency Checklist: Regulatory Impact Analysis, October 28, 2010. Available at: https://www.whitehouse.gov/wp-content/uploads/legacy_drupal_files/omb/inforeg/inforeg/regpol/RIA_Checklist.pdf (accessed June 13, 2024).

Chapter 2 - Statutory and Executive Directives for Conducting Economic Analyses

Federal agencies are subject to statutes and executive orders (EOs) that direct them to conduct specific types of economic analyses. Many are potentially relevant for all U.S Environmental Protection Agency (EPA) programs; others target individual programs. The scopes of the directives calling for economic analyses vary substantially. In some cases, a statute or EO may be limited in its applicability to those regulatory actions that exceed a specified threshold in significance or impact. To determine whether a regulatory action meets such a threshold and is covered by the statutory or EO provisions, the agency may need to conduct a preliminary economic analysis. Covered regulatory actions may need:

- Economic analysis (e.g., analysis of benefits and costs pursuant to EO 12866, "Regulatory Planning and Review");
- Procedural steps (e.g., consultation with affected state and local governments pursuant to EO 13132, "Federalism"); or
- A combination of both economic analysis and procedural steps.

This chapter identifies directives for conducting economic analyses that may apply to all EPA programs (see Table 2.1) and thresholds that trigger an economic analysis or additional procedural steps for a regulatory action.[1] It also summarizes general provisions calling for economic analyses in selected statutes and EOs and provides direction for analysts seeking guidance on compliance with them. References to applicable Office of Management and Budget (OMB) and EPA guidelines for each EO or statute are provided. For further information about the type and scope of analysis directed, the program's Office of General Counsel (OGC) attorney is a good resource.[2] This chapter does not address provisions of the statutes and EOs that do *not* call for economic analysis.

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1 Although not discussed here, 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.

2 For OGC's reference guide on cross-cutting statutory and EO reviews that may apply to rules, see U.S. EPA (2003b, 2005).

Table 2.1 - Overview of Executive Orders and Statues

Executive Order/Statute	Economic Threshold*	Guidance/ Information Available
EO 12866, Regulatory Planning and Review (1993) as amended by Executive Order 14094, "Modernizing Regulatory Review" (2023)	Specific	EPA, OMB
EO 12898, Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations (1994)	General	EPA
EO 13045, Protection of Children from Environmental Health Risks and Safety Risks (1997)	Specific	EPA
EO 13132, Federalism (1999)	Specific	EPA
EO 13175, Consultation and Coordination with Indian Tribal Governments (2000)	General	EPA, OMB
EO 13211, Actions Concerning Regulations that Significantly Affect Energy Supply, Distribution, or Use (2001)	Specific	OMB
EO 13563, Improving Regulation and Regulatory Review (2011)	Specific	OMB
EO 13707, Using Behavioral Science Insights to Better Serve the American People (2015)	General	White House Memo
EO 14096, Revitalizing Our Nation's Commitment to Environmental Justice for All (2023)	General	EPA
Regulatory Flexibility Act (RFA), as Amended by the Small Business Regulatory Enforcement Fairness Act of 1996 (SBREFA)	Specific	EPA
Unfunded Mandates Reform Act of 1995 (UMRA)	Specific	EPA, OMB
Paperwork Reduction Act of 1995 (PRA)	Specific	EPA, OMB
The Foundations for Evidence-Based Policymaking Act of 2018	None	OMB

** Economic Threshold: "Specific" if EO or statute provides specific numeric threshold or detailed criteria; "General" if EO or statute provides only general description or statement.*

2.1 Executive Orders

2.1.1 Executive Order 12866,3 "Regulatory Planning and Review" as amended by Executive Order 14094, "Modernizing Regulatory Review"

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

1. Have an annual effect on the economy of $200 million or more (adjusted every 3 years by the Administrator of OIRA for changes in gross domestic product); 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, territorial, or tribal governments or communities;[4]
2. Create a serious inconsistency or otherwise interfere with an action taken or planned by another agency;
3. Materially alter the budgetary impact of entitlements, grants, user fees or loan programs or the rights and obligations of recipients thereof; or
4. Raise legal or policy issues for which centralized review would meaningfully further the President's priorities or the principles set forth in this Executive order, as specifically authorized in a timely manner by the Administrator of OIRA in each case.

EO 12866 does not distinguish between regulatory and deregulatory actions. Meeting one or more of the threshold criteria triggers the classification of a regulatory action as "significant." OMB categorizes a regulatory action that meets the first criterion as significant under Section 3(f)(1) of Executive Order 12866 (as amended) (formerly referred to as "economically" significant).[5] The determination of significance under Section 3(f)(1) is multi-faceted. Rules that have an annual effect that meets the $200 million threshold (as adjusted every 3 years) are deemed significant under Section 3(f)(1). OMB clarified that they interpret the EO 12866 threshold as being based on the annual costs, benefits, <u>or</u> transfers of the regulatory action in any one year.[6]

The word "or" is important: $200 million (updated every 3 years) in annual benefits, or costs, or transfers is sufficient to meet the threshold.[7] Note that the threshold determination is not based on net effects, so if any category meets the threshold, the rule would be significant under 3(f)(1). For example, suppose Congress passes a new law that requires the EPA to collect user fees from an industry that manufactures chemicals. The user fees will be used to defray EPA's costs associated with an existing obligation to conduct risk evaluations of new chemicals. Previously, funds to pay the EPA's costs to conduct these evaluations were provided by Congress through its annual congressional appropriation. This new rule requires the EPA to recoup these costs from industry. Assume that the fees to be collected from industry total $220 million per year. In this case, no new

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3 EO 13563, "Improving Regulation and Regulatory Review," issued in January 2011, supplements and reaffirms the provisions of EO 12866. It emphasizes the importance of reducing regulatory costs and burdens and maintaining flexibility and freedom of choice. See Section 2.1.7 in this chapter for more information on EO 13563.

4 EO 14094 increased the 12866 threshold from $100 to $200 million and added the inflation adjustment every three years.

5 See OMB 2023.

6 OMB 2023a.

7 OMB 2023a.

burden is being placed on society. The $220 million is simply a transfer of payments from businesses to government; however, because the transfer is more than $200 million annually, this action is 3(f)(1) significant. By contrast, a rule with $120 million in benefits and $120 million in costs would not be sufficient to meet the dollar threshold under Section 3(f)(1).

In addition, rules that "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, territorial, or tribal governments or communities" are also deemed 3(f)(1) significant. These criteria are independent of the $200 million threshold to trigger the "significant under 3(f)(1)" designation.

It is important to note that meeting the $200 million threshold can include consideration of unquantified effects as well as quantified effects. There may be impacts that are unquantified due to lack of data or valuation methods, but if the judgement of the EPA, or ultimately OMB, is that the combined quantified and unquantified annual effects are likely to exceed $200 million, the regulation would be considered 3(f)(1) significant. OMB clarifies that the threshold determination should also consider effects that may seem "indirect" or "ancillary."[8]

In practice, while the threshold for 3(f)(1) significance is important, the level of analysis can vary. OMB clarifies, "Different regulations may call for different emphases in the analysis, depending on the nature and complexity of the regulatory issues and the sensitivity of the benefit and cost estimates to the key modeling choices."[9]

Per amendments made to EO 12866 by EO 14094, OMB will automatically update the threshold for 3(f)(1) significance every three years, indexed to GDP growth.[10]

Analyses contingent on threshold: Regulatory actions designated “significant” are subject to EO 12866 review by OMB. The process of making this determination is discussed in "EPA's Action Development Process: Guidance for EPA Staff on Developing Quality Actions."[11] For all significant regulatory actions, the agency shall provide to OMB a statement of the need for the regulatory action and an assessment of potential benefits and costs (Section 6(a)(3)(B)). The analysis of benefits and costs increases in complexity and detail for 3(f)(1) significant rules (i.e., those that fall under the definition in the first bullet above). For these rules, the EO directs that, in addition to assessing potential costs and benefits, agencies must include the underlying analysis informing that assessment, quantify benefits and costs to the extent feasible, assess the benefits and costs of potentially effective and reasonably feasible alternative approaches, and provide the underlying analysis of that alternatives assessment (Section 6(a)(3)(C)). OMB's Circular A-4 (discussed below) states that analysts should generally analyze at least three options for each key attribute or provision: the proposed or finalized option; at least one option that achieves additional benefits; and at least one option that costs less.[12]

Guidance: OMB’s Circular A-4 (2023) provides guidance to federal agencies on the development of regulatory analysis for 3(f)(1) significant rules as directed by EO 12866 as well as for other regulatory analysis either required or undertaken at the agency's discretion. Circular A-4 is

8 OMB 2023a.

9 OMB 2023, p. 4.

10 EO 12866 did not provide for an inflation adjustment, resulting in the $100 million threshold becoming more stringent as inflation increased over the years.

11 U.S. EPA 2024.

12 OMB 2023, p. 21.

intended to assist analysts in conducting high-quality and evidence-based regulatory analysis and to standardize the way benefits and costs of federal regulatory actions are measured and reported. Parts of Circular A-4 guidance are standardized for rules that are 3(f)(1) significant. For example, agencies are asked to provide a prominent standardized accounting statement, with one or more tables summarizing costs and benefits (including monetized; quantified, but not monetized; and unquantified), at a standardized consumption discount rate (updated every 3 years) for the main analysis along with reporting of the undiscounted annual stream of benefits and costs.[13] In other respects, OMB notes that "you cannot conduct a good regulatory analysis according to a formula. Conducting high-quality analysis requires competent professional judgment."[14] OMB published additional supporting information in a separate document entitled OMB Circular No. A-4: Explanation and Response to Public Input.[15]

The *Guidelines* provide more in-depth Agency guidance, building on the OMB's guidance with a focus on approaches and methods that are relevant to environmental regulations. Chapters 3 through 8 of this document provide more detailed guidance for fulfilling the EO 12866 benefit-cost analysis provisions, consistent with directions in OMB's Circular A-4. Chapters 9 and 10 provide guidance on addressing distributional effects of environmental regulation, with a focus on economic impact analysis examining compliance costs effects (e.g., profitability, employment, prices) in Chapter 9 and on environmental justice and life stage considerations in Chapter 10.[16]

2.1.2 Executive Order 12898, “Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations” and Executive Order 14096, "Revitalizing Our Nation's Commitment to Environmental Justice for All"

Threshold: No specific threshold; EO 12898 directs each agency, to the greatest extent practicable and permitted by law, to "make achieving environmental justice part of Its mission." EO 14096 supplements EO 12898 and calls on the federal government to "build upon and strengthen its commitment to deliver environmental justice."

Analyses contingent on threshold: EO 12898 directs agencies, to the greatest extent practicable and permitted by law, to "identify[] and address[], . . . disproportionately high and adverse human health or environmental effects of its programs, policies and activities on minority populations and low-income populations."[17] Among other directives and consistent with EO 12898, EO 14096 calls on agencies to, as appropriate and consistent with applicable law, "identify, analyze, and address”:

13 See Chapter 11 of this document, Presentation of Analysis and Results, for agency guidance on presenting economic analysis results.

14 OMB 2023, p. 4.

15 OMB 2023a.

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

17 See EO 12898 Sec. 1-101. EO 14096 uses the phrase “disproportionate and adverse” instead of “disproportionately high and adverse,” as used in EO 12898. According to the White House Fact Sheet on EO 14096, these phrases have the same meaning. Removing the word “high” is intended in to eliminate potential

1. Disproportionate and adverse human health and environmental effects…, including those related to climate change and cumulative impacts of environmental and other burdens on communities with environmental justice concerns;
2. Historical inequities, systemic barriers, or actions related to any Federal regulation, policy, or practice that impair the ability of communities with environmental justice concerns to achieve or maintain a healthy and sustainable environment; and
3. Barriers related to Federal activities that impair the ability of communities with environmental justice concerns to receive equitable access to human health or environmental benefits, including benefits related to natural disaster recovery and climate mitigation, adaptation, and resilience.[18]

Guidance: The EPA's "Technical Guidance for Assessing Environmental Justice in Regulatory Analysis" is designed to outline analytic expectations and discuss technical approaches and methods that can be used by EPA analysts to evaluate the environmental justice (EJ) effects of regulatory actions.[19] This technical guidance is also useful for understanding what role analysis can play in ensuring that EJ concerns are appropriately considered and addressed in the development of regulatory actions, to the extent practicable and permitted by law. Chapter 10 of this document addresses environmental justice analysis, including guidance on considering the distribution of exposure, health outcomes, benefits and/or costs when evaluating impacts on these specific populations.

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

Threshold: Economically significant regulatory actions as described by EO 12866 (now referred to as 3(f)(1) significant, per above) that involve environmental health risk or safety risk that an agency has reason to believe may disproportionately affect children.

Analyses contingent on threshold: An evaluation of the health or safety effects of the planned regulation on children (section 5(a)) or an explanation of why not conducted. The agency shall also provide an explanation of why the planned regulation is preferable to other potentially effective and reasonably feasible alternatives the agency is considering (Section 5(b)).

Guidance: The EPA has prepared guidance to assist EPA staff on the implementation of EO 13045.[20] The EPA's *Children's Health Valuation Handbook* discusses special issues related to estimation of the value of health risk reductions to children.[21] The Office of Children's Health Protection also provides

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misunderstanding that agencies should only be considering large disproportionate effects. See FACT SHEET: President Biden Signs Executive Order to Revitalize Our Nation's Commitment to Environmental Justice for All, The White House (Apr. 21, 2023), https://www.whitehouse.gov/briefing-room/statements-releases/2023/04/21/fact-sheet-president-biden-signsexecutive-order-to-revitalize-our-nations-commitment-to-environmental-justice-for-all/. EO 14096 also includes a definition of "environmental justice" which expands on the demographic categories laid out in EO 12898, such as by including Tribal affiliation and individuals with disabilities. See EO 14096 Sec. 2(b).

18 See EO 14096 Sec. 3(a)(i), (iii), and (iv).

19 U.S. EPA 2016.

20 U.S. EPA 2024.

21 U.S. EPA 2003a.

online information with links to resource materials on guidance and tools.[22] Guidance in Chapter 10 of this document addresses analyses of impacts on children.

2.1.4 Executive Order 13132, "Federalism"

Threshold: Rules that have "federalism implications" that either impose substantial compliance costs on state and local governments or preempt state or local law. According to EPA policy, rules are considered to impose substantial compliance costs if:

The action is likely to result in the expenditure by state and local governments, in the aggregate, of $25 million or more in any one year; or

The action is likely to result in expenditures by small governments that equal or exceed 1% of their annual revenues. [23]

Exception: An action that imposes substantial compliance costs (meets the $25 million threshold or the 1% test) does not have a federalism implication if: (1) the action is expressly required by statute (without any discretion by the EPA); or (2) there are federal funds available to cover the compliance costs.

Analyses contingent on threshold: For actions with federalism implications, agencies shall conduct pre-proposal consultation with elected state/local officials or their representative national organizations. Rules must include a Federalism Summary Impact Statement in the preamble, and a signed Federalism Certification from the Agency's designated official should be provided to OMB for rules subject to OMB review under EO 12866 along with any written communications that the EPA received from state or local officials.

Guidance: Specific guidance on EO 13132 can be found in the internal EPA document "Guidance on Executive Order 13132: Federalism".[24]

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

Threshold: Regulations that have substantial direct effects on one or more American Indian tribe, on the relationship between the federal government and tribes, or on the distribution of power and responsibilities between the federal government and tribes and that: (1) impose substantial direct compliance costs on Indian tribal governments that are not required by statute, or (2) preempt tribal law.

Analyses contingent on threshold: To the extent practicable and permitted by law, EO 13175 directs the Agency to either provide the funds necessary to pay the Tribal governments' direct compliance costs, if applicable, or prior to the formal promulgation of the regulation, to (1) consult with Tribal officials early in the process of developing the proposed regulation; (2) make any written communications submitted to the Agency by Tribal officials available to the Director of OMB; and (3) include in the preamble of the regulation a Tribal Summary Impact Statement. The

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22 See https://www.epa.gov/children/guidance-tools-and-glossary-key-terms (accessed July 31, 2024).

23 U.S. EPA 2008.

24 U.S. EPA 2008.

Statement should include a description of the extent of the Agency's prior consultation with Tribal governments; a summary of the nature of the Tribe's concerns and the Agency's position supporting the need to issue the regulation; and a statement of the extent to which the concerns of Tribal governments have been met.

Guidance: OMB issued Guidance for Implementing EO 13175 in 2010 to provide direction for compliance and documentation,[25] and the White House issued Presidential Memoranda in 2009, 2021 and 2022 to support implementation of EO 13175.[26] The 2021 Presidential Memo (Tribal Consultation and Strengthening Nation-to-Nation Relationships) reaffirms the policy in the 2009 Presidential Memo (Tribal Consultation) and directs agencies to submit detailed plans of action to implement the policies and directives. The 2022 Presidential Memo (Uniform Standards for Tribal Consultation) establishes uniform minimum standards to be implemented across all agencies regarding how Tribal consultations are to be conducted. The EPA updated its Policy on Consultation and Coordination with Indian Tribes in 2023 to establish national guidelines and institutional controls for consultation across the EPA. This policy states, "The U.S. Environmental Protection Agency's policy is to consult on a government-to-government basis with federally recognized Tribal governments when EPA actions or decisions *may* affect Tribes." [emphasis added].[27] Chapter 10 of this document addresses environmental justice analyses focusing on people of color, low-income populations, and/or Indigenous populations.

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

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

Analyses contingent on threshold: Submission of a detailed Statement of Energy Effects to OMB. The Statement of Energy Effects must address any expected adverse effects on energy supply, distribution or use, the reasonable alternatives to the action, and the expected effects of such alternatives on energy supply, distribution, and use.

Guidance: OMB Issued Memoranda in 2001 (M-01-27 Guidance for Implementing EO 13211) and 2021 (M-21-12 on Furthering Compliance with Executive Order 13211).[28] M-21-12 affirms and amends M-01-27 to reflect changes in market conditions since 2001 with additional examples of qualifying "adverse effects" of regulatory actions.

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25 OMB 2010.

26 The White House 2021 and 2022.

27 U.S. EPA 2023, p. 1.

28 OMB 2001 and OMB 2021.

2.1.7 Executive Order 13563, "Improving Regulation and Regulatory Review"

Threshold: Significant regulatory actions under EO 12866 as amended by EO 14094 (reaffirms EO 12866 and includes additional provisions).[29]

Analyses contingent on threshold: As mentioned, EO 13563 supplements and reaffirms the provisions of EO 12866 (as amended). EO 13563 states, "Our regulatory system must protect public health, welfare, safety, and our environment while promoting economic growth, innovation, competitiveness, and job creation." It emphasizes the importance of reducing regulatory costs and burdens and maintaining flexibility and freedom of choice. The EO highlights the importance of scientific integrity, and retrospective analyses of existing rules.

Among other directives, agencies must use best available techniques to quantify costs and benefits, give the public meaningful opportunity to comment online, include relevant scientific and technical findings in the rulemaking docket, consider the combined effects of their regulations on particular sectors and industries and promote coordination across agencies. With regard to existing regulations, EO 13563 instructs agencies to periodically review their significant regulations with the goal of making their regulatory programs more effective or less burdensome. Per OMB guidance, agencies are particularly encouraged to identify actions for review that will significantly reduce existing regulatory burdens and promote economic growth and job creation. Chapter 5 includes a discussion of retrospective review and analysis; see Text Box 5.1 on Retrospective Analysis.

Guidance: OMB issued implementation guidance in three memos: M-11-10 February 2, 2011; M-11-19 April 25; 2011; M-11-25 June 14, 2011.[30]

2.1.8 Executive Order 13707, "Using Behavioral Science Insights to Better Serve the American People"

Threshold: No specific threshold; the EO encourages agencies to "identify policies, programs, and operations where applying behavioral science insights may yield substantial improvements in public welfare, program outcomes, and program cost effectiveness…"

Analyses contingent on threshold: Agencies are encouraged to use behavioral science insights when designing policies and specifically when determining access to programs, presenting Information to the public, structuring choices within programs and designing incentives.

Guidance: The White House Social and Behavioral Sciences Team issued implementation guidance in a memo on September 15, 2016.[31] Chapter 4 of this document includes a discussion of behavioral economics.

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29 OMB 2011a.

30 See EO 13563 and OMB 2011a, 2011b, 2011c.

31 Executive Office of the President, Office of Science and Technology Policy. 2016.

2.2 Statutes

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

Threshold: Regulations that may have a "significant economic impact on a substantial number of small entities," (SISNOSE), including small businesses, governments and non-profit organizations. The RFA does not define the terms **significant** or **substantial**.

Analyses contingent on threshold: For rules that may have a SISNOSE, agencies are required to prepare an initial regulatory flexibility analysis (IRFA) and a final regulatory flexibility analysis (FRFA) examining potential adverse economic impacts on small entities and complying with a number of procedural and analytical requirements to solicit and consider flexible regulatory options that minimize adverse economic impacts on small entities and address significant issues raised in public comments. The IRFA and FRFA, or summaries thereof, are published with the proposed and final rules, respectively,

Guidance: The EPA has issued specific guidance for complying with RFA/SBREFA requirements in the
"EPA Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act".[32] The guidance identifies approaches for determining whether a specific rule may have a SISNOSE but provides flexibility to use alternative methods or reach different conclusions where appropriate in the context of a specific rule. See also Chapter 9 of this document on economic impact analysis.

2.2.2 Unfunded Mandates Reform Act (UMRA) (2 U.S.C 48 P.L. 104-4)

Threshold one (Sections 202 and 205 of UMRA): Regulatory actions that include federal mandates "that may result in the expenditure by State, local, and tribal governments, in the aggregate, or by the private sector, of $100 million or more (adjusted annually for inflation) in any one year."[33] An action contains a federal mandate if it imposes an enforceable duty on state, local or tribal governments or the private sector.

Analyses contingent on threshold one: Section 202 of UMRA requires preparation of a written statement that includes the legal authority for the action; a BCA; a distributional analysis; estimates of macroeconomic impacts; a description of an agency's pre-proposal consultation with elected representatives of the affected state, local or tribal governments; and a summary of concerns raised and how they were addressed. Section 205 of UMRA requires an agency to consider a reasonable number of regulatory alternatives and select the least costly, most cost-effective, or least burdensome alternative that achieves the objectives of the rule, or to publish with the final rule an explanation from the agency head of why such alternative was not chosen.

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32 U.S. EPA 2006a

33 Note that the threshold in this case is adjusted annually for inflation (since enactment). Generally, the EPA uses the U.S. Bureau of Economic Analysis gross domestic product implicit price deflator to adjust the $100 million UMRA threshold for inflation each year (e.g., the UMRA threshold was $186 million in 2024$). Note that EO 14094 increased the 12866 threshold from $100 to $200 million and added an inflation adjustment every three years.

OMB's Circular A-4 (2023) notes that the analytical concepts under EO 12866 are similar to the analytical concepts under UMRA, and "an analysis produced pursuant to Executive Order 12866 will usually satisfy the analytic requirements for a written statement under the Unfunded Mandates Reform Act."

Threshold two (Section 203 of UMRA): Regulatory requirements that might "significantly" or "uniquely" affect small governments. Small governments include governments of cities, counties, towns, townships, villages, school districts or special districts with a population of less than 50,000.

Requirements contingent on threshold two: Agencies must solicit involvement from, and conduct outreach to, potentially affected elected officers of small governments (or their designated employees) during development and implementation.

Guidance: The EPA has issued "Interim Guidance on the Unfunded Mandates Reform Act of 1995" 1995), and OMB issued a memo on "Guidance for Implementing Title II of S.1" that provides general guidance on complying with requirements contingent on each of the two thresholds under UMRA.[34]

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

Threshold: Any action that requires or requests record-keeping, reporting or disclosure or includes other information collection activities imposed upon or posed to 10 or more persons,[35] other than federal agency employees.

Requirements contingent on threshold: The agency must submit an information collection request (ICR) to OMB for review and approval and meet other procedural requirements including public notice and opportunity for comment. The ICR should: (1) describe the information to be collected, (2) give the reason the information is needed and (3) estimate the time and cost for the public to respond to the request.

Guidance: Both guidance and templates for completing an ICR and associated Federal Register (FR) notices can be found on the EPA's intranet site, "ICR Center."[36]

2.2.4 The Foundations for Evidence-Based Policymaking Act (5 U.S.C. 101 P.L. 115-435)

Threshold: No specific threshold.

Requirements contingent on threshold: The Foundations for Evidence-Based Policymaking Act of 2018 ("Evidence Act"), mandates federal evidence-building activities, where evidence is broadly defined and includes foundational fact finding, performance measurement, policy analysis and program evaluation.[37] The act does not specify what evidence-building activities agencies should conduct but instead calls on agencies to significantly rethink how they currently plan and organize

34 U.S. EPA 1995 and OMB 1995

35 Exceptions include "listening sessions with interested parties; asking non-standardized questions on a particular process, theme, or issue…; directly observing the experiences of program applicants and participants." (OMB 2022).

36 See https://work.epa.gov/icr (accessed August 1, 2024, internal EPA website).

37 OMB 2019.

evidence building, data management and data access functions to ensure they have the evidence they need for informed decision making. Prospective and retrospective economic analyses of agency programs and regulations are evidence-building activities under the Evidence Act and data used or produced in economic analyses may be subject to Title II of the Evidence Act (the Open Government Data Act), including the requirement of being open by default.

Guidance: In July 2019, OMB issued a memorandum on Phase 1 Implementation of the Foundations for Evidence-Based Policymaking Act of 2018: Learning Agendas, Personnel and Planning Guidance. OMB notes that in their annual evaluation plans, "agencies should also discuss any evaluation activities that relate to its proposed regulatory actions in the Unified Agenda of Federal Regulatory and Deregulatory Actions, recognizing that these activities often need to occur well before the development of economically significant regulatory actions".[38]

38 OMB 2019, p. 34. In 2021, OMB Issued a guidance memorandum on "Evidence-Based Policymaking: Learning Agendas and Annual Evaluation Plans" (see OMB 2021a) that reaffirms and expands on previous OMB guidance on Learning Agendas and Annual Evaluation Plans, including OMB M-19-23.

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5 U.S.C. 601-612: The Regulatory Flexibility Act (P.L. 96-354), as amended by the Small Business Enforcement Fairness Act (P.L. 104-121), March 29, 1996. Available at: https://www.govinfo.gov/content/pkg/PLAW-104publ121/pdf/PLAW-104publ121.pdf (see page 11, Title II) (accessed October 23, 2024).

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Chapter 3 - Need for Regulatory Action and Evaluation of Policy Options

The essential components of an economic analysis are (1) a clear statement of the need for regulatory action describing the problem to be addressed by the policy and (2) a detailed evaluation of policy options. The statement of need should include a description of the market, institutional, or behavioral distortions being addressed, an explanation of why the market and other institutions have failed to correct these problems, and a justification for federal action to address them.

The economic analysis should consider and evaluate multiple policy options that address the environmental problem. This is true for analyses of proposed and final rules, even when the Agency has settled on a specific option. When identifying policy options, the analysis should describe any statutory or judicial requirements that must be considered. The options should include those permissible under the relevant statutory authority and may include those that are unavailable but with other advantages. The options may differ in their levels of stringency, compliance dates, and requirements based on entity size and location, or they may represent entirely different regulatory approaches. Detailing possible options is a necessary step in establishing why the selected option is the appropriate choice.

3.1 The Statement of Need

Consistent with Executive Order (EO) 12866 and Office of Management and Budget (OMB) Circular A-4 (2023), each economic analysis should include a statement of need that provides: (1) a clear description of the problem being addressed and the significance of that problem, (2) the failures of private markets or public institutions that warrant agency action, and (3) an assessment of whether Federal regulation is the best way to correct the problem.[1] This statement sets the stage for the

1 EO 12866 states, "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). The Office of Management and Budget's guidance for how to comply with EO 12866, Circular A-4 (OMB 2023), provides recommendations to federal agencies on the development of economic analyses supporting regulatory actions. OMB (2023, p. 14) states that "including a summary in regulatory analyses of the needs being addressed may provide useful background and help ensure that the description of the needs informs the scope of the analyses (and vice versa) to the extent relevant, appropriate, and consistent with the best available evidence and best practices for objective analysis."

subsequent benefit-cost analysis (BCA) and allows one to judge whether the policy adequately addresses the problem.

3.1.1 Problem Description

The statement of need should begin with a brief review of the problem or public need to be addressed by the policy. While not always the case, the compelling public need for U.S Environmental Protection Agency (EPA) regulations is generally to address an environmental problem. In this case, the following considerations are often relevant:

- The primary environmental contaminants causing the problem and their concentrations.
- 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.
- The expected change in the environmental problem over time, absent additional regulation.
- Available and potential abatement and mitigation techniques and technologies.
- The amount or proportion of the environmental problem likely to be corrected by federal action.
- Any existing state, local and other federal activities that partially or fully address the problem.

3.1.2 Reasons for Regulatory Action

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. That is, it should define the reason or social purpose for the regulatory action. This identification is an important component of policy development because the underlying failure itself often suggests the most appropriate remedy for the problem (see Chapter 4). A regulation can be promulgated for a number of social purposes. For pollution problems, the social purpose is commonly to correct a "market failure." Other potential reasons for regulatory action include addressing behavioral biases; improving the efficiency and effectiveness of government operations; promoting distributional fairness and advancing equity; and protecting civil rights and civil liberties.

3.1.2.1 Market Failure

A market failure occurs when the allocation of goods and services by the free market is not economically efficient. The most common causes of market failure are externalities, overutilization of common property resources, under-provision of public goods, market power, and inadequate or asymmetric information.[2] While there are other social purposes for government regulation,

2 For further discussion of market failure, types of market failures and externalities see Scitovsky (1954), Bator (1958), Buchanan and Stubblebine (1962), Mishan (1969), Baumol and Oates (1988), Cornes and Sandler (1996), Hanley et al. (2019), Perman et al. (2003), and Tietenberg and Lewis (2014). OMB (2023) also describes different categories of market failure as well as other reasons for regulation. Section A-2 of these Guidelines provides further discussion of externalities.

correcting a market failure, particularly addressing an externality, is most likely the driver behind environmental policy.

As defined by Keohane and Olmstead (2016), "An externality results when the actions of one individual (or firm) have a direct, unintentional, and uncompensated effect on the well-being of other individuals or the profits of other firms."[3] Technically, externalities occur when the outputs and inputs chosen by one individual enter the utility or production function of another without passing through markets or contracts. Put another way, externalities occur when the market does not account for the effect of one party's activities on another party's well-being without compensation.

Consider, for example, a factory that produces smoke as a by-product of manufacturing that, in turn, affects individuals living downwind. The factory does not weigh the costs of its actions on the downwind community when making production decisions. Although the factory imposes an externality on the downwind community, the mere existence of an externality is not enough to justify a regulation. Under certain conditions, namely, the ability to bargain, availability of complete information, and presence of low transaction costs, externalities can be internalized by the free market (Coase 1960). Text Box 3.1 describes this Coasian solution in more detail.

It is important to differentiate externalities from other external effects when an individual or firm is affected by the behavior of others. For example, a negative outcome caused by another individual is not an externality if the affected individual rationally and willingly accepts the risk of that outcome through a private transaction between them. This may occur when a worker accepts a job with a greater risk of injury in exchange for a higher wage. However, this assumes complete and perfect markets with full information and that the transaction stipulations reflect and incorporate the expected risk such that no externality is associated with increased risk of injury. Similarly, external effects that function through the price system (e.g., higher prices faced by certain consumers because of rising demand) or zero-sum transfers from one person to another (e.g., through taxes or redistribution of consumer and producer surplus) are not externalities by definition and do not constitute a market failure. For example, if person A outbids person B in an auction, person B may be made worse off than if they had won the auction but were unwilling to pay the higher bid. This is a result of the price system working to ensure scarce resources go to those willing to pay the most for them, avoiding an inefficient allocation of resources.[4]

3 Keohane and Olmstead (2016) go on to say, "Note three keywords in the definition: direct, unintentional, and uncompensated. For example, because your health and happiness depend in part on how clean the air is, automobile drivers have a direct effect on your well-being. Unintentional is included in the definition to rule out acts of spite or malice. (It is the effect rather than the action that is unintentional. I may decide deliberately to use a gasoline-powered lawnmower, without the intent of my action being to pollute the air or disturb the neighbors.) Finally, uncompensated implies that the responsible actor does not compensate the damaged parties (or is not fined) for his actions. This rules out market transactions or bargaining between individuals" [emphasis in original].

4 External effects operating through the price system are referred to as pecuniary externalities.

Text Box 3.1 - Coasian Solution

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

When property rights can be defined and have been allocated, a welfare-maximizing solution can be reached regardless of which party is assigned the property rights, although the distribution of the gains from bargaining will differ. Take for example a farm whose pesticide application to its crops pollutes the well water of nearby homeowners. If property rights of the watershed are assigned to the homeowners, and information is available to them about potential damages from the pollution, then the farm may negotiate with the homeowners about its continued use of the pesticide. Potential compensation from the farm to the homeowners agreed upon through such negotiations need not be in the form of cash but could involve investments to reduce the water contamination or land swaps. (e.g., Deryugina et al. 2021), If property rights of the watershed are given to the farm, then the homeowners could negotiate to pay the farm to stop applying the pesticide.

The effectiveness of such agreements is contingent on meeting additional conditions: bargaining must be possible, damages must be known, and transaction costs must be low. These conditions are more likely to be met when there are only a small number of individuals involved. If either party is unwilling to negotiate or faces high transaction costs, then no private agreement will be reached. Asymmetric information or bargaining power can also hinder a socially optimal solution. Going back to the example, consider a case where there are many farms in the watershed using the pesticide on their crops, and it may be difficult to identify the relative contribution of each farm's effluent on damages experienced by the homeowner. Clearly, homeowners would have more difficulty in negotiating an agreement with many farms than they would in negotiating with a single farm. However, technological advances in data sharing and networking can increase the likelihood of finding a Coasian solution. Advances in internet search and the availability of monitoring devices that can lower transactions costs and reduce information asymmetries, and social networks can make it easier for groups to communicate and arrive at bargained solutions. Deryugina et al. (2021) discusses several Coasian solutions to actual environmental problems.

When left unaddressed, externalities prevent the market from achieving economic efficiency and reduce economic welfare. This can occur in the presence of high transaction costs that make it difficult for private parties to internalize the cost of damages through bargaining, legal action, or other means such that both parties are no worse off. High transaction costs may result when activities that pose environmental risks are difficult to link to the resulting damages because they occur over long periods or occur in a different location than where the pollution originates.[5] If these

5 The concept of an externality is closely tied to the concept of a public good, which is a good that either can be used simultaneously by many (i.e., nonrival) or that is difficult to prevent others from using (i.e., nonexcludable). The environment is a classic example of a public good.

high transaction costs are overcome and the parties can internalize the cost of the damage, then scarce resources will again be efficiently allocated by the market. If the cost of damages cannot be internalized, then government intervention may be necessary to fully address the externality.[6]

But even the presence of an unaddressed externality is not enough to justify a regulation; what is required is a compelling need for government intervention at any level of government (federal, state or local). That is, there must be some form of evidence that government intervention can improve economic welfare.[7] Government regulation may not be warranted if the benefits of regulation do not justify the costs. Circumstances where this may occur include when a regulation designed to reduce a negative externality (e.g., direct emission controls) exacerbates pre-existing distortions. In this case, government intervention could make things worse.

There should also be some evidence that the externality will persist. If the market will correct itself through innovation and technological change or the externality will cease to exist through private transactions, then government intervention may not be necessary. A BCA can determine whether government intervention to remove the externality can improve economic efficiency even if the externality only exists for a short time absent additional regulation (i.e., it is resolved in the baseline after the short time). Furthermore, even if an externality warrants government intervention, it may not warrant direct, prescriptive regulation. Some externalities may be addressed more efficiently through other means such as providing information, requiring firms to carry insurance, defining legal liability, or assigning property rights. The nature of the externality may determine the best approach for government action (see Chapter 4).

3.1.2.2 Other Social Purposes for Regulatory Action

While correcting a market failure, particularly an externality, is the most common justification for environmental regulation, there are other underlying institutional or behavioral distortions that may justify regulatory action or government intervention. These include addressing behavioral biases; improving the efficiency and effectiveness of government operations, promoting distributional fairness and advancing equity; and protecting civil rights and civil liberties. Additionally, regulation may be justified for multiple interconnected reasons, such as addressing a market failure and promoting distributional fairness.

If the social purpose of a regulation is not to address a market failure (e.g., to improve Agency processes or solely to define a statutory term), then the statement of need still should include a description of the problem being addressed and an explanation of why government action is necessary to address this problem. If the purpose of a regulation is to protect sensitive subpopulations or address other distributional impacts rather than, or in addition to, addressing a market failure, that should be stated in the statement of need.

One possible social purpose is addressing behavioral biases. The behavioral economics literature has documented situations in which individuals appear to act in ways that are inconsistent with

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6 As shown in Section A-5, there is an optimal level at which an externality should be addressed by a regulation. At this optimal level, further reduction in the externality is inefficient. Therefore, in the simple case where there is only one externality and it is controlled by an existing regulation, the existing regulation is not sufficiently stringent if the additional benefits from reducing the externality further will exceed the additional cost, and therefore additional regulation would be net-beneficial. Similarly, an existing regulation may be too stringent such that additional regulation would lead to negative net benefits.

7 Lusk (2013) provides a useful nine-point checklist for externalities that require prescriptive regulation.

rational choice, sometime referred to as "behavioral failures" or "behavioral anomalies" (Shogren and Taylor 2008). In such situations, it is possible that government intervention could lead to a more efficient allocation of resources than the free market outcome. However, because the mission of EPA is to protect human health and the environment, behavioral failure absent an environmental externality is not a typical justification for regulation at EPA. If insights from behavioral economics are used as a justification for regulation, analysts should provide robust empirical evidence supporting the existence of behavioral anomalies in the affected market and rules out other explanations consistent with rational behavior, such as hidden costs. Chapter 4 includes more discussion of behavioral economics and its implication for policy design.

3.1.3 Need for Federal Action

The final component of the statement of need for the regulatory action is an evaluation and explanation 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.[8] Federal involvement is often required for environmental problems that cross jurisdictional boundaries (e.g., when pollution in one state affects the population of another). In some cases, federal involvement is mandated by statute or directed by an EO as described in Chapter 2. A federal regulation could be justified by comparing its expected performance to realistic alternatives that rely on other institutional arrangements. This component of the statement of need for regulatory action, justifying federal regulation, should verify that the policy action is necessary, within the jurisdiction of the relevant statutory authorities, and yields results that will be preferable to no action. Finally, the statement of need should identify those aspects of the regulation necessitated by statutory requirements and those that are discretionary.

3.2 General Guidance on Policy Options to Evaluate

3.2.1 Need to Assess Multiple Options

Each analysis should evaluate multiple policy options. Following the statement of need, the economic analysis should identify and describe in detail all policy options or potential regulatory alternatives that were considered. This includes clearly explaining which options were selected for emphasis and further analysis and why other important options were not. Since the BCA informs the public, stakeholders and Congress and other decision makers of the effects of the policy assessing a robust set of policy options is important.

The identification of policy options should describe any statutory or judicial requirements that must be considered when designing the regulation, how those requirements may influence the

8 As discussed in Chapter 2, EO 13132, "Federalism," describes principles of federalism and identifies requirements for federal preemption of state or local law. Also, there is a robust economics literature on the pros and cons of regulating environmental quality at different jurisdictional levels that may be informative when determining whether federal regulation is appropriate as a substitute or complement to state or local regulation (e.g., Oates 2002). See also Circular A-4 (OMB 2023) on "Showing Whether Federal Regulation Is the Best Way to Solve the Problem."

options considered and how the proposed or finalized option satisfies them.[9] For example, the description should identify any economic considerations (e.g., costs incurred by regulated entities) and discretionary provisions in the statute that may be used to shape the form and stringency of the regulation. The analysis may also identify options that are more efficient or cost-effective even if the regulatory approaches may be prohibited by statutory or judicial requirements (see also OMB 2023). For example, the Supreme Court has held that the Clean Air Act requires that National Ambient Air Quality Standards be set based on health or welfare considerations only; the Act bars EPA from considering the costs of implementing them when setting the standards.

At a minimum, the economic analysis should fully assess and present three options for consideration: the proposed or finalized option; a more stringent option; and a less stringent one.[10,11] The incremental benefits and costs for each option, as well as other important criteria (e.g., distributional consequences), should be compared across the options. Measuring the incremental benefits and costs of successively more stringent regulatory options provides a clear indication of the most economically efficient option, provided important benefits and costs can be quantified and monetized. If options cannot be characterized by regulatory stringency (e.g., they differ by the provisions included), the economic analysis should still analyze at least three options, including one that achieves greater benefits and one that costs less than the proposed or finalized option (see also OMB 2023).

Assessing at least three options applies in any circumstance. It is not adequate to evaluate only the selected option, even for a final rule that establishes the option to be promulgated. Similarly, in cases where the design of the regulation is dictated by statute, presenting multiple options is still necessary when the regulation is proposed or finalized -- even though the Agency may have no discretion in its design. Assessing multiple options helps inform the public about the anticipated benefits and costs of the Agency's final action compared to options not pursued, it is imperative that the analysis assesses multiple options.

The analysis should also consider whether there are alternatives to federal regulation that may address the market failure or other regulatory objective (e.g., distributional concern) more efficiently. Alternatives may include using existing product liability rules to encourage firms to internalize the costs of the environmental damages, introducing market-oriented approaches such as fees, penalties, subsidies, marketable permits, and offsets, or the potential for state or local regulation. Even if options are not available due to statutory restrictions, the economic analysis should discuss the limitations of the statutory requirements and, if possible, estimate the

9 Often, consideration of different regulatory options is required or encouraged by statute (e.g., different stringencies of emissions standards). Any qualitative or quantitative analysis that supports these considerations should be summarized in the BCA, even if estimates of the benefits and costs of those options were not produced.

10 An exception may occur if the proposed or finalized option is at or near the limit of technical feasibility, in which case the analysis might not need to examine a more stringent option. However, it is possible that even if abatement of an environmental contaminant using on-site controls is technically infeasible, the value of the good or activity whose production creates the contaminant may be less than the harm the contaminant causes. In such circumstances, a more stringent option that shifts production away from the good or activity should be evaluated.

11 While developing a regulation, the decision maker may choose the more stringent or less stringent option after weighing the results of the analysis. Doing so demonstrates the usefulness of the analysis. In this circumstance, the analysis should include an additional option to satisfy this guidance if time allows. If there is insufficient time to evaluate an additional option, the other two options should still be presented, and the analysis should explain why the central option was not selected.

opportunity cost of not being allowed to pursue these options. There is no prohibition against analyzing these options.[12]

When a rule includes several distinct regulatory provisions, the benefits and costs of each provision should be analyzed both separately and jointly (i.e., as a package of provisions).[13] Doing so may yield insights such as identifying unnecessary or otherwise undesirable regulatory requirements. For example, evaluating provisions independently may identify those provisions for which their costs exceed their benefits, even when the benefits of a regulation in its entirety exceed its costs. Jointly analyzing multiple provisions becomes more complicated when the existence of one provision affects the benefits or costs arising from another. Even so, it is still possible to evaluate a specific provision by estimating the net benefits of a regulatory option with and without that provision.

Ultimately, the number of options to evaluate and their design is a matter of judgment, but the analysis should strive for a balance between thoroughness and analytic capacity. Realistically, analyzing all possible combinations of provisions is impractical if their number is large and interactions between provisions are common. Generally, some options can be eliminated through a preliminary and less rigorous analysis, leaving a more manageable number to be evaluated in the formal BCA. For a proposed rule, it may be useful to provide an economic analysis that illuminates important tradeoffs associated with key specific aspects of the rule on which the Agency is soliciting comment.

3.2.2 Policy Design Options

The analysis should carefully describe the policy design being evaluated and, when the costs or benefits vary substantially with alternative policy designs, assess alternative design options.[14] The policy design includes the core regulatory approach as well as key features of its implementation and structure. Prescriptive regulation (e.g., technology, design, or performance standards) is common in Federal environmental regulations. Performance standards, which specify the allowable limit but not the way regulated entities must achieve that limit, are generally less costly than standards that dictate technologies or techniques. Economic analyses may include assessments of policy designs that currently are not statutorily allowed to highlight potential tradeoffs between the required approach and other more desirable approaches (for example, more flexible market-based approaches such as emissions taxes and allowance trading systems that may be prohibited).

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12 OMB Circular A-4 (2023) states, "Your analysis of the effects of the regulation should not presuppose that there is a need for the regulation, and your analysis of the potential need for the regulation should not presuppose the effectiveness of your regulation." (p.14) and "If legal or other constraints prevent the selection of a regulatory action that best satisfies the philosophy and principles of Executive Orders 12866, you may consider identifying these constraints and estimating their opportunity cost (and effects more generally). Such information may, for example, be useful to Congress under the Regulatory Right-to-Know Act or in considering statutory reforms." (pp.22-23)

13 When the benefits or costs of a regulation or one of its provisions are highly uncertain, an option may include a voluntary program or pilot project or additional data collection prior to regulation. See Chapter 4 for further discussion of these options.

14 Chapter 4 provides a detailed description of different regulatory approaches, including a detailed discussion of considerations for selecting among different regulatory approaches.

Aspects of the market failure may help identify which types of regulatory approaches to consider. A key principle in the design of environmental regulations is that the regulatory structure and incentives should align with the environmental objective. For example, if the effect of emissions on human health depends on the proximity to the emission, then generally the optimal regulation should more stringently control emissions from emitters that are closer to population centers. Another example is that regulations should impose requirements on emissions rather than the inputs associated with the emissions provided emissions monitoring costs are not too high relative to the costs of monitoring input use.

Evaluating regulatory features other than stringency and regulatory approach may also help identify better policy designs. Options that vary these regulatory features, both alone and in combination, should be considered (see also OMB 2023). These features include the entities that are subject to the regulation.[15] By varying policy design features in the options considered, the analysis may identify approaches that increase net benefits or reduce the impact on certain groups. These features include but are not limited to:

- **Compliance dates:** Providing more time before a regulation takes effect may reduce costs by allowing the regulated entities additional planning time, which can be weighed against a possible reduction in benefits.
- **Enforcement methods:** Alternatives include regular on-site inspections, random monitoring, periodic reporting and noncompliance penalties, which may have different costs and efficacy.
- **Requirements for different-sized firms or facilities:** In some cases, small firms or facilities may face proportionately higher compliance costs, especially if there are large, fixed compliance costs.[16] When a market-based approach cannot be used, varying the regulatory stringency or pollution requirement by firm size may increase economic efficiency.
- **Requirements for different geographic regions:** Differentiating requirements by region may be desirable if there is significant regional variation in pollution reduction benefits or the costs of compliance.
- **Requirement for facilities of different vintages:** New facilities may face lower costs of compliance than older facilities because of the relative ease with which abatement methods can be integrated into their production processes. Also, pollution control investments may be in use longer at new facilities, and therefore may yield greater benefits over time.

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15 The coverage of a regulation may include different market sectors or different entities within a sector. Generally, the statutes that EPA implements identify the groups of similar emitting sources that would be subject to a particular regulation, although there is often some flexibility in defining the types of entities included in each group, the requirements for different subgroups and some regulatory choices may influence subsequent requirements for multiple sectors.

16 Chapter 2 describes analysis for examining potential adverse economic impacts on small entities and procedures to solicit and consider flexible regulatory options that minimize adverse economic impacts on small entities under 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). These are required for rules with a "significant economic impact on a substantial number of small entities." Chapter 9 outlines the analytic tasks associated with complying with the RFA.

However, stricter requirements for new facilities than old ones may lead to inefficient investment patterns (e.g., firms delaying investments to avoid stricter regulation).[17]

It is important to account for and present both the total benefits and costs of each option and the incremental benefits and costs among the options. As discussed in depth in Chapter 5, it is important to account for all of the benefits and costs for all policy options because any options where benefits exceed costs is an improvement in economic efficiency according to the potential Pareto principle.[18,19] By this standard, selecting any option with positive net benefits would improve societal welfare. However, the *most* economically efficient option is the one that produces the largest increase in net benefits. While the option with the highest net benefits is obvious from the presentation of total benefits and costs, presenting the incremental benefits and costs of each option compared to the next less-stringent alternative helps to indicate if there is an even more economically efficient option other than those being considered. In general, economic efficiency is maximized (i.e., net benefits are highest) when incremental benefits are equal to incremental costs.[20]

Determining which option is the most economically efficient may be more challenging when there are consequences that society would be willing to pay for (or avoid) but that cannot be quantified or monetized. As discussed In Chapter 5 and elsewhere in these Guidelines, effects should be quantified, even if they cannot be monetized, and discussed qualitatively if not. Differences between consequences that are not quantified or monetized should also be compared among policy options. In particular, different policy options may have different distributional impacts, even without significantly changing the benefits and costs of the regulation, and this difference may not be obvious when only evaluating the total costs and benefits. It may be important to consider which regulatory alternatives may generate important differences in distributional effects.

Furthermore, carefully detailing the sources of the benefits and costs of a rule, rather than looking only at its total net benefits, may help identify other policy options. For example, a regulation that is designed to reduce releases of one contaminant may result in an increase or decrease in releases of other contaminants. Again, the benefits and costs from all the changes in contaminant levels should be accounted for in a BCA. However, when an action produces benefits from reductions in contaminants other than those related to the statutory objective of the regulation, and the benefits associated with these reductions in other contaminants are a large share of total benefits, or when net-benefits would be negative without them, then the analysis should identify other policy options that include directly regulating those contaminants.[21]

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17 Chapter 8 provides additional discussion of the advantages and disadvantages of vintage-differentiated regulations. Chapter 4 describes regulatory designs that can address some of the disadvantages.

18 The potential Pareto principle, or the compensation principle, states that economic welfare is improved by an action if the benefits of the action outweigh the costs (provided both benefits and costs can be measured accurately) because the gainers (those who benefit) could, theoretically, compensate the losers (those who bear the costs) and still be better off. Section A.3 of these Guidelines provides a further description of the potential Pareto principle.

19 Executive Order 12866 and OMB (2023) also consistently affirm that all benefits and costs should be assessed in BCA of regulatory actions.

20 The proposed or finalized option should also be reasonably robust to alternative potential baseline conditions. See Section 5.6 on uncertainty.

21 The statutory objective of the regulation is the specific objective of the statutory provision under which the regulation is promulgated.

In addition, an analysis of a policy option in which the other contaminant(s) are regulated directly, either separately or simultaneously with the regulation being analyzed, may be warranted.[22] If there are interactions in the control of contaminants, the most economically-efficient approach to their control requires simultaneously determining the appropriate policy design for each (e.g., Tietenberg, 1973). If there are important interactions in the control of multiple contaminants, options that jointly consider the appropriate design for each should be identified and may be analyzed, even if such considerations are not currently permissible. Correspondingly, there may be costs from increases in other environmental contaminants that are not associated with the statutory objective of the regulation.[23] If the effects of these increases due to the regulation are large, analysis of options to mitigate them may be warranted.

22 Chapter 5 provides further discussion and guidance on how to treat in an economic analysis those benefits from environmental contaminants other than those related to the statutory objective.

23 Such costs attributable to increases in other pollutants (and other environmental contaminants) should be accounted for even if future regulation might reduce them.

Chapter 3 References

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Chapter 4 - Regulatory and Non-Regulatory Approaches to Environmental Policy

This chapter describes several regulatory and non-regulatory approaches used in environmental policymaking. It also highlights a few key advantages and disadvantages of each approach, provides an overview of cross-cutting policy design issues, and offers references for those interested in a more in-depth discussion. This chapter covers four general approaches to environmental policymaking: (1) command-and-control regulation; (2) market-based approaches; (3) hybrid and other approaches; and (4) voluntary programs.[1] While command-and-control regulation has been a commonly used approach to environmental regulation in the United States, market-based and hybrid approaches can sometimes offer increased flexibility and lower costs. Voluntary programs may encourage environmental improvements or allow new approaches to be tested in areas not traditionally regulated by the U.S. Environmental Protection Agency (EPA).

The policy approaches discussed here are conceptually distinct, but they can sometimes be designed in ways to achieve similar benefits and costs. The approaches can also be combined into hybrid policy instruments, and multiple instruments can be used in tandem to address environmental problems caused by multiple market failures.[2] As such, the approaches discussed in this chapter represent an overlapping continuum of policy design tools.

4.1 Traditional Command-and-Control or Prescriptive Regulation

A prescriptive regulation is a policy that stipulates how much pollution an individual source or plant is allowed to emit and/or what types of control equipment or approaches it must use to reduce pollution. Prescriptive regulations are also known as "direct regulatory instruments" or "command-and-control" regulations (Goulder and Parry 2008; Ellerman 2006). Despite the introduction of potentially more cost-effective approaches for regulating emissions, this type of

1 Baumol and Oates (1988), particularly Chapters 10-14; Kolstad (2010); Field and Field (2021); Tietenberg and Lewis (2014); and Phaneuf and Requate (2016) are useful references on the economic foundations of many of the approaches presented here.

2 This chapter uses the terms "approaches" and "instruments" interchangeably when discussing various policy or regulatory tools.

regulation is still often used and is sometimes required by law. It is almost always available as a "backstop" if other approaches do not achieve desired pollution limits.

A common approach to prescriptive regulation is to issue a license or permit to an individual facility or firm that specifies the allowable level of pollution and the conditions under which it can be released into the environment. For instance, a permit issued to a hazardous waste treatment facility typically stipulates what waste management activities can be conducted at the site. It may also include requirements for safety and training, insurance, monitoring and reporting. The EPA may also set minimum standards when the licenses or permits are issued by another authority, such as states or tribes.

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

When abatement approaches and costs are similar across regulated sources, a source-level standard may be reasonably cost-effective. However, when abatement costs vary substantially across polluters, reallocating abatement activities so that some polluters abate more than others could lead to substantial cost savings. For example, if reallocation were possible through a less prescriptive market-oriented approach, a polluter facing relatively high abatement costs could continue to emit at its current level but would have to pay an emissions tax or purchase allowances, while a polluter with relatively low abatement costs could reduce its emissions, allowing it to avoid the tax or sell its allowances (see Section 4.3 for more discussion of these approaches).[3] A prescriptive regulation usually does not allow for reallocation of abatement activities to take place — each entity is expected to achieve a specified emission rate or use certain abatement technologies.

Prescriptive regulations can involve restricting—or in the most stringent case, prohibiting—the production, use, or disposal of specific products or substances. For instance, the EPA has banned most uses of chlorofluorocarbons (CFCs) and certain pesticides. This approach to regulation is potentially useful in cases where the level of pollution that maximizes social welfare is at or near zero. Prescriptive regulations include technology or design standards and performance-based standards, discussed below.

4.1.1 Technology or Design Standards

A technology or design standard mandates the use of specific control technologies or production processes an individual facility must use. This type of standard constrains firm behavior by mandating how a source must reduce pollution, regardless of whether such an action is cost-effective. Technology standards may be particularly useful in cases where the cost of emissions monitoring is high but determining whether a specific technology or production process has been

3 Tietenberg and Lewis (2014) discussed empirical studies on the cost-effectiveness of prescriptive air pollution regulations. Of the 10 studies included, eight found that prescriptive regulations cost substantially more than the most cost-effective strategy. Harrington et al. (2004) compared the costs and outcomes of command-and-control and market-based approaches in the United States and Europe. Newell and Stavins (2003) generated rules of thumb to help determine when market-based incentives may result in cost savings over prescriptive regulations.

put in place (and is operating properly) to meet a standard is relatively easy. However, since these types of standards specify the abatement technology required to reduce emissions, sources do not have an incentive to invest in more cost-effective types of abatement or to explore new and innovative abatement strategies that are not permitted by regulation. Also, because permitting authority is often delegated to the states, approval of a technology in one state does not ensure its use is allowed in another.

Key Advantages[4]

- Technology or design standards can yield environmental improvements with a high level of certainty.
- Technology or design standards can approximate an economically efficient outcome if the regulated industry has relatively homogeneous abatement costs across firms.
- If it is costly or infeasible to directly monitor emissions or environmental damages, technology standards may provide an easier approach to monitor compliance with regulatory requirements.

Key Disadvantages

- Technology or design standards are less likely to be economically efficient when there are a large number of diverse firms with varying abatement options because they do not allow for flexibility in the approach to pollution reduction or in the distribution of pollution reduction across sources.
- These standards reduce incentives for innovation of new technologies and approaches to achieve the environmental improvements at lower cost.
- These standards could motivate rent-seeking by firms producing pollution control technologies.

4.1.2 Performance-Based Standards

A performance-based standard requires that polluters meet a source-level emission standard but allows a polluter to choose among available methods to comply with the standard. At times, the available methods are constrained by additional criteria specified in a regulation. Performance-based standards that are technology-based do not specify a particular technology, but rather consider what is possible for available and affordable technology to achieve when establishing a limit on emissions.[5]

A performance-based standard can be defined in terms of 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 reduce output or to reduce emissions per unit of output by changing the technology or input mix. Therefore, an emission rate can be more restrictive depending on how it is defined. If the emission rate is defined per unit of output, then it only allows a source to meet the standard by reducing emissions per unit of output and not through reducing output. In some applications, it

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4 The discussion of key advantages and disadvantages of each approach is intended to highlight a few notable features but is not intended to be exhaustive.

5 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."

may even create incentives to increase output (Holland et al. 2009). If the rate is defined as an average amount of emissions over a certain time period, then the source may reduce output or emissions per unit of output to meet the standard.

Regulators can account for some variability in costs by allowing prescriptive regulations to vary according to size of the polluting entity, production process, geographic location, or other firm or product attributes that are not the direct target of the regulation. If the attribute is correlated with compliance costs, this type of attribute-based standard can improve economic efficiency relative to a uniform standard by helping to equate marginal compliance costs across firms in the absence of trading. However, the efficiency gains from equalizing marginal costs can be partially offset if firms comply with the standard by modifying the attribute instead of reducing pollution. For example, in 2011 the U.S. introduced a "footprint based" vehicle fuel economy standard, which subjected larger cars to a less stringent standard. The footprint-based standard was intended to discourage automakers from downsizing vehicles. However, researchers have found that such standards can incentivize automakers to design larger cars instead, eroding some of the gains in fuel savings expected from the standard (Whitefoot and Skerlos 2012; Ito and Sallee 2018).[6]

While performance-based standards encourage firms to meet the standard at lower cost than technology standards, they generally do not provide incentives to reduce pollution beyond what is required. There is still limited incentive for regulated firms to develop new, less expensive and potentially superior technologies compared to market-based policies (Swift 2000; Johnstone et al. 2010).

Key Advantages

- Performance standards, like technology and design standards, can yield environmental improvements with a high level of certainty.
- Performance standards can allow more flexibility to achieve environmental benefits at lower cost compared to technology or design standards.
- Performance standards create greater incentives for technological innovation than technology or design standards.

Key Disadvantages

- Performance standards are unlikely to be as economically efficient as market-based policies if abatement costs vary substantially across sources.
- Performance standards do not incentivize sources with low abatement costs to make environmental improvements beyond what the standard requires.
- If technological innovation yields lower-cost abatement opportunities in the future, the standard may need to be tightened over time to more closely approximate an economically efficient outcome.

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6 Attribute-based standards have been used in both prescriptive, non-tradable performance standards and market-based, tradable performance standards, of which the corporate average fuel economy standards for vehicles are an example. Tradable performance standards are a type of hybrid instrument discussed in Section 4.3.1.2. Making the standard tradable negates the potential economic efficiency gains associated with attribute-based standards (Ito and Sallee 2018).

4.2 Market-Based Approaches

Market-based regulatory approaches create an incentive for the private sector to incorporate pollution abatement into production or consumption decisions and prompt innovation to explore cheaper methods of abatement. Market-based approaches can differ from more traditional regulatory approaches in terms of economic efficiency, cost-effectiveness and the distribution of benefits and costs. Because market-based approaches do not mandate that each polluter meet a given emission standard, they typically allow firms more flexibility than prescriptive regulations and capitalize on the heterogeneity of abatement costs across polluters to reduce aggregate pollution efficiently. Environmental economists generally favor market-based policies because they tend to be less costly, they place a lower information burden on the regulator, and they provide incentives for technological advances.

Market-based policies create incentives for regulated firms to find the cheapest way to reduce pollution. This may involve a reduction in output (and in the extreme, exiting the industry), a change in inputs, the installation of pollution control equipment or a process change that prevents the creation of pollution. Polluters decide individually how much to control their emissions based on the costs of control and the financial incentives created by the policy.

Four market-based approaches are discussed in this section:

- Allowance trading systems;
- Emissions taxes;
- Environmental subsidies; and
- Tax-subsidy combinations.[7]

While operationally different, these market-based approaches put similar incentives in place. This is particularly true of emissions taxes and cap-and-trade systems, which can be designed to achieve the same goal at equivalent cost.

4.2.1 Allowance Trading Systems

Several forms of emissions trading exist, including cap-and-trade and project-based trading systems. The common element across these programs is that sources can trade credits, offsets or allowances so that those with opportunities to reduce emissions at lower costs have an incentive to do so. Emission-rate trading systems, a hybrid approach between tradable allowances and command-and-control, are discussed in Section 4.3.1.3.

4.2.1.1 Cap-and-Trade Systems

In a cap-and-trade system, the government sets the level of aggregate emissions, allowances are distributed to polluters, and a market is established in which allowances may be bought or sold. An allowance is a right to emit one unit of pollution; polluters must own an allowance for each unit emitted. The price of emission allowances is determined by supply and demand in the market and can vary over time.

7 Goulder and Parry (2008), Olmstead (2012), and Keohane and Olmstead (2016) compile theoretical and empirical information on the use of economic incentives.

For a uniformly mixed pollutant where marginal damages are identical for all sources and in all locations, if the cap is set at the economically efficient level, then the equilibrium price of allowances adjusts so that it equals the marginal damages from a unit of pollution. This equivalency implies that any externality associated with emissions is completely internalized by the firm. For polluters with marginal abatement costs greater than the allowance price, the cheapest option is to purchase allowances and continue to emit. For polluters with marginal abatement costs less than the allowance price, the cheapest option is to reduce emissions and forgo purchasing allowances (or to sell any allowances that they own at the market price). As long as the price of allowances differs from individual firms' marginal abatement costs, firms will continue to buy or sell them. Trading will occur until marginal abatement costs equalize across all firms.[8] Assuming no other market failures, an allowance price that is lower than the marginal damages from pollution implies that the cap is set at an inefficiently high level.

When the government sells allowances at auction, the revenue represents a transfer from the purchasers to the government. Allowance auctions can be designed in a variety of ways. Typically, allowances are purchased through a bidding process that reveals buyers' willingness to pay, with allowances going to the highest bidder.

The government could also decide to allocate allowances to polluters for free according to a specified rule either at the outset of the program or on an annual or other ongoing basis. This represents a transfer from the government to polluting firms, some of which may find that the value of allowances exceeds the firm's aggregate abatement costs (i.e., rents). Economic rents are any payment to the owner of capital or a resource above what it would cost to induce them to engage in a certain behavior.[9] The way in which allowances are allocated can also affect firm entry, exit, and production decisions. For example, allocating allowances based on historical emissions can create perverse incentives for old, dirty plants to continue to operate to qualify for allowances. Another alternative is to allocate allowances based on current production, which encourages firms to increase output to capture a greater number of future allocations (Fischer and Fox 2007; Lange and Maniloff 2021).

Additional considerations in designing an effective cap-and-trade system include the number of market participants, transaction costs, banking and hotspots. The United States' experience suggests that a market characterized by low transaction costs and being "thick" with many 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 can inhibit competitive behavior, and fewer trading opportunities result in lower cost savings. Likewise, the number of trades that occur could be significantly hindered by burdensome requirements that increase the transaction costs associated with each trade. Text Box 4.1 provides an example where thin markets resulted in few trades.

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8 Schmalensee and Stavins (2017) provide an overview of emission trading programs and lessons learned regarding implementation, system design and performance.

9 Tietenberg (2006) defined scarcity rent as, "producer's surplus which persists in long-run competitive equilibrium." In the context of a cap-and-trade market, these rents occur because firms are given allowances that can be bought and sold in the market. For a discussion of scarcity rents created by environmental regulations through pollution restrictions and captured by firms in the form of higher profits, see Fullerton and Metcalf (2001). Buchanan and Tullock (1975) discussed the potential for scarcity rents under a cap-and-trade system where permits are distributed for free.

Text Box 4.1 Water Quality Trading of Nonpoint Sources

In 2003, the EPA issued a "Water Quality Trading Policy" (U.S. EPA 2003) that encouraged 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. A 2019 memo announced additional flexibilities available to states and tribes to further facilitate the uptake of water quality trading, particularly between point and nonpoint sources. The memo cited the increased availability of effective nonpoint emission reducing technologies and practices and enhanced monitoring capabilities as reasons to modernize the 2003 policy (U.S. EPA 2019).

Under the Clean Water Act, states are required to establish Total Maximum Daily Loads (TMDLs) of pollutants for impaired water bodies. The TMDL is not a regulation and does not establish an enforceable cap on discharges to the watershed, but it does provide an approach for allocating pollutant discharges among point and nonpoint sources. Point sources are regulated under the Clean Water Act by the EPA and, as such, are required to hold National Pollutant Discharge Elimination System (NPDES) permits that limit discharges. Where a TMDL exists, the point source NPDES discharge limit is informed by the TMDL allocation. Nonpoint sources are not regulated under the Clean Water Act. However, many water bodies are still threatened by pollution from these 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 other coastal waters; and decrease fish populations in the Chesapeake Bay and other areas.

To account for uncertainties and differences associated with nonpoint source pollution, trading ratios are often applied. These ratios account for the differential effects resulting from a variety of factors, which may include:

- Location of the sources in the watershed relative to the downstream area of concern;
- Distance between the allowance buyer and seller;
- Uncertainty about nonpoint source reductions;
- Equivalency of different forms of the same pollutant discharged by the trading partners; and
- Additional water quality improvements above and beyond those required by regulation.

Trading can allow continued growth in production while providing nonpoint sources with an incentive to reduce pollution through participation in the market. If it is cheaper for a nonpoint source to reduce pollution than to forgo revenues earned from the sale of any unused credits to point sources, economic theory predicts that the nonpoint source will choose to emit less pollution.

As of 2014, the EPA had identified 19 nutrient trading programs in 11 states, with the majority of trades occurring in just three states — Connecticut, Pennsylvania and Virginia (GAO 2017). Trading has been limited in many of these programs for several reasons. First, as previously mentioned there is no enforceable cap on discharges that applies to both point and nonpoint sources within a watershed. Reductions by nonpoint sources are voluntary absent state-level mandates. Point-source dischargers often explore trading as a way to expand production while meeting the requirements of their individual permits, but there is no general signal in the market to do so, and it can be challenging to encourage nonpoint source involvement. Second, these are often thin markets (i.e., markets with few trades). The lack of participants can make it difficult or expensive for an entity to identify and complete a trade. Third, while best management practices (BMPs) are typically used to define a pollution reduction credit from a nonpoint source, uncertain or changing climatic conditions, river flow and stream conditions make it difficult to measure the effect of a BMP on downstream water quality. This uncertainty makes it difficult to define appropriate trading ratios between point and nonpoint sources (Morgan and Wolverton 2008; U.S. EPA 2008). Such uncertainty also makes measuring and enforcing a pollution reduction from a nonpoint source difficult.

Banking introduces increased flexibility into a trading system by allowing polluters to save unused allowances for future use. A firm may reduce emissions below the allowance level earlier and bank 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.

Cap-and-trade systems for non-uniformly mixed pollutants have the potential to create temporal or spatial spikes or "hotspots" — areas with particularly high pollution concentrations. Market-based policy should therefore be carefully designed and consider localized effects. While one potential solution to this problem is to adjust trading ratios (i.e., the rate at which allowances from one source can be traded with another) to equalize impacts, determining the appropriate adjustments to these ratios can be costly and difficult. Another possible solution is zone-based trading. Two reviews of the literature on SO_2 and NO_x trading programs (see Text Box 4.2) found little evidence of spatial or temporal spikes in pollution (Burtraw et al. 2005; Harrington et al. 2004). In fact, they have led to smoothing of emissions across space in some cases. However, it is still important to consider the potential for localized effects when designing cap-and-trade policies. See Section 4.5.1 for a discussion of distributional concerns.

4.2.1.2 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 flexibility in complying with emission limits or facility-level permits.[10] A bubble allows a facility to consider all sources of emissions of a specific pollutant within the facility to achieve an overall target level of emissions or environmental improvement. To meet air quality standards for particulate matter, EPA employed a compliance bubble alternative that allowed pulp and paper mills to set site-specific emission limits as long as total emissions from all sources within the site were less than or equal to the standard. This flexibility resulted in lower the compliance costs (Morgan et al. 2014).

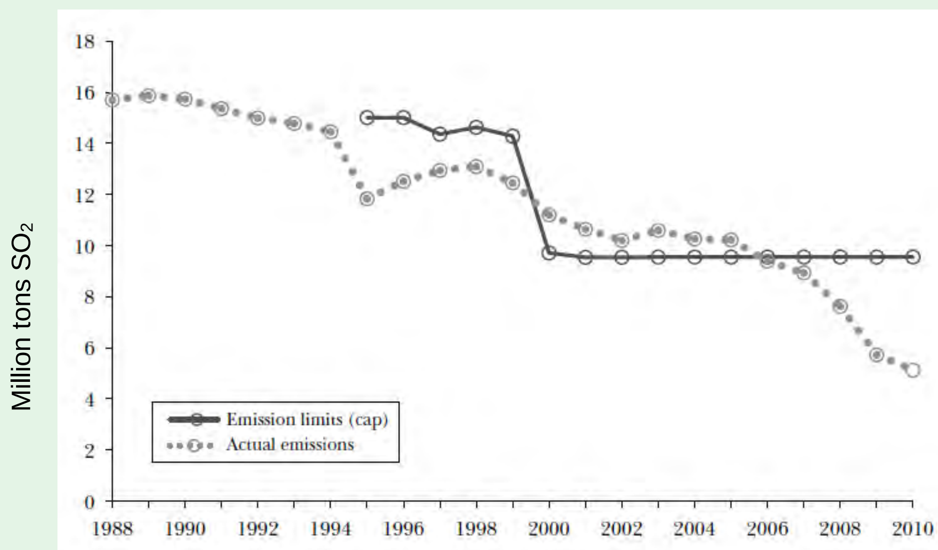
10 Bennear and Coglianese (2012) evaluated how these types of flexibilities have worked in the United States.

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

In 1995, Title IV of the 1990 Clean Air Act Amendments established a cap-and-trade system for SO2 emissions to address acid rain. The 263 highest SO2-emitting units at 110 electric utility plants were selected to participate in Phase I of the trading program to limit SO2 emissions to 8.7 million tons in 1995. Most of the plants that participated in Phase I were coal-fired units located east of the Mississippi River. Allowances were allocated to units on a historical basis for plants to use, sell to other units, or "bank" for use in later years. Phase I plants were required to install continuous emission monitoring systems, allowing for easy monitoring and enforcement of emission restrictions in accordance with the allowances. The second phase of the program, initiated in 2000, imposed a national SO2 emissions cap of 10 million tons and brought almost all SO2-emitting units into the system.

Evaluations of the Phase I suggest that the SO2 trading system significantly reduced emissions. Compliance costs were estimated to be 15 to 90% lower than an equally stringent command-and-control alternative. The success of the program continued into Phase II. Chan et al. (2018) estimated Phase II annual cost savings at $700 million compared to a simulated uniform performance standard.

SO_2 Caps and Emissions, 1988-2010



In the figure above, Schmalensee and Stavins (2013) reported that emissions declined by 36% between 1990-2004, even as coal-fired electricity generation increased. One reason for such large emission reductions was the ability to bank allowances for future use. In addition, incentives to innovate continued to reduce abatement costs over time (Bellas and Lange 2011; Frey 2013). Railroad deregulation and investment by utilities in mining and infrastructure also played a role by making low-sulfur coal cheaper. That said, researchers observed less inter-firm trading than expected, meaning that marginal abatement costs were not equalized across plants (Swift 2001; Swinton 2004). Estimates of the SO2 allowance program's annual benefits range from $59-116 billion with estimated annual costs of $0.5 to $2 billion (in 2000$) (Schmalensee and Stavins 2013).

Congress did not grant the EPA the authority to adjust the cap in response to new information on either the costs or benefits of reducing emissions. For this reason, the EPA pursued additional reductions in SO_2 emissions via more traditional regulatory approaches, which restricted the ability of sources to trade and reduced allowance prices to zero by 2012 (Schmalensee and Stavins 2013).

For more information, see Chestnut and Mills (2005), U.S. EPA (2007), Schmalensee and Stavins (2013), Chan et al. (2018), and Evans and Woodward (2013).

An offset allows a new polluter to negotiate with an existing source to secure a reduction in the latter's emissions. Offsets, which entail cross-firm emissions trading, have at times been hindered by high administrative and transaction costs. However, regulators can improve the economic efficiency of offsets by allowing third parties, who are not themselves polluters, to participate in the market. For instance, evidence suggests that some water quality offset programs that operated through an intermediary or clearinghouse eliminated the need for direct negotiations between buyer and seller and lowered these costs substantially (Woodward and Kaiser 2002; Morgan and Wolverton 2008). Offsets have also been included in cap-and-trade programs for greenhouse gas emissions such as the Clean Development Mechanism of the Kyoto Protocol. Such systems allow entities covered under the cap to purchase offsets for emission reductions or carbon sequestration from firms in industries or locations not covered under the program, increasing the flexibility and reducing the costs of meeting the aggregate greenhouse gas emissions target.

Key Advantages

- Like other market-based policies, tradable allowance systems can be more economically efficient than prescriptive regulations, particularly when there are many heterogenous market participants.
- Like other market-based policies, tradable allowances create incentives for innovation as firms compete for new ways to reduce emissions at the lowest cost.
- Tradable allowance systems provide more certainty about the total level of emissions than emissions taxes or subsidies; as such, they may be preferable to emissions taxes when marginal damages increase with the level of emissions.

Key Disadvantages

- If the pollution cap is set at an inefficiently high (low) level, then allowance prices will be lower (higher) than the marginal damages from pollution, and an inefficiently low (high) level of abatement will occur.
- Tradable allowance systems raise complicated issues regarding the distribution of allowances, including auction design and rent-seeking by regulated firms.
- Tradable allowance systems can result in emissions spikes or hotspots. Regulators can set trading ratios or use additional instruments to avoid hotspots or to address heterogeneous pollution damages, but such additional requirements raise analytical and administrative challenges.

4.2.2 Emissions Taxes

Emissions taxes are a charge per unit of pollution that is imposed by the government. Under an emissions tax, the polluter will abate emissions up to the point at which the additional cost of abating one more unit of pollution is equal to the tax. For any remaining emissions, the polluter prefers to pay the tax rather than to abate further. The tax will result in an economically efficient outcome if it is set equal to the external damage caused by the last unit of pollution emitted.[11]

User or product charges are a variation on emissions taxes. These charges may be imposed on users of publicly operated facilities or on intermediate or final products whose use or disposal harms the environment. User or product charges may be effective approximations of an emissions tax when the product is closely related to environmental damage. User and product charges will not result in an efficient level of pollution if they are set at a level sufficient to recover only the *private costs* of operating a public system, rather than incorporating the marginal social damages of pollution.

Emissions taxes, like tradable allowance systems that distribute the allowances using an auction, raise revenue for the government. The welfare and distributional effects of an emissions tax depend on how the revenues are used and how the tax interacts with other distortions in the economy. If distributed to households or firms, the revenues can be used to compensate individuals made worse off by the policy or to address other distributional priorities of the policymaker, though it can be difficult to accurately target individuals for compensation (Cronin, Fullerton and Sexton 2019). If the revenues are instead used to reduce other distortionary taxes, such as labor taxes, then this "revenue recycling" could yield economic gains due to a resulting increase in employment or investment (e.g., Goulder 2000). However, emissions taxes or allowances can also exacerbate pre-existing tax distortions, causing an increase in deadweight loss. Analysts should consider the opportunity costs associated with collecting and spending public funds. Section 8.3.1 of these *Guidelines* discusses general equilibrium approaches to examine these types of economy-wide effects.

Emissions taxes should lead to outcomes similar to those from allowance trading systems when both are designed to achieve the same level of emissions. Rather than specifying the total quantity of emissions, taxes specify the effective “price” of emitting pollutants. However, these two types of policy instruments differ in their usefulness when there is uncertainty about the costs or benefits of abatement. Section 4.5.5 discusses instrument choice under uncertainty.

Key Advantages[12]

- Like tradable allowances, emissions taxes are an economically efficient approach to incentivize pollution reduction, allowing flexibility to reduce emissions multiple ways and/or to pay the tax for remaining emissions.
- Like tradable allowances that are distributed via auction, emissions taxes raise revenue that can be used to compensate individuals made worse off by the policy or to offset other distortionary taxes, increasing economic efficiency throughout the economy.
- Emissions taxes are advantageous in situations where there is uncertainty about abatement costs, but damages do not change much with additional pollution.

Key Disadvantages

- Emissions taxes do not set source-level or aggregate limits on emissions and do not eliminate the potential for emissions spikes or hotspots.
- Emissions taxes may be difficult to implement efficiently when pollution damages vary over space and time.

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11 These taxes are called “Pigovian” after the economist, Arthur Pigou, who first formalized them (Pigou 1932).

12 See Fullerton, Leicester and Smith (2010) for more discussion of the advantages and disadvantages of emissions taxes.

- Emissions taxes are less well-suited to situations in which contaminant releases are difficult to measure and are not directly related to a marketed input or output.

4.2.3 Environmental Subsidies

A subsidy is a payment or financial assistance made to encourage a certain behavior. Subsidies paid by the government to firms or consumers for technology-neutral reductions in pollution create similar abatement incentives as emissions taxes. Economic theory predicts that firms will reduce pollution up to the point where the additional private costs are equal to the subsidy.

Unlike an emissions tax, an environmental subsidy lowers a firm's total and average costs of production, encouraging production by both existing and new firms. The result may be a decrease in emissions from individual polluters but a smaller net decrease (or even an increase) in overall pollution.[13] However, it is possible to minimize the entry and exit of firms resulting from subsidies by redefining the subsidy as a partial repayment of verified abatement costs, instead of defining it as a per-unit payment for emissions reductions relative to a baseline. Defining the subsidy in this way also minimizes strategic behavior because no baseline must be specified.[14] An environmental subsidy also differs from an emissions tax because it requires government expenditure (versus generating government revenue).

Government funding for research and development of technologies to reduce pollution and improve environmental quality is another form of subsidy. The private market does not always have an incentive to invest in the socially optimal level of innovation and diffusion of new technologies because these activities can create positive information spillovers that benefit other firms. In addition, network externalities, which occur when the net benefits of adopting a new technology increase as the number of users increases, can limit the spread of otherwise promising innovations (Jaffe, Newell and Stavins 2005).[15] Subsidies for technology development and demonstration can be used to address these types of market failure, complementing other environmental policy approaches. Research on new technologies and approaches to improve environmental quality may also yield data that could be useful in future analyses of regulatory or non-regulatory approaches to environmental policy.

Cost-sharing constitutes another type of subsidy, with examples that include reduced interest rates, accelerated depreciation, direct capital grants, loan assistance or guarantees for investments, and government "buy-backs." Under a buy-back program, the government offers a payment for the return of an older, high-polluting product or a rebate on a new, cleaner substitute if the older model is turned in. For example, the EPA has funded changeout programs to encourage the replacement of old wood stoves with EPA-certified gas, electric or wood appliances that reduce indoor air pollution

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13 See Sterner and Coria (2012) and Goulder and Parry (2008) for a discussion and examples of environmental subsidies.

14 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 reality to receive credit for large improvements. If firms or consumers are responsible for paying for emissions above a given level, they may try to lobby for that level to be set at a fairly high level so that they pay less in fines or taxes.

15 Electric vehicle adoption provides one example of network externalities. The cost and convenience of electric vehicle use depends on the availability of a network of electric charging stations. Spreading the cost of this infrastructure across many users lowers the costs for each individual user.

(U.S. EPA 2014). In 2009, the U.S. ran a program called "Cash for Clunkers" that offered rebates for trading in old, fuel-inefficient, but still drivable vehicles for new, fuel-efficient vehicles to stimulate auto sales during a recession.

The effectiveness of subsidies depends on the degree to which they motivate behavior that would not have already occurred without the subsidy (an effect called "additionality"). In the Cash for Clunkers program, researchers estimated that most of the funds were received by consumers who would have purchased a vehicle in 2009 regardless, though the program did induce sales of more fuel-efficient vehicles than would have been purchased without the subsidy (Li, Spiller and Lin 2013). Similar to allowance trading systems, auctions can be incorporated into subsidy programs to incentivize participants to reveal their opportunity costs and avoid payments in excess of this amount. In these programs, sometimes referred to as conservation or reverse auctions, subsidies are awarded to the lowest bidder (de Vries and Hanley 2016). The effectiveness of subsidies targeted to households also depends on transaction costs and other non-financial barriers such as lack of trust and inattention. For example, programs subsidizing home energy efficient retrofits and lead water pipe replacements have had low adoption rates, even when coupled with information and outreach, possibly due to the hassles of completing extensive paperwork or having construction work in the home (Fowlie et al. 2015; Klemick et al. 2024).

A subsidy for specific technologies — a policy approach that is sometimes termed "picking winners" — is typically not as economically efficient as a subsidy (or tax) per unit of emission reduction or other environmental outcomes. This is because, similar to prescriptive regulation, such programs do not encourage flexibility in the way firms or individuals reduce their adverse environmental impacts, and the government may not have good information on what technology will ultimately be the most efficient abatement option.

Key Advantages

- Technology-neutral environmental subsidies are an economically efficient way to encourage pollution reduction because they create incentives to reduce emissions up to the point at which marginal abatement costs equal the subsidy.
- Subsidies for research and development of new pollution abatement technologies and approaches can help mitigate market failures that inhibit technological innovation.
- Subsidies provide flexibility to polluters about whether and how much to abate and impose no mandatory requirements on the public.

Key Disadvantages

- Subsidies have limited effectiveness if most market participants would have undertaken the environmentally beneficial action without the subsidy — in this case, the subsidy acts as a transfer and results in no net social benefit.
- Subsidies for specific technologies are typically less efficient than technology-neutral subsidies because they allow less flexibility for achieving environmental improvements.
- Like emissions taxes, subsidies provide less certainty that source-specific or aggregate emissions will remain below a particular level; emission spikes or hotspots could occur.

4.2.4 Tax-Subsidy Combinations

Emissions taxes and subsidies can be combined to achieve the same level of abatement as when each instrument is used alone. One example of this type of instrument is a deposit-refund system. Under a deposit-refund system, firms or consumers pay an upfront deposit that serves as a tax on the production or use of certain goods. A refund is then provided if firms or consumers demonstrate that they used a cleaner form of production or engaged in proper disposal, acting as a subsidy.[16]

A tax and subsidy combination functions best when there is a direct relationship between use of a product and emissions. For instance, a tax on the production or use of hydrochlorofluorocarbons (HCFCs) combined with a refund for HCFCs recycled or collected in a closed system is a good proxy for an HCFCs emissions tax.

The main advantage of a combined tax and subsidy is that both parts apply to a market transaction. Because the taxed and subsidized items are easily observable in the market, this type of economic instrument is 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 the government does, they can measure and report their actions with greater precision and at a potentially lower cost.

A disadvantage of the combined tax-subsidy system is potentially high implementation and administrative costs. In addition, while it is possible to adjust an emissions tax to account for variation in marginal damages, a tax on output cannot be matched temporally or spatially to emissions during production. Likewise, if inputs contribute differentially to environmental damages, then it is necessary to tax them at different rates to achieve economic efficiency. When firms are heterogeneous and select a different set of inputs or abatement options based on firm-specific cost considerations, then the subsidy needs to be adjusted for these differences. Given these complications, other market-based approaches may have lower implementation costs when emissions are easily monitored.

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 upfront funds to a pool. Such funds may be used for pollution abatement or to compensate individuals harmed by pollution if environmental damages occur. If the company demonstrates it has fulfilled certain obligations, the contribution is usually refunded.[17]

Key Advantages[18]

- Like tradable allowances and emissions taxes, tax-subsidy combinations can be an economically efficient approach to achieve environmental improvements.
- A tax-subsidy combination can be useful when it is less costly to observe market outputs and inputs than it is to observe emissions or environmental damages.
- Performance bonds, a conceptually similar approach, create a pool of funds that can be used to abate pollution or to compensate individuals affected by environmental damages.

Key Disadvantages

- Tax-subsidy combinations can involve high implementation and administrative costs.

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16 When a deposit-refund encourages firms to use a less-polluting input, a deposit on output induces the firm to reduce its use of all inputs, both clean and dirty (i.e., the output effect). The refund provides the firm an incentive to switch to a specific input such as a cleaner fuel (i.e., the input substitution effect).

17 For more information on the use of financial assurance or performance bonds, see Davis (2015), Dana and Wiseman (2013), and Boyd (2002).

18 The main advantages and disadvantages of deposit-refund systems are discussed in Walls (2013) and Fullerton and Wolverton (2001, 2005). Fullerton and West (2010), Walls (2013), and Sterner and Coria (2012) provide more discussion and examples of tax-subsidy combinations.

- It is difficult to adjust tax-subsidy combinations to account for heterogeneity in environmental damages.
- Like other market-based policies, the lack of a limit on individual sources means that hotspots with a high concentration of pollution can occur.

4.3 Hybrid and Other Approaches

In addition to the instruments discussed above, several other approaches have been used alone or in combination. This section discusses the following approaches:

- Combining prescriptive and market-based approaches;
- Liability rules and insurance requirements;
- Information disclosure; and
- Behavioral economics and "nudge" approaches.

4.3.1 Combining Prescriptive and Market-Based Approaches

Some policies combine aspects of prescriptive and market-based policies. As such, they may not represent the most economically efficient approach. The cost of the policy is likely to be greater than what would be achieved using a pure market-based approach. Nevertheless, such approaches are appealing to policymakers because they combine the certainty associated with a standard or technology with some flexibility, allowing firms to comply at a lower cost. Combining standards and pricing and tradable performance standards are two hybrid approaches.

4.3.1.1 Combining Standards and Pricing

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 the quantity of emissions and leave open the possibility that pollution may be excessively high. Some researchers suggest a policy that limits both costs and pollution by combining quantity and pricing instruments, such as a “safety-valve” or price collar (Jacoby and Ellerman 2004; Fell and Morgenstern 2010; Fell et al. 2012). In the case of a prescriptive standard and tax combination, an emission standard is imposed on all polluters, but polluters can pay a unit tax for emissions above the standard. Safety-valve systems can also be entirely market-based, by combining a cap-and-trade policy with an emissions tax (i.e., an allowance price ceiling and/or floor) if allowance prices go above or below a certain level (Burtraw, Palmer and Kahn 2010).

This policy combination has several attractive features. First, it allows for more certainty in the expected environmental and health effects of the policy than would occur with a pricing approach alone.[19] Second, overall abatement costs are lower than under a prescriptive standard because polluters with low abatement costs reduce pollution while polluters with high abatement costs pay taxes.

19 Section 4.5.5 elaborates on instrument choice under uncertainty.

4.3.1.2 Tradable Performance Standards

A tradable performance standard establishes a standard or emission rate, as described in section 4.1.2, but introduces credit trading and banking as additional flexibilities. Sources that perform better than the standard can earn credits and sell them to sources that perform worse or can save them for future use. A credit allows a source to emit one unit of a pollutant in excess of what would normally be permitted (e.g., reducing emissions below a baseline or cap).

In rate-based trading systems, sources able to reduce their emission rate at low cost have an incentive to do so since they can sell the resulting credits to sources facing higher abatement costs. Rate-based trading programs have been used in the United States to phase out lead in gasoline (Newell and Rogers 2006; Schmalensee and Stavins 2017) and to control emissions of light-duty and heavy-duty vehicles (Bento et al. 2020; Leard and McConnell 2017). Similarly, state Renewable Portfolio Standards require the use of renewable energy sources — such as wind or solar — for electricity generation, but they incorporate tradable credits so that firms can meet the overall standard at least cost. These approaches encourage cleaner transportation or electricity, but they do not allow reducing output or consumption as a way to comply with the standard (e.g., reducing vehicle miles traveled or electricity consumption). Emissions may increase under these programs if sources increase their production or if new sources enter the market. The regulating authority may need to periodically impose new standards to maintain the desired emission target, which may lead to uncertainty in the long term for regulated sources.

Key Advantages

- Combining prescriptive and market-based approaches can achieve a particular emission rate or technology adoption target at least cost.
- Combining approaches can increase certainty about achieving an emission rate or technology adoption target.
- "Safety-valve" systems that combine cap-and-trade with an emissions tax (a price floor and/or price ceiling) achieve the economic efficiency of market-based policies while mitigating uncertainty about abatement costs and emission reductions.

Key Disadvantages

- A combined prescriptive and market-based policy is typically not the most economically efficient approach because it limits flexibility in the way that environmental improvements are achieved.
- A tradable emission rate is not the most efficient approach to improving environmental quality because it does not create incentives to reduce output or consumption.
- Like prescriptive regulations, if the hybrid approach does not set an overall limit on emissions across the regulated sector, then it is possible for total emissions to increase even if source-level emissions decline.

4.3.2 Information Disclosure

Market failure due to imperfect information occurs when firms or consumers are unable to make optimal decisions due to lack of information on emission levels, health and ecological risks, or approaches to mitigate these risks. Asymmetric information exists when one party in the transaction has more information than others, which can also yield suboptimal outcomes. Regulations requiring disclosure of environmental information can minimize inefficiencies associated with imperfect or asymmetric information.[20] Information disclosure can also be an important component of non-regulatory EPA programs. By collecting and making information publicly available, firms, government agencies and consumers can become better informed about the environmental and human health consequences of their production and consumption decisions.

In some cases, the availability of this information may encourage more environmentally benign activities and discourage environmentally detrimental ones. For example, warning labels on hazardous substances that describe risks or safe-handling procedures may encourage consumers to take greater precautions or switch to less-damaging substitutes. A community with information on a nearby firm's pollution activity may exert pressure on the firm to reduce emissions, even if regulations to limit pollution are weak or nonexistent.[21]

Requirements for information disclosure need not be tied to an emission standard. However, such requirements might allow members of the public to easily understand the level of emissions in the context of existing standards. As with market-based instruments, polluters have the flexibility to respond to community pressure by reducing emissions in the cheapest way possible.

The use of information disclosure or labeling rules has other advantages. When expensive emissions monitoring is required to collect information, reporting requirements that switch the burden of proof for monitoring and reporting from the government to the firm might result in lower costs, because firms are often in a better position to monitor their own emissions. However, random inspections may be needed to ensure that monitoring equipment functions properly and that firms report results accurately. Information disclosed to regulators or the public through such programs could be useful for analysis of other potential regulatory approaches in the future.

Information disclosure alone does not typically result in an economically efficient level of pollution when externalities are present. Several conditions are necessary for it to be effective and welfare-improving. The information must be complete and accurate. Consumers must be able to access the information and understand it.[22] In addition to complete information, Coase (1960) identified low transaction costs and the possibility of bargaining as two conditions necessary for a private agreement between affected parties to lead to an efficient level of pollution (see Text Box 3.1). A community's ability to bargain with or exert pressure on an emitting plant may be related to socioeconomic status. Lower income, less-educated populations may face more barriers to political participation and be less likely to have their concerns addressed than richer, well-educated populations (Hamilton, 1993; Arora and Cason, 1999; Earnhart, 2004). The effect that public pressure has on behavior may also vary by firm and depend on factors such as the firm's market power and societal reputation. Finally, even if information is complete and consumers can access it readily — which may be strong assumptions — individuals do not always act to further their own best interests, as discussed in Section 4.3.4.

The most studied environmental disclosure program is the Toxics Release Inventory (TRI), but researchers offer a mixed view on the extent to which it has changed firm behavior. Some studies have found that high-polluting firms experienced stock price declines on the day the TRI was publicly released, and that those with the largest drop in stock prices reduced reported emissions

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

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

22 As noted in Sunstein (2011), "accurate disclosure of information can be ineffective if the information is too abstract, vague, detailed, complex, poorly framed, or overwhelming to be useful."

the most in subsequent years (Hamilton 1995; Konar and Cohen 1997).[23] Others found no evidence of a negative stock price effect (Bui 2005). Bae et al. (2010) found that making raw TRI data available did not significantly change environmental risk, even when emissions declined. However, when data were processed and presented to the public in more digestible terms, they found a significant decline in environmental risk. The economics literature has also found evidence that consumers respond to product labels in specific cases.[24]

Key Advantages

- Information disclosure requirements can help address market failures due to imperfect or asymmetric information.
- Information programs can complement other approaches, including emission standards, market-based approaches and nudges.
- Reporting requirements that make new information available to the public and to government agencies can yield new data useful for developing improved regulatory analyses in the future.

Key Disadvantages

- On their own, information disclosure requirements are not well-suited to addressing market failures due to externalities; conditions necessary for bargaining to result in an economically efficient outcome are not typical of most markets.
- It may be particularly difficult for disadvantaged communities to influence polluters to reduce their environmental damages in response to information disclosure.
- Information programs have not been studied extensively, and empirical evidence on their effectiveness is mixed.

4.3.3 Liability Rules and Insurance Requirements

Liability rules impose a legal responsibility for polluters to pay for environmental damages after they occur. These instruments serve two main purposes: (1) to create an economic incentive for firms to incorporate the cost of environmental damages into their decision-making processes; and (2) to compensate harmed individuals when damages occur. These rules are used to guide compensation decisions when the court rules in favor of the victim. To the extent that polluters are aware that they will be held liable before the release occurs, they have an incentive to minimize damages to others.

While a liability rule can be constructed to mimic an economically efficient market solution in certain cases, there are reasons to expect that this efficiency may not be achieved. First, payments need not reflect the social damages. The amount that polluters are required to pay after damages have occurred is dependent on the legal system and may be limited by an inability to prove the full extent of damages or by the ability of the firm to pay. Second, liability rules can generate large

23 Khanna, Quimio and Bojilova (1998), Bui and Mayer (2003), Banzhaf and Walsh (2008), and Mastromonaco (2015) also have investigated how the TRI has affected firm behavior, stock market valuation and housing markets.

24 For example, Teisl, et.al (2002) and Bjørner, et. al (2004) studied the effects of labels for dolphin-safe tuna and paper products, respectively, on consumer purchases. Brounen and Kok (2011) examined the extent to which energy performance labels are capitalized into housing prices.

transaction costs, both in terms of assessing the environmental damage caused and the resources used to take legal action.[25]

Liability rules are most useful in cases where damages requiring compensation are infrequent (e.g., accidental releases) and where monitoring compliance with other regulatory approaches is difficult. Finally, the scope of liability may affect overall economic efficiency. Under the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA), for example, new owners of contaminated land are defined to be potentially responsible parties that can be held liable for past pollution, creating disincentives for the redevelopment of contaminated land (Jenkins, Kopits and Simpson 2009).[26] Depending on the effectiveness of liability rules to provide incentives to firms to minimize environmental damages, they can be either an alternative or a complement to other regulatory approaches.

Strict liability and negligence are two types of liability rules. Under strict liability, polluters are held responsible for all health and environmental damages caused by their pollution, regardless of actions taken to prevent the damages. Under negligence, polluters are liable only if they do not exhibit "due standard of care." Regulations that impose strict liability on polluters may reduce the transactions costs of legal actions brought by affected parties. This approach may induce polluters to alter their behavior to reduce the probability of a pollution release that causes damages (Akey and Appel 2021).

Requiring polluters to carry insurance is another approach that can be used to reward risk-reducing and penalize risk-increasing behavior through the setting and adjustment of insurance premiums. This instrument also generates a pool of money that can be used for remediation when contamination occurs. Dana and Wiseman (2013) discussed this approach in the context of oil and gas well development. Insurance has also been discussed to pool risk against extreme weather events in the context of climate change (Linnerooth-Bayer and Hochrainer-Stigler 2015).

Key Advantages

- Liability rules and insurance requirements can incentivize polluters to adopt behaviors that reduce the risk of environmental damages.
- Liability rules and insurance requirements are most useful for situations in which environmental damages are infrequent and monitoring compliance with other types of regulatory requirements is costly.
- Liability rules and insurance requirements both involve ways to compensate those harmed by contaminant releases.

Key Disadvantages

- Payments by polluters to harmed individuals under liability rules are determined by the legal system and need not be equal to social damages; therefore, on their own, they may not create an incentive for polluters to undertake an economically efficient level of mitigation.
- Insurance requirements will only yield an efficient level of environmental protection if premiums are set to encourage firms to undertake abatement up to the level at which marginal costs equals marginal social damages.

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25 Segerson (1995) and Alberini and Austin (2001) discussed different types of liability rules and their efficiency properties.

26 The Small Business Liability Relief and Brownfields Revitalization Act eased some of CERCLA's liability provisions to encourage the redevelopment of potentially contaminated industrial sites, known as brownfields.

- Determining payments through the legal system entails high transaction costs, including resources used in the legal process and to measure environmental damages.

4.3.4 Behavioral Economics and "Nudge" Approaches

The neoclassical economics paradigm that has helped inform the design of market-based and other policy instruments makes several simplifications about human behavior — for instance, that people are rational, well-informed, self-interested and disciplined. While these may be reasonable assumptions in many contexts, they do not always hold in the real world. Behavioral economics is a subfield at the intersection of economics and psychology that examines departures from the neoclassical economics model. Such behavioral anomalies include cognitive limitations, altruism, inequality aversion, procrastination, status quo bias and loss aversion, among others.[27]

Behavior that is altruistic, short-sighted or inattentive may have important implications for the way environmental policies are designed and enforced.[28] Inattentive or impatient behavior may help explain some consumers' reluctance to invest in energy-saving appliances or fuel-efficient cars that cost more upfront but save money in the long run. Altruism and social norms may lead people to purchase eco-labeled products even absent regulation or price signals.

Insights from behavioral economics can be relevant to the design of many types of policy instruments. In addition, they present the opportunity to design policies that "nudge" people to make choices that improve their well-being. Nudges have been proposed as an approach to encourage socially beneficial actions by making small changes to the context in which people make decisions. Thaler and Sunstein (2021) define a nudge as "any aspect of the choice architecture that alters people's behavior in a predictable way without forbidding any options or significantly changing their economic incentives," elaborating that, "the intervention must be easy and cheap to avoid. Nudges are not taxes, fines, subsidies, bans, or mandates."

While market-based policies are typically designed to correct externalities, nudges may be especially relevant in situations where the market under-provides environmental quality due to lack of information, cognitive limitations, procrastination, or other behavioral anomalies. In contrast to the use of information disclosure alone, a nudge or nudges emphasize the visual design, timing, delivery method and other aspects of the way information is presented to make it more salient and useful. Other strategies that have been used as nudges include default rules that require individuals or firms to opt out of a program instead of opting in, moral persuasion or pro-social messages that appeal to a sense of altruism or fairness, ordering choices to put the most beneficial option first, and the use of social norms that tap into individuals' desire to match or outperform their peers.[29] Examples of nudges outside the realm of environmental policy include automatic

27 Loss aversion occurs when individuals facing risky choices place greater weight on losses compared to gains of an equivalent value. Empirical research suggests that many people tend to give losses double the weight of gains (Kahneman and Tversky 1979, Tversky and Kahneman 1992). Loss aversion can contribute to status quo bias, which describes a preference for avoiding any change from the current situation.

28 Shogren and Taylor (2008), Shogren, Parkhurst, and Banerjee (2010), and Croson and Treich (2014) provide in-depth discussions of the intersection between behavioral economics and environmental economics.

29 Executive Order 13707, "Using Behavioral Insights to Better Serve the American People" (The White House, Sept. 15, 2015), encouraged federal agencies to consider behavioral science strategies with particular attention to access to programs, presentation of information to the public, the structure of choices within programs and the design of financial and non-financial incentives.

enrollment of employees into retirement savings plans (Madrian and Shea 2001) and rearranging cafeterias to make healthy foods more convenient or eye-catching (Hanks et al. 2012).

There are many potential applications of nudges to environmental policy. For example, research has shown that providing residential consumers with real-time information about electricity consumption and prices can reduce electricity use, which can lead to decreased pollution from fossil-powered electricity generation. Signals conveyed visually, such as a "glowing orb" that changes color to reflect changes in prices or demand, have been shown to be particularly effective.[30] Residential consumers who received reports comparing their own consumption of water or electricity to that of their neighbors also reduced their resource consumption (Allcott 2011b, Ferraro and Price 2013). Text Box 4.3 describes a few EPA examples.

Nudges that are effective in one situation are not always transferable to different contexts. For example, the residential electricity consumption reports mentioned above led to larger reductions in electricity use for high-user households and for environmentalists, while they have been less effective for other households (Allcott 2015). In addition, research on electricity consumption has yielded mixed results on the effectiveness of combining various nudges and financial incentives (Pellerano et al. 2017; Brandon et al. 2019). These examples highlight the importance of using rigorous empirical approaches such as randomized controlled trials to test the effectiveness of new nudges before adopting them on a wide scale (List and Metcalfe 2014; Allcott and Mullainathan 2010; Hahn and Metcalfe 2016).

Beyond nudges, behavioral economics insights can be applied in the design of other policy instruments. The implementation of plastic bag taxes provides one example. Standard economic models predict that individual consumers will respond similarly to market incentives regardless of whether they are presented as a tax on damaging activities or a subsidy for beneficial activities. However, research has found that consumers faced with a fee for disposable bags cut their bag use by more than 40%, but no change occurred in response to a subsidy for reusable bag use (Homonoff 2018). This result is consistent with loss aversion and suggests that consumer responsiveness to market-based policies can depend on how the incentives are framed.

Key Advantages

- Nudges can address environmental problems that occur or are exacerbated due to inattention, impatience or other behaviors inconsistent with rational choice theory.
- Nudges can complement other approaches, particularly information disclosure requirements.
- Nudges are low-cost and impose no mandatory requirements on the public.

Key Disadvantages

- On their own, nudges are not well-suited to addressing market failure due to externalities.
- Nudges are not well-suited to addressing sectors in which rational, profit-maximizing behavior is well documented.
- Empirical evidence on nudges' effectiveness in improving environmental outcomes is limited.

30 Allcott (2011a) and Jessoe and Rapson (2014) focused on real-time electricity pricing, while Houde et al. (2013) examined the effect of real-time electricity consumption information.

Text Box 4.3 - Nudging Through Labels

Product labels represent an intriguing opportunity to examine whether the way information is presented can nudge consumers toward environmentally friendly purchases. For example, some research has found that the EPA's ENERGY STAR logo encourages investments in energy efficient appliances more effectively than information on energy use and expenditures alone (Newell and Siikamäki 2014).

The EPA also collaborated with the U.S. Department of Transportation in 2011 on the redesign of labels to convey the fuel efficiency and environmental attributes of light duty vehicles. They considered elements like color, layout, graphics and alternative rating scales. One issue they confronted was which metric to use to represent fuel economy. Research by Larrick and Soll (2008) pointed out that miles per gallon (mpg) can mislead consumers about fuel expenses and tailpipe emissions because mpg is not linearly related to fuel consumption. Consumers are especially likely to undervalue small changes in mpg for less fuel-efficient vehicles because most are not aware that shifting from 10 to 12 mpg, for example, saves more fuel than increasing from 33 to 50 mpg for the same number of miles driven. Larrick and Soll proposed "gallons per 100 miles" as an alternative measure of fuel economy that is linear in fuel consumption. The agencies used focus group testing to compare the different metrics and found that many participants preferred mpg due to its familiarity (U.S. EPA 2010a, 2010b). For the final label, the agencies kept the mpg metric, as required by law, but also included the gallons per 100 miles information in smaller print. In addition, the label prominently featured fuel cost savings compared to the average new vehicle, a highly relevant metric for consumers that allows for easy comparisons across vehicles.

4.4 Voluntary Programs

The EPA has sometimes used voluntary programs as an alternative to regulations to reduce emissions and other environmental hazards. Many EPA voluntary programs encourage polluting entities to go beyond what is mandated by regulation. Other voluntary programs address environmental quality in areas that may be regulated in the future but are currently not regulated.[31] Voluntary programs can offer the EPA the opportunity to pilot new approaches or to work with new industries before implementing a regulation with mandatory requirements.

The EPA typically designs voluntary programs through consultation with affected industries or consumers. In many cases, voluntary programs facilitate problem solving between the EPA and industry because information on practices that reduce pollutants and waste are shared through the consultative process. Voluntary programs also frequently encourage peer education and information sharing among participants. Data on abatement costs that are generated or disclosed through voluntary programs could help to inform future programs, analysis or regulatory action in the sector.

31 While this chapter only discusses EPA-led voluntary program, other government agencies, industry, non-profits and international organizations have also organized voluntary programs to address environmental issues.

Voluntary programs can have either broad environmental objectives targeting a variety of firms from different industries or focus on specific environmental problems relevant to a single industrial sector.[32] They often use one or more of the following four approaches:

- **Encourage firms or facilities to set specific environmental goals.** Goals can be EPA-specified, program-wide targets designed to provide a consistent objective across firms. In other cases, goals are qualitative and process-oriented so that a firm may set a unique target.
- **Promote firm environmental awareness and encourage process change within firms.** Programs designed to promote environmental awareness and process change often involve implementing a system to evaluate firms' operations and to provide information on new technologies. These programs may also promote or recognize use of third-party industry standards for products and materials.
- **Publicly recognize firm participation.** Voluntary programs that publicly recognize firm participation are designed to provide green consumers and investors with new information that may alter their consumption and investment patterns in favor of cleaner firms. Firms may also use their environmental achievements to differentiate their products from competitors' products.[33]
- **Use labeling to identify environmentally responsible products**. Product labeling can be applied to either intermediate inputs or final goods. Labels on intermediate goods encourage firms to purchase environmentally responsible inputs. Labels on final goods allow consumers to identify goods produced using a relatively clean production process. Section 4.3.4 and Text Box 4.3 discuss how labeling can be made more effective by using behavioral economics concepts.

The economics literature has not systematically evaluated the effectiveness of these four approaches. Like mandatory information disclosure programs, economic theory suggests that approaches involving sharing information among firms or labelling consumer products may be most useful in situations where imperfect or asymmetric information leads to adverse environmental outcomes.

Most empirical studies of EPA voluntary programs have focused on a few large, multi-sector programs such as 33/50, Green Lights and ENERGY STAR. They have found mixed evidence regarding the extent to which these programs have reduced emissions. Many studies failed to account for what would have occurred absent the program, potentially overstating reductions. The potential for beneficial information or technology spillovers from program participants to other firms in the target industry can make it difficult to measure a program's impact (Lyon and Maxwell 2007).[34] Many smaller regulatory programs remain unstudied.

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32 See Brouhle et al. (2005), Lyon and Maxwell (2007), Borck and Coglianese (2009), and Prakash and Potoski (2012) for discussions of how voluntary programs have been used in U.S. environmental policy.

33 See Konar and Cohen (2001), Videras and Alberini (2000), Brouhle, Griffiths and Wolverton (2005), Morgenstern and Pizer (2007), and Borck and Coglianese (2011) for more information on the main arguments for why firms participate in voluntary programs.

34 One thread of literature points to the role a regulatory threat plays in improving voluntary program effectiveness. When the threat of regulation is weak, abatement levels are lower. However, when the threat of regulation is strong, Segerson and Wu (2006) showed that levels achieved are closer to those that would be achieved under a standards-based approach. See also Morgenstern and Pizer (2007); Brouhle, Griffiths and

Key Advantages

- Voluntary programs allow agencies the opportunity to pilot new approaches to working with industries or on environmental problems not yet subject to regulation, which could be particularly useful if there is substantial uncertainty about the benefits or costs of regulation.
- Voluntary programs involving data gathering and reporting by program participants could yield new data useful for future analyses or regulatory actions.
- Voluntary programs with a labeling or information disclosure component could be well-suited to address market failures due to asymmetric or imperfect information.

Key Disadvantages

- If voluntary programs only attract participants that are already industry leaders in environmental protection, they may not yield significant improvements in environmental outcomes relative to a baseline without the voluntary program.
- Economic theory suggests that firms or individuals are unlikely to participate if the private costs of participating exceed the private benefits, even if the social net benefits of participating are positive.
- Empirical studies on the effectiveness of voluntary programs in improving environmental outcomes are limited, and the available evidence is mixed.

4.5 Cross-Cutting Issues When Comparing Regulatory and Non-Regulatory Approaches

Using a simplified theoretical framework, Fullerton (2001) demonstrated that a variety of regulatory and non-regulatory approaches can be designed to achieve the same level of economic efficiency.[35] In practice, there are likely to be important tradeoffs across approaches. Economic analysis can play an important role in identifying these tradeoffs.

Analysts can provide insight into the approaches that maximize net benefits and how they vary in efficiency over time. One regulatory feature that reduces economic efficiency is "grandfathering" — a practice in which older polluters are exempted from new regulations or are subjected to a less stringent standard than newer polluters. Grandfathering creates a bias against constructing new facilities and investing in new pollution control technology or production processes. As a result, grandfathered older facilities with higher emission rates tend to remain active longer than they would if the same emission standard applied to all polluters (Helfand, 1991; Stavins, 2006).

In general, varying regulatory requirements by firm age, size, location or other attributes can increase efficiency by helping to equalize heterogeneous abatement costs or benefits, but it can also reduce the efficiency of a policy if the design creates perverse incentives to shift production away from more regulated firms or products toward those that are less regulated. Further distortions are introduced when a small change in behavior results in a discrete change in regulatory requirements. Such discontinuities or "notches" incentivize firms to strategically avoid going over

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Wolverton (2009); Lange (2009); Vidovic and Khanna (2011); Kim and Lyon (2011); Brouhle, Graham and Harrington (2013); and Ferrara and Lange (2014).

35 Fullerton (2001) assumed no administrative costs, perfect information, no enforcement issues, perfect labor mobility and competitive firms.

the specified threshold to minimize the cost of compliance (Ito and Sallee 2018). For example, some policies exempt firms below a certain size, which can discourage firms from expanding or consolidating to avoid the regulation (Blinder and Rosen 1985; Sallee and Slemrod 2012). Chapter 3 provides more discussion of these policy design features.

There are several other cross-cutting issues that may be useful to analyze when evaluating potential tradeoffs across approaches. These include distributional impacts and environmental justice; administrative, monitoring and enforcement costs; interactions with other distortions; degree of flexibility; information requirements and uncertainty; and the nature of the environmental problem.[36] Analysts can evaluate these factors using methods discussed in Chapters 7 through 10. Stringency is another important consideration for regulatory design that is discussed in Chapter 3.

4.5.1 Distributional Impacts and Environmental Justice

The distribution of costs and benefits across firms, workers, governments, households and individuals over time and space is often of interest to decision makers.37 For example, market-based instruments that directly affect the price of the goods produced by polluting firms will likely have different distributional and environmental justice consequences than prescriptive regulations (Berck and Helfand 2005; Rozenberg et al. 2020; Zhao and Mattauch 2022). A commonly expressed concern is that market-based policies to reduce greenhouse gas emissions may lead to increases in other pollutants in already overburdened neighborhoods.[38]

The distribution of economic rents may also differ across approaches. If allowances are auctioned or sold to polluters, the distributional consequences of a cap-and-trade policy will be similar to those of emissions taxes. If allowances are instead distributed for free, distributional consequences will depend on the allocation approach (e.g., historical output or inputs; updating), who receives the allowances and the ability of the recipients to pass the costs onto their customers. Likewise, for approaches that raise revenue — such as emissions taxes or a cap-and-trade policy that auctions allowances — the way revenue is used will affect the distributional outcomes (Burtraw et al. 2010).[39]

36 Many of these criteria are also highlighted in Fullerton (2001). Another criterion discussed by Fullerton (2001) is political and ethical considerations. The approach ultimately chosen will also depend on statutory and other legal limitations. This chapter does not expand on these considerations because analysts have a limited role to play in evaluating them.

37 See Chapter 9 for approaches to quantify the economic impacts of approaches under consideration. See Chapter 10 for discussion of impacts on minority, low-income or Indigenous populations and on children and older adults.

38 In the context of California, research has not reached consensus on the degree to which its carbon cap-and-trade policy has exacerbated existing criteria air emissions in communities of color or low-income communities (e.g., Fowlie et al., 2012; Grainger and Ruangmas, 2017; Mansur and Sheriff, 2021; Hernandez-Cortez and Meng, 2023; Cushing et al., 2018).

39 To explicitly weight economic efficiency alongside distributional or environmental justice considerations, analysts would need to employ a social welfare function that aggregates welfare across individuals into a single value to allow an explicit ranking of different policy options (see Adler 2008, 2012). However, a social welfare function is based on a normative judgement, and while it makes the criteria explicit regarding how society

Differing treatment applied to sources based on age, size, location or other attributes can also affect the distribution of revenues, expenses and rents within the economy under both prescriptive and market-based approaches.

4.5.2 Administrative, Monitoring and Enforcement Costs

Analysts can help shed light on differences in the cost of administering, monitoring and enforcing the approaches under consideration. For instance, what are the costs and foreseeable challenges for ensuring compliance? Is pollution observable, or will it need to be estimated based on inputs and technology used? Are technologies available to decrease the costs of monitoring and reporting?

When pollutant emissions or concentrations can be easily measured, it is more feasible to directly regulate the level of the pollutant. For example, continuous emissions monitoring equipment at power plants allowed for direct measurement of pollution and facilitated the use of a cap-and trade system to regulate SO_2 emissions (see Text Box 4.2). If a source has fewer allowances than the monitored emission levels at the end of a compliance period, it is in noncompliance and the source must provide allowances to cover its environmental obligation and pay a penalty.[40]

If monitoring and enforcement costs are high, a regulation may fail to deliver environmental benefits due to widespread noncompliance (e.g., illegal dumping).[41] In these cases, it may be easier to regulate a related input or output to leverage approaches that incentivize sources to reveal information about their production or abatement processes (e.g., a tax and subsidy combination). Mandating the use of specific abatement technologies can sometimes reduce the cost of monitoring compliance, as noted in Section 4.1.1. In addition, it may be easier to monitor and enforce regulations on a smaller number of "upstream" sources (e.g., oil refineries) rather than a larger set of "downstream" sources (e.g., gasoline consumers) (Mansur 2012).

4.5.3 Interactions with Other Distortions

Analysts should consider the potential distortionary effects of any policy option considered. Even if a policy is relatively efficient on its own, it may interact with pre-existing environmental, trade, tax or agricultural policies in ways that exacerbate distortions in the economy and result in additional social costs. One such distortion occurs when imperfect competition due to market power results in lower output than would occur in a competitive market, which results in a loss in economic welfare. Policy instruments that cause firms to restrict output (e.g., an emissions tax) may create additional inefficiencies in sectors where firms have some degree of market power (Baumol and Oates 1988; Fowlie et al. 2016). A combination of market-based instruments may work more effectively than a single instrument in this instance.

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prefers to distribute resources across individuals, there is no consensus regarding those preferences. Thus, distributional information is typically analyzed and presented separate from efficiency considerations.

40 The U.S. Acid Rain Trading Program has high levels of compliance and requires fewer than 50 EPA staff to administer since penalties are automatically levied for each ton of excess emissions (Napolitano et al., 2007).

41 However, Sigman (2012) presented a theoretical model showing that compliance need not decrease when regulations are broadened beyond industries with low-cost monitoring to include those where monitoring costs are higher, but abatement costs are lower.

If costs differ between existing and new firms, the use of certain instruments may cause a change in market structure that favors existing firms by creating barriers to entry and allowing existing firms a certain amount of control over price. Cap-and-trade systems that set aside a certain number of allowances for new firms may guard against such barriers.

Instruments that involve the government collecting revenue, such as auctioned allowances or taxes, may create opportunities to reduce distortions.[42] At the same time, society also incurs a welfare loss from raising revenues through taxes due to the difference between the value of an additional dollar raised by the government and the value of that dollar to a private individual (termed the marginal cost of public funds). See Chapter 8 and Appendix A for more discussion of analyzing welfare effects under sectoral or economy-wide distortions.

4.5.4 Degree of Flexibility and Dynamic Adjustment

Even if a regulation is set at an economically efficient level at the outset, changing conditions over time can result in inefficient levels of pollution control. To what extent does the approach allow for automatic adjustments in requirements or stringency over time in response to new information or technological improvements? Is the approach flexible enough to accommodate transition costs? Does the approach encourage innovation in abatement techniques that decrease the cost of compliance with environmental regulations over time?[43]

For instance, market-based approaches often differ from prescriptive approaches to regulation by encouraging firms to find the cheapest way to reduce emissions. The incentive to innovate means that the marginal abatement cost curve may shift downward over time as cheaper compliance options become available. If innovation causes the cost of pollution control to fall, the marginal cost of decreasing pollution levels could drop below the marginal benefit. A cap-and-trade approach incorporating a price floor below which allowances are removed from the market is one approach to dynamically adjusting a regulation. Similarly, a price ceiling above which additional allowances are introduced to the market can be used to ensure that marginal costs do not rise too far above marginal benefits (Fell et al. 2012). Features such as banking and borrowing also afford regulated plants some flexibility in the timing of reductions.

4.5.5 Information Requirements and Effects on Uncertainty

What information is required to implement the approach? How well does the approach perform under imperfect or asymmetric information, or when there is uncertainty about costs and/or benefits? Can the approach be designed in a way that will reveal new information about costs and benefits that can reduce uncertainty if additional analysis or regulatory action is considered in the future?

When abatement costs and benefits are certain, price-based instruments (e.g., emissions taxes) and quantity-based instruments (e.g., cap-and-trade) are theoretically equivalent and can be designed to achieve the same outcomes. However, this result may not hold when there is uncertainty about

42 For more information on the how revenues raised via market-based instruments affect social welfare, see Bovenberg and Goulder (1996), Goulder (2013), Jorgenson et al. (2013), and McKibbin et al. (2015).

43 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), Kerr and Newell (2003), Requate (2005), and Newell (2010).

the benefits and costs of pollution control, or when marginal benefits and costs change substantially with the stringency of the pollution control target (Weitzman 1974).[44] If uncertainty associated with the abatement costs exists but damages do not change much with additional pollution, then policymakers can limit costs by using a price instrument without having much impact on the benefits of the policy. If, on the other hand, there is more uncertainty associated with the benefits of controlling pollution, and policymakers wish to guard against high environmental damages, a quantity instrument is preferable. In some circumstances, this may come to resemble a more prescriptive approach that specifies zero allowable source-level emissions to avoid potentially costly or damaging mistakes. Hybrid approaches that combine features of price and quantity instruments or add other flexibilities to address uncertainty can also vary with respect to dynamic efficiency (Pizer 2002; Weitzman 2020; see Section 4.3.1.3).

Other types of regulatory and non-regulatory approaches may reveal information about emissions, abatement approaches, or abatement costs that can facilitate both retrospective analysis to understand how well an approach is working and prospective analysis of potential future regulatory actions.[45] Monitoring and reporting requirements can be used to compel regulated entities to release data on emissions or abatement approaches. Allowance trading systems can reveal information to regulators and the public about abatement costs because the equilibrium allowance price indicates the marginal cost of compliance with the regulation. Subsidy programs may also require participants to reveal information about abatement activities.

In some instances where there is a high level of uncertainty about costs and/or benefits, a voluntary program or pilot project may be a compelling alternative to regulation. Such an approach encourages environmental improvements but allows the government and regulated community to test out different abatement technologies or process changes and gather information on what works and what does not. Research and development efforts may also contribute to better understanding the costs and benefits of regulating. As technology improves or more data become available, analysts will be better able to analyze a variety of approaches. Value of information analysis could be used to examine whether more resources should be invested to reduce uncertainty before developing a regulation and which aspects of uncertainty to prioritize (Finkel and Evans 1987).[46]

4.5.6 The Nature of the Environmental Problem

Another important issue is the type of environmental problem being addressed. Are the sources heterogeneous? Does the pollutant vary across time and space? Do emissions derive from a point source or a nonpoint source? Do the pollutants persist in the environment or dissipate rapidly?[47] Point sources, which emit at identifiable and specific locations, are typically easier to control than

44 Pezzey and Jotzo (2012) built on Weitzman (1974) by examining how revenue recycling affects the welfare implications of a price- versus quantity-based market instrument under uncertainty.

45 Chapter 5 (Text Box 5.1) provides more discussion of retrospective analysis.

46 For more discussion and examples of value of information analysis in environmental policy, see Cullen and Frey (1999), Dekay et al. (2002), Keisler et al. (2014), Marchese et al. (2018), Thompson and Evans (1997), and Yokota and Thompson (2004).

47 For a more discussion of how the nature of the environmental problem affects instrument choice, see Kahn (2005); Goulder and Parry (2008); Parry and Williams (1999); Sterner and Coria (2012); Tietenberg and Lewis (2014); and Xabadia, Goetz and Zilberman (2008).

diffuse, numerous nonpoint sources and are often responsive to a variety of approaches. Monitoring and control of nonpoint source emissions are more challenging (see Text Box 4.1). In instances where both point and nonpoint sources contribute to a pollution problem, a case can be made for a tax-subsidy combination (with taxes directed toward point sources and subsidies to nonpoint sources) or an allowance trading system with offsets.

Flow pollutants that dissipate quickly are responsive to a wide variety of market and hybrid instruments. In contrast, stock pollutants that persist in the environment may require strict limits to prevent bioaccumulation or detrimental health effects at small doses, making direct regulation appealing. Approaches that set a limit on the overall quantity of pollution may be also preferred if there are discontinuities or threshold values above which sudden or large changes in environmental damages could occur (Pindyck 2007). For pollutants that do not mix uniformly, it is important to account for differences in baseline pollution levels and in emissions across more- and less-polluted areas. Damages can also vary by time of day or season. For example, health impacts associated with vehicle emissions may be larger during rush hour because 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.

Chapter 4 References

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Chapter 5 - Setting the Foundation: Scope, Baseline and Other Analytic Design Considerations

This chapter provides an overview of a broad set of issues related to the design of an economic analysis. These include (1) the appropriate scope of a benefit-cost analysis (BCA), (2) how to specify the baseline, (3) how to account for behavioral and technological change, (4) what to assume about regulatory compliance and (5) how to address analytic uncertainty, among others. Identifying key issues or questions surrounding these decisions early in the regulatory development process is important because they can have a profound impact on analytic outcomes. Subsequent chapters on benefits (Chapter 7), costs (Chapter 8), economic impacts (Chapter 9) and environmental justice and other distributional analyses (Chapter 10) delve into these considerations in more depth. The discussion of analytic design considerations focuses on their application to prospective analyses, though they are equally applicable to retrospective analysis of existing regulations (see Text Box 5.1 for more discussion).

5.1 Scope of Analysis

Several early analytic decisions determine the scope of a BCA of a regulation. Specifically, analysts must consider whose costs and benefits to count in a regulatory analysis and the types of markets and non-market effects that should be evaluated, including those that cannot be quantified.

A comprehensive approach to benefit-cost analysis is required to assess whether it is conceivable for those who experience a net gain from a regulatory action to potentially compensate those who experience a net loss.[1] These benefits and costs may occur in private markets as well as through changes in externalities. Analysts should carefully consider how various benefits and costs may materialize as a result of the regulatory action by looking beyond effects on regulated entities and changes in the regulated contaminant(s). Without a comprehensive accounting of benefits and costs, the analysis may provide misleading conclusions regarding the sign and magnitude of net benefits and the relative rankings of the analyzed regulatory options (Farrow 2013).[2] As discussed

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1 These gains and losses are measured by an individual's willingness to pay or willingness to accept. See Section A-3 for a discussion of the Kaldor-Hicks potential compensation test that underlies the economic practice of BCA.

2 EO 12866 and OMB's Circular A-4 (2023) require and affirm that all benefits and costs resulting from a policy change should be considered in a BCA. For example, Circular A-4 states, "Your analysis should look beyond the obvious benefits and costs of your regulation and consider any important additional benefits or costs, when in later chapters, the BCA also should clearly identify each source of benefits and costs and present it in a disaggregated and informative way (see Chapter 11).

While in principle the analyst should account for all benefits and costs, in practice, not all changes in economic welfare can be quantified and monetized due to limitations in tools, data and resources. In these cases, analysts are advised to prioritize quantifying those effects that are likely to have the greatest influence on net benefits and the relative ranking of the options under consideration. Since the results of a BCA are therefore likely incomplete, they should be presented and interpreted with care. The BCA should identify effects that could not be quantified or monetized (along with an explanation of why they were not included), describe evidence on the potential magnitude of the benefits and costs from these effects, and explicitly document and discuss any other analytic limitations and omissions. Furthermore, equal effort should be made to account for both benefits and costs so the analysis provides an assessment of net benefits that is balanced and as accurate as possible. While this section provides guidance on the scope of a BCA, Chapters 9 and 10 provide guidance on the scope of economic impact and distributional analysis.[3]

5.1.1 Standing

One of the first scoping questions an analyst must answer is, who has economic standing? Or put another way, whose gains and losses should be accounted for in the analysis? The most inclusive answer is all persons who may be affected by the policy regardless of where (or when) they live. Regulatory analysis often focuses on the costs that accrue to regulated sources, regardless of the nationality of the owners of affected physical assets, and the benefits to individuals that reside within the country's national boundaries. This approach reflects the fact that these are the two groups primarily affected by most regulations.[4]

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feasible. An additional benefit may be a favorable regulation that is unrelated to the main purpose of the regulation..., while an additional cost may be an adverse impact...that occurs due to a regulation and is not already accounted for in the obvious costs of the regulation. These sorts of effects sometimes are referred to by other names: for example, indirect or ancillary benefits and costs, co-benefits, or countervailing risks."

3 While Section 5.1 focuses on the scope of a BCA, the same set of issues applies broadly to economic impact and distributional analysis. An exception is that it may be worthwhile to estimate certain welfare effects for a distributional analysis even when those effects do not fundamentally change the net benefits of regulatory options under consideration.

4 Regulations often only apply to activities within a national border by residents and firms who have consented to adhere to the same set of rules and values for collective decision-making. In addition, most domestic policies are expected to have relatively negligible effects on other countries (Gayer and Viscusi, 2016; Kopp et al. 1997; Whittington et al. 1986).

Text Box 5.1 - Retrospective Analysis

The principles for prospective analysis also apply to estimating benefits, costs, or economic impacts from existing regulations. A retrospective analysis can provide an opportunity to understand whether a regulation has achieved its objectives — for example, whether the regulation improved societal welfare as expected. Retrospective analysis may identify compliance pathways, behavioral responses, or consequences that may not have been fully anticipated at rule promulgation. Retrospective analyses may also suggest ways to improve prospective analysis — for instance, if certain consequences of regulation are routinely underestimated ex-ante, methods to anticipate these effects may be developed. Ultimately, retrospective analysis may result in improvements in regulatory design.

While the importance of retrospective analysis in policy evaluation and regulatory reform is well-recognized, ex-post studies of EPA regulations are relatively rare (U.S. EPA 2014; Aldy 2014; Morgenstern 2018; Fraas et al 2023). Absent systematic data collection, retrospective analyses of the benefits, costs, or economic impacts of regulations have been conducted opportunistically (Fraas et al 2023; Aldy et al 2022; Cropper et al 2018). In addition, retrospective assessments have struggled with issues such as "how to evaluate a highly heterogeneous industry with a limited set of information, how to form a reasonable counterfactual, and how to disentangle the costs [or benefits] of compliance from other factors" (U.S. EPA, 2014). Another challenge has been identifying metrics that can be measured ex post that are relevant to the regulatory outcomes of interest (Morgenstern 2018).

Because of the many challenges inherent in conducting robust retrospective analysis, studies of EPA regulations are often relatively narrow in scope in that they only evaluate a subset of the questions of interest. For example, a study may examine how emissions have changed post-regulation but due to data limitations, may not evaluate the extent to which changes in risk or health outcomes have occurred. Likewise, researchers may identify the mix of compliance strategies that were used or offer insights into specific aspects of unit costs but not have enough information to assess their costs in aggregate (Fraas et al 2023).

Given sufficient data, analysts can use a variety of techniques to conduct rigorous retrospective review. One approach is to use statistical techniques to control for other exogenous factors that affected firm or consumer behavior over time. If a set of similar facilities remained unregulated over the time period, then it may also be possible to compare the regulated firms' behavior to a reasonable counterfactual. If data for several years before and after the regulation became effective is available, it may also be possible to analyze how benefits or costs changed over time. This would potentially allow one to evaluate whether a regulation induced technological change or affected employment, for example. Though used less in published retrospective analysis, another approach is to use computational models to address statistical and data challenges. Even when the model chosen is scientifically defensible, fit for purpose, appropriately parameterized and reasonably transparent, separating out the effects of the regulation from other changes that would have occurred anyway (i.e., in the counterfactual) is still a challenge.

The EPA is exploring additional steps to better institutionalize the practice of conducting retrospective review and analysis. For example, this could be through the development of a systematic approach to identifying the types of rules most amenable to retrospective analysis, best practices for retrospective analysis, and how to identify analytic requirements for such analysis. Data needs could be identified and avenues for ex-post data collection integrated into the regulation (while also accounting for the cost and time needed for firms to collect such information). In this way, the EPA could learn from past experience and improve both policy designs in future regulatory actions and analytic approaches in future prospective analyses.

In certain contexts, however, it may be important to include effects beyond national boundaries. This is particularly relevant to consider when evaluating a regulation's impact on a global public good.[5] It is also important to be cognizant of analytic challenges when attempting to disaggregate benefits and costs accruing to domestic and foreign citizens and residents. For example, to limit economic standing to citizens and residents of the United States, one may need to determine how to treat multinational firms with plants in the United States but shareholders elsewhere and how to estimate the extent to which impacts on foreign companies and citizens have feedback effects on U.S. citizens and residents.[6]

The basis for the decision about the scope of the analysis should be transparent and clear and should focus on capturing the significant effects of a regulation. Analysts should ensure that the application is supported by the available data and that standing is consistently applied when estimating costs and benefits; in other words, if a group has standing for estimating costs, it should also have standing for benefit estimation.

5.1.2 Market Effects

Another scoping question is: which markets will be affected by the regulation? The ways in which a regulation may affect different markets helps inform the analytic approach to take (see Chapters 7 and 8 for more discussion). Ideally, the analyst should comprehensively capture all costs and benefits of a regulation. In practice, this may not always be feasible due to limitations in available data, methodologies, or resources. When prioritizing which costs and benefits to include, consider the effect of the regulation on related markets.

A "distorted" market is one in which factors such as pre-existing taxes, externalities, regulations, or imperfectly competitive markets move consumers or firms away from what would occur under perfect competition.[7] In the absence of market distortions, focusing on the impacts within the market may be sufficient. While a policy may have effects on other markets, market-clearing conditions ensure that they are effectively canceled out from an aggregate welfare perspective (Farrow and Rose 2018; Just et al. 2004).

Every market is distorted to some degree. In particular, effects in related markets are important to consider when there are both pre-existing distortions in these markets and there are significant

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5 For example, when emissions of a pollutant contribute to damages around the world regardless of where they are emitted, it is important to consider how U.S. mitigation activities may affect international reciprocity and cooperation in addressing the same pollutant, as any international mitigation actions will provide a benefit to U.S. citizens and residents. There may also be cases where international or domestic legal obligations require or support calculation of regulatory effects accruing beyond national boundaries. For more discussion of when the effects of U.S. policy on non-residents might be relevant in regulatory analysis, see OMB (2023).

6 For example, impacts that occur outside U.S. borders can impact the welfare of individuals and the profits of firms that reside in the U.S. because of their connection to the global economy. This can occur through effects on supply chains, international markets, trade, tourism, and other activities. Other challenges might include how to account for leakage due to regulatory requirements that are not harmonized across countries or how to treat impacts on U.S. citizens or assets residing outside U.S. borders. See National Academies (2017) for a detailed discussion of challenges in the context of quantifying the effects of changes in greenhouse gas emissions.

7 Perfectly competitive markets are characterized by the following conditions: all economic agents have complete information; there are no barriers to entry or exit; firms have constant returns to scale; and there are no taxes, subsidies or policies that create a wedge between the price suppliers receive for a good and the price consumers pay for it. The term "externality" is discussed in Chapter 3.

cross-price effects between the regulated sector and these other economic sectors (Harberger 1964; Boardman et al. 2018; U.S. EPA 2017). Related markets may include those for major inputs to the regulated sector, products that use the regulated sector's output as an input, and products that are substitutes or complements to the regulated sector's output. A key question for the analyst to consider given market distortions is: when is it reasonable to assume away these effects (e.g., Hahn and Hird 1990)? Evidence suggests that effects outside of the regulated sector, and therefore changes in welfare, may be substantial even with a relatively small sector-specific regulation (Marten et al. 2019; Goulder and Williams 2003). The presence of a distortion alone, however, may not warrant a broader analytic approach, particularly if the value of information from accounting for its effect on costs and benefits is relatively small. Analysts should take special care to justify their choice of which markets to explicitly analyze as part of the regulatory analysis and identify key assumptions and limitations underlying this choice.[8,9]

5.1.3 Externalities

BCA should aim to comprehensively evaluate all benefits and costs resulting from the regulation, which includes welfare effects from all changes in externalities due to changes in environmental contaminants as well as any other externalities.[10] If some of these effects cannot be quantified or monetized, they should be evaluated qualitatively (including a discussion of their potential magnitude).

Welfare effects from changes in externalities could be favorable or adverse. Analogous to how a regulation's interactions with existing market distortions (e.g., pre-existing taxes, asymmetric information) could lead to additional social costs, a regulation could ameliorate or exacerbate other pre-existing externalities. Changes in other environmental contaminants may arise from the compliance methods of regulated sources. For example, the use of an abatement technology by regulated sources to reduce emissions of a pollutant into one medium (e.g., air) may change the emissions of another pollutant into the same medium (e.g., from the same smokestack) or cause changes in emissions of pollutants into another medium (e.g., water).

Changes in other environmental contaminants may also occur as a result of market interactions induced by the regulation. For example, more stringent vehicle emissions standards can lead to changes in upstream oil refinery emissions. Section 5.5.6 discusses the importance of ensuring that projected changes in contaminants are consistent with expected market behavior, considers

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8 Analysts should also keep in mind that even in cases where effects in other sectors contribute little to the overall social cost or benefits of the policy, they may have important distributional consequences that warrant a broader analytic treatment than one that focuses solely on the directly regulated market. See Chapters 9 and 10 for more discussion.

9 Choosing the model that is most appropriate for capturing the key impacts of a policy is sometimes referred to as "horses for courses." Just as the best horse for a race depends on the features of the course, the best economic model(s) to evaluate the benefits and costs of a regulation depend on the features of the regulation and the affected markets. Text Box 5.3 discusses model selection criteria more generally.

10 These effects are among the distortions discussed in Section 5.1.2 as the presence of an externality represents a deviation from perfect and complete markets. Such a deviation may be ameliorated or exacerbated by behavioral changes induced by a regulation. The costs and benefits from unaddressed externalities differ from the costs and benefits of the production of marketed goods in that the welfare effects due to changes in an externality are not reflected in the market prices of those sectors and activities that cause the externality. See Chapter 3 for further discussion of externalities.

interactions with other regulations and provides several common examples of how changes in other contaminants arise in analyses of U.S. Environmental Protection (EPA) regulations. This guidance also applies to expected changes in externalities other than those associated with environmental contaminants. For example, changes in vehicle emissions standards may reduce the marginal cost of driving due to greater fuel efficiency and lead to an increase in vehicle miles traveled that affects road safety, congestion, and other transport-related externalities. These welfare effects should also be accounted for in the BCA and, if they cannot be accounted for because of limited resources, data, and other limitations, they should be described qualitatively.

When presenting the results of the BCA, identifying benefits and costs that are specifically contemplated by the statutory provision under which the regulation is being promulgated — when it is possible to do so — provides transparency. For example, in a BCA of a regulation promulgated under a Clean Air Act provision whose objective is reducing hazardous air pollutants (HAPs), it is helpful to clearly distinguish the air pollution benefits resulting from reductions in HAP emissions from other welfare effects resulting from the expected compliance strategies of regulated entities.[11] Yet, when calculating net benefits all welfare effects should be included, as it is the total willingness to pay for all changes induced by a regulation that determines whether the regulation increases economic efficiency.

5.2 Baseline

Establishing the baseline of an economic analysis is a critical step for accurately evaluating benefits and costs. Because a BCA considers the impact of a policy or regulation in relation to the baseline, its specification can have a profound influence on the results of the analysis. The level of detail presented in the baseline is also an important determinant of the type of analysis that can be conducted when evaluating regulatory options.

5.2.1 Baseline Definition

The baseline is defined as the best assessment of the way the world would evolve absent the proposed regulation. It is the primary point of comparison for assessing the effects of the regulatory options under consideration. Specifically, the BCA models two states of the world: the expected state without the regulation (the baseline scenario) and the expected state with the proposed regulation in effect (the policy scenario(s)). The effects of each policy scenario are measured by examining the differences in net benefits between the scenarios and the baseline.

The baseline describes the expected future of the environmental problem and level of environmental contaminants along with the affected markets and population in the absence of the proposed regulation. While the policy scenario is described in a similar fashion to the baseline, it reflects different environmental and/or market outcomes.

Figure 5.1 illustrates the difference between the baseline and a policy scenario, although there may be multiple policy scenarios under consideration. An economic analysis begins with a description of the state of the world in the current period as a foundation before any analytic scenarios are constructed. The current state of the world includes a description of the environmental problem as

11 This means that if the air pollution reduction also reduces harmful deposition of the pollutant into the water, the benefits from reducing water pollution should be distinguished from the benefits arising from the reduction of the pollutant in the air.

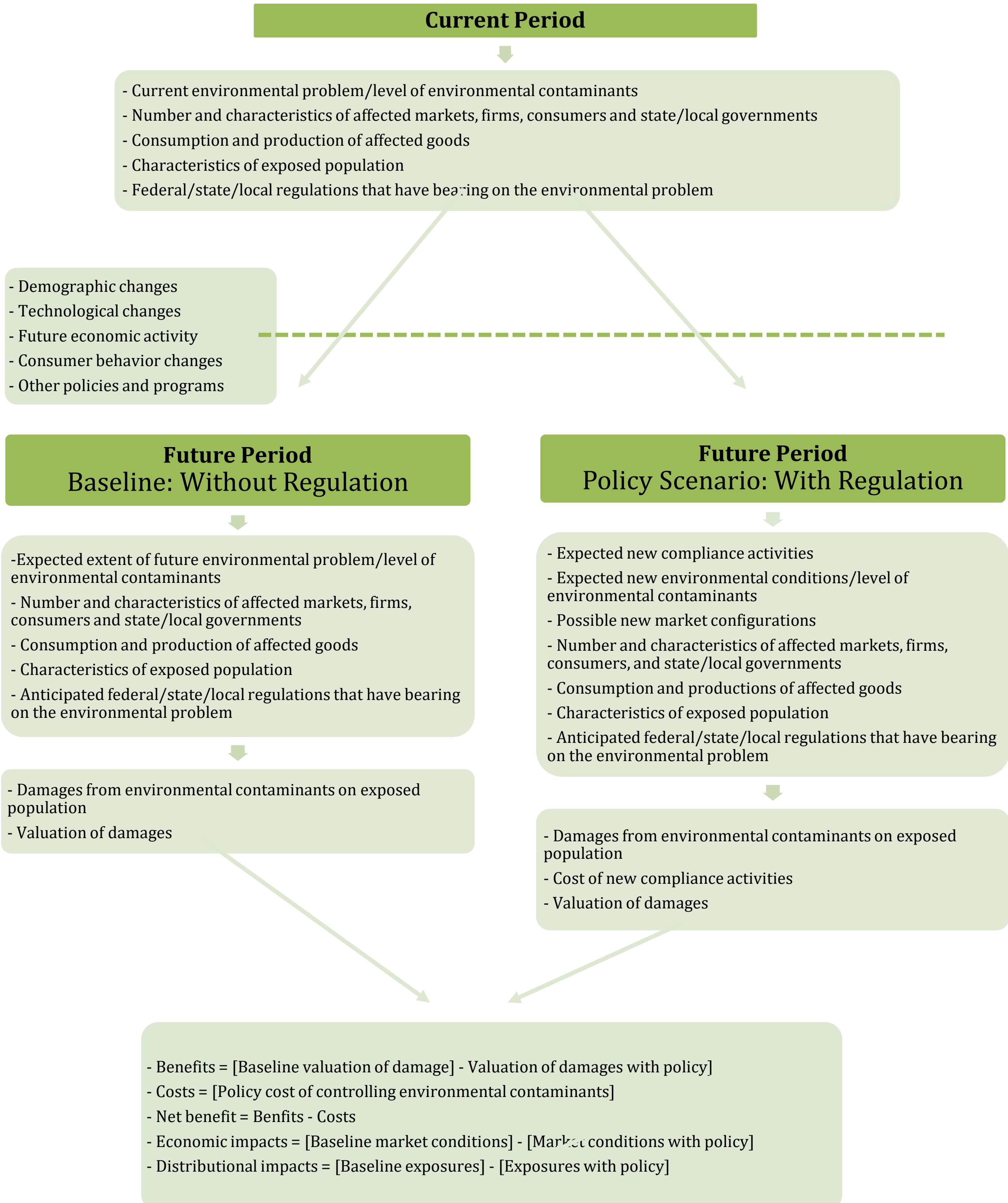
well as other variables such as the level of environmental contaminants; the number and characteristics of the affected markets, firms, consumers and state and local governments; the consumption and production of affected goods within and beyond the regulated market; characteristics of the exposed or otherwise affected population; and existing federal, state and local regulations that may affect the environmental problem. Based on the description of the current state of the world, the next step is to develop a projection of the future state of the world without the regulation, which is referred to as the baseline. This step is done by characterizing how economic and environmental conditions are expected to change over time. Changes may occur in demographics, the pace and direction of technology, energy and other prices, sector-specific economic activity, consumer behavior and other related policies and programs that are already in place. The baseline should reflect likely outcomes, or "business as usual" — not an outlier scenario. The policy scenario is evaluated in a similar fashion, but the economic and environmental conditions reflect the future state of the world with the regulation in place. The two scenarios are then compared.

It is important to note that the comparison of the world with the policy, to the world without the policy is distinct — and quite different — from a comparison of the state of the world before the action to the state of the world after the action. In other words, the baseline is a future scenario without the regulatory program under consideration; it is not a scenario assuming no change from current conditions. The economy and other factors may change over the time horizon of analysis even in the absence of regulation, so a proper baseline should incorporate assumptions about the changes in the economy that may affect relevant benefits and costs.

In most cases, future economic and environmental conditions in the baseline are expected to have changed solely in response to factors unrelated to the regulation under consideration. On occasion this may not be the case. For example, a regulation under consideration may extend the compliance period of an existing regulation. In this case, the baseline specification might incorporate the expiration of the existing program. However, changes between the baseline and policy scenario should be solely attributable to the introduction of the regulation. The economic and environmental characteristics specified in the baseline should be used in the policy scenario unless the policy scenario is anticipated to change those characteristics. This is what makes the baseline the relevant point of comparison for the policy. In general, the construction of the baseline needs to be balanced to equally identify factors that may meaningfully affect both benefits and costs. For example, the analyst should not assume a great deal of technological innovation in one sector (e.g., the pollution abatement sector) and ignore potential technology improvements in other sectors.

The final step in an analysis, as illustrated in Figure 5.1, is to use the information from the baseline and policy scenarios as a basis for estimating the benefits, costs, economic impacts, and distributional impacts of the regulatory option(s) under consideration. The damages from exposure to environmental contaminant levels in the baseline and policy scenarios can be valued using appropriate economic techniques (see Chapter 7: Analyzing Benefits). The value of the change in damages in the policy scenario are the benefits of the policy. The new compliance activities and other effects identified in the policy scenario can be used to quantify the costs of the policy (see Chapter 8: Analyzing Costs). The figure provides examples of economic and distributional impacts that may occur (for additional examples and explanation, see Chapters 9 and 10).

Figure 5.1 - Structure of a Benefit-Cost Analysis



5.2.2 Guiding Principles of Baseline Specification

In specifying the baseline, analysts should employ the following guiding principles:

1. Clearly specify the environmental problem that the regulation addresses and the regulatory approach being considered in the statement of need.
2. Identify all required variables for the analysis.
3. Clearly specify the current and future state of relevant economic and regulatory variables.
4. Focus on the components of the analysis that have the greatest influence on the results.
5. Clearly identify all assumptions made in specifying the baseline conditions.
6. Detail all aspects of the baseline specification that are uncertain.
7. Use the baseline assumptions consistently throughout the analysis of a regulation.

Though these principles exhibit a common-sense approach to baseline specification, the analyst is advised to provide statements on each of these points. Failure to do so may result in a confusing presentation and misinterpretation of the economic results.

Clearly specify the environmental problem that the regulation addresses and the regulatory approach being considered in the statement of need. As discussed in Chapter 3, the analysis should begin with a statement of need for regulatory action and an evaluation of policy options. The statement of need provides a description of the problem being addressed and the significance of that problem, the failures of private markets or public institutions that warrant agency action, and an assessment of whether Federal regulation is the best way to correct the problem. This statement should also include a description of the current regulatory environment and the regulated entities and other affected parties. The evaluation of policy options should describe all policy options or potential regulatory or non-regulatory approaches that were considered and how they were chosen.[12] The statement of need and description of the policy options will help clarify the appropriate baseline to be used.

In general, the baseline will assume no change in behavior to comply with the new regulation or existing regulations; but in some cases, a different baseline may be considered. For example, if an industry is *certain* to be regulated by some other method (e.g., by court order or state action) but that regulation has not yet been implemented, then the baseline should include it. Also, it is common practice to assume full compliance with existing regulatory requirements in the baseline even if there is noncompliance, although a separate analysis assuming less-than-full compliance may determine the implication of this assumption (see Sections 5.5.4 and 5.6.1 for more discussion of this issue).

Identify all required variables 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 regulation to economic behavior to environmental changes to impacts on humans or ecosystems. The models, parameters and variables required for the baseline analysis should be chosen so that they can inform all subsequent analyses.

Differences between the baseline and policy scenario may include changes in use or production of toxic substances, production processes and costs, pollutant emissions and ambient concentrations, and incidence rates for adverse health and environmental outcomes associated with exposure to pollutants. This does not mean that the analyst must identify all the variables that could possibly change, but the analyst should recognize all relevant variables needed to compare the baseline

12 See Chapter 4 for a description of various regulatory and non-regulatory approaches.

scenario to the policy scenario. At a minimum, the analyst should identify the variables that are expected to have the largest impact on costs and benefits within and across policy options.

Specify the current and future state of relevant economic and regulatory variables. Future baseline trajectories of certain types of economic variables such as energy prices, the level and growth of economic activity and population growth may be important for modeling the effects of a regulation. Even small changes in the rate of economic growth may, over time, result in considerable differences in emissions and control costs. Assuming no change in the baseline economic activity may lead to incorrect results.[13] Likewise, assumptions about the future growth and age distribution of the population affected by a regulation are important for predicting the number of individuals exposed or even the magnitude of aggregate damages. Other variables, such as broad trends in consumer spending patterns and technological growth, are also important for modeling the effects of a regulation but are more difficult to estimate. In these cases, the analyst should specify the baseline levels for these variables and changes over time and explicitly discuss all assumptions. If other policies or programs influence baseline conditions, they should also be in the baseline. For example, changes in farm subsidy programs may influence future pesticide use. Accounting for the way existing regulations affect compliance behavior and economic and environmental outcomes of a new regulation assures that the BCA properly accounts for the cumulative effects of all relevant regulations. In an ideal analysis, all potential influences on baseline conditions, and on the costs and benefits of policy options, would be examined and estimated. However, it is up to the analyst to determine if these influences warrant consideration in the regulatory analysis (e.g., because they may change the rank ordering of the analyzed options). If certain influences are known but not considered significant enough to be included in the quantitative analysis, they should be discussed qualitatively. However, in certain circumstances it may be worthwhile to quantify them to confirm or demonstrate that they are small.

Concentrate on the components that have the greatest influence on the results. The analyst should concentrate analytic efforts on 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 conducting the analysis, and the availability of models and data. If several components of the baseline are uncertain, the analyst should concentrate on components that have the greatest influence on the costs and/or benefits and can be refined through additional analysis or data collection. Analysts should pay special attention to the components that will be used to calculate costs and benefits and those that are important in the evaluation and selection of a policy option.

Identify all assumptions made in specifying the baseline conditions. The analyst should explain key assumptions in detail, including those related to changes in consumer and producer behavior, and how these trends may be affected by the regulatory options. Analysts should look for trends in economic activity or pollution control technologies that occur for reasons unrelated to environmental regulation. For example, as a consumer's income increases over time, 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 expected to have significant influence on the evaluation of regulatory alternatives, the analyst should explain and identify the assumptions used

13 For example, if the regulated industry is in significant decline, or is moving overseas, this information should be accounted for in the baseline. In such cases, incremental costs to the regulated community (and corresponding benefits from the regulation) are likely to be less than if the targeted industry were stable or growing.

in the analysis, with the goal of laying out the assumptions so that other analysts (with access to the appropriate models) would be able to replicate the baseline specification.

Detail all aspects of the baseline that are uncertain. Because the analyst does not have perfect foresight, baseline conditions cannot be characterized with certainty. 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 used and referenced. In general, uncertainties underlying the baseline conditions should be treated in the same way as other types of uncertainties in the analysis.

It is also important to discuss information that was not included in the analysis due to scientific uncertainty. For example, a regulated pollutant may have a suspected health or ecological effect but no available human dose-response function. In this case, the effects generally are not quantified in the analysis, but why the effects were excluded should be discussed — especially if the expected magnitude is such that it could significantly affect the net benefit calculation. Analysts should also explain how scientific uncertainty affects model choice and parameter values. Important aspects of the analysis which are not included in the baseline due to scientific uncertainty should be included in an uncertainty section(s) of the analysis (see Section 5.6 below). Significant uncertainty in important variables may require the construction of alternative baselines (discussed below). While sensitivity analysis is usually a better choice, multiple baselines may provide insights when evaluating different policy options.

Use the baseline assumptions consistently for all analyses of a regulation. The economic and environmental characteristics used in the baseline should be consistent with those used for the policy scenario(s). For example, the calculation of both costs and benefits should draw upon estimates derived using the same underlying assumptions about future economic and environmental conditions. If the benefits and costs are derived using multiple economic and environmental models, then the baseline conditions applied in those models should be compared to ensure that they are consistent. Likewise, when comparing and ranking alternative regulatory options, comparison to the same baseline should be used for all options under consideration.[14]

In some cases, it may be useful to single out a sector for more detailed analysis, or a follow-on analysis might be needed to assess impacts on a specific set of households based on their socioeconomic characteristics, region, or sector. In this case, it may not be possible to specify a baseline that is fully consistent with the primary analysis, but the analyst should endeavor to make them as similar as possible. The analyst also should explicitly describe the differences between the two baselines and any uncertainty associated with them.

Use consistent dollar years across the baseline and policy scenarios. The baseline and policy scenarios should be presented consistently and should use a recent common dollar year throughout the analysis. The dollar year is the year to which the purchasing power of a dollar is indexed. This is important because inflation decreases the purchasing power of money. So, if costs and benefits are reported in 2022 dollars, for example, this means that the value of those costs and benefits are denominated to be comparable to market prices in 2022. All nominal values, which are those not adjusted for inflation, should be converted to real values by adjusting them to the same dollar year

14 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.

using an appropriate index of inflation, and the index(es) used should be explicitly stated.[15] Similarly, if the costs in an analysis are reported in a particular dollar year (e.g., 2020 dollars) but the benefits are reported in a different dollar year (e.g., 2022 dollars), one of the estimates should be adjusted for inflation so that they are reported in the same dollar year. The choice of dollar year should always be made clear. In addition, the reporting year for annual costs and benefits, distinct from the dollar year, should be made clear in both the text and tables. For example, if an economic analysis is using a 2022 dollar year, but the costs and benefits for the rule are reported for the year 2024, both the text and tables should be clear that the values are for 2024, in 2022 dollars.

5.2.3 Multiple Baselines

In most cases, a single, well-defined baseline is generally all that is needed as a point of comparison. However, there are a few situations where it may be informative to compare the policy options to more than one baseline. Multiple baseline scenarios are needed when it is difficult to identify a single, reasonable description of the world in the absence of the proposed regulation. For instance, if the current level of compliance with existing regulations is not known and may substantially influence the net benefits, then it may be necessary to compare the policy scenario to both a full compliance baseline (the standard assumption) as well as a partial compliance baseline. Also, if the impact of other rules currently under consideration fundamentally affects the analysis of the rule being analyzed, then multiple scenarios with and without these rules in the baseline may be necessary. For example, for the 2019 rule to repeal the 2015 rule defining "Waters of the United States," the degree to which states would continue to regulate their waters at the 2015 standard was uncertain. Since the states' decisions dramatically affected the avoided costs and forgone benefits of the repeal, multiple baselines were used to illustrate the range of potential impacts (U.S. EPA 2019).

The decision to include multiple baselines should not be taken lightly since it may result in a complex set of modeling choices and analytic findings. Multiple baselines increase the possibility of erroneous comparisons of costs and benefits if the modeling choices and results are not communicated clearly. The number of baselines should be limited but still cover the key dimensions of the analysis and any phenomena in the baseline that are uncertain. Each baseline-to-policy comparison should be internally consistent in its definition and use of baseline assumptions.

5.3 Multiple Rules

Although regulations that have been finalized clearly belong in the baseline of a proposed rule, the baseline specification may be complicated by regulations other than the one being promulgated nearing completion. It is important to consider how these other regulations affect market conditions and the degree to which they might influence the costs or benefits associated with the policy of interest. This is true not only for multiple rules promulgated by the EPA, but 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, such as the

15 Commonly used indices include the Bureau of Labor Statistics' Consumer and Produce Price Indices (CPI and PPI), the Bureau of Economic Analysis' Gross Domestic Product (GDP) deflator, and engineer cost indices. The most appropriate index will depend on the application.

Occupational Safety and Health Administration (OSHA), Department of Transportation (DOT) and Department of Energy (DOE), develop rules that may affect some of the same entities as EPA regulations.

5.3.1 Linked Rules

When rules affect the same industry or when multiple rules are needed to achieve a policy objective, it may be possible to analyze these rules together, provided that they can be promulgated at the same time. For example, the EPA may issue a rule covering both the effluent limitation guidelines (ELGs) for an industry, providing technical requirements, and requirements under the National Pollution Discharge Elimination System (NPDES), providing details of the permitting system (e.g., U.S. EPA 2002). Since the ELGs and NPDES work together to achieve one objective, it makes sense to analyze them together. In some cases, linked rules may affect the same industry but have different enabling statutes. For example, in 1997, EPA issued a single rule for the pulp and paper industry covering the National Emission Standards for Hazardous Air Pollutants under the Clean Air Act and the Effluent Limitations Guidelines for Pretreatment and New Source Performance Standards under the Clean Water Act (U.S. EPA 1997).

The best approach for linked rules that are promulgated at the same time is to include them all in the same analysis. Analyzing multiple rules as if they were one rule simplifies the baseline specification by comparing them to the world in which none of the linked rules are in place. However, it is important to make sure that evaluating them together does not conceal significant differences in the net benefits of the individual requirements. For example, a linked rule might establish emission limits for several different pollutants each with distinct control technologies and separate benefits. In this case, the analyst should follow the guidance on policy options presented in Chapter 3. When a rule includes several distinct regulatory provisions, the benefits and costs of each provision should be analyzed both separately and jointly, estimating the net benefits of a regulatory option with and without that specific provision.

When statutory requirements and judicial deadlines complicate promulgating multiple rules as one, coordination between distinct 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 in a way that avoids omissions or double counting.

In some cases, there is a link between rules that are not being promulgated at the same time. A new rule may affect the associated compliance behavior of an existing rule. For example, regulations that establish Maximum Contaminant Levels (MCLs) for drinking water may affect Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) cleanup standards, as the MCLs are the in-situ cleanup standards for surface and groundwater water (42 U.S.C. 9621). In this case, the general guidance that all benefits and costs should be assessed in BCAs of regulatory actions should be followed.

5.3.2 Unlinked Rules

In some cases, it is not feasible to analyze a collection of rules being developed at the same time in a single analysis. 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, but the order in which the rules are being analyzed should be stated explicitly. If two rules are issued in sequence but some of the costs of complying with the second rule are incurred in the process of complying with the first rule, then these costs should be included in the baseline and should not be considered as costs of the second rule. Only the incremental benefits and costs should be included in the second rule. For example, in 2005, the baseline of the Clean Air Mercury Rule (CAMR) included mercury emission reductions from the previously issued Clean Air Interstate Rule (CAIR) (see Text Box 5.2).

The assumption commonly made when rules cannot be evaluated together is to consider the actual or statutory timing of the rules and use this to establish the sequence in which they are analyzed. 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. For this 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 an orderly sequence of events that allows the attribution of changes in behavior to a unique regulation, there may be no clear way to allocate the costs and benefits of a package of policies being developed at the same time to each individual regulation. By implication, there is no theoretically correct order for conducting a sequential analysis of multiple policies that are promulgated simultaneously. In this case, analysts should make a reasonable assumption and explain it, detailing which rules are included in the baseline (see Text Box 5.2). If the impact of other rules on the costs and benefits of the rule under consideration is small, then this may be all that is necessary; it may not be worth additional time and resources to reconcile the baseline of rules being developed at the same time. On the other hand, when the impact on the costs and benefits is large or if there are few overlapping rules, then a sensitivity analysis can be included to test the implications of including or omitting other regulations.

In this sensitivity analysis, it may be possible to use the overlapping nature of the regulations to allow for some regulatory flexibility in compliance dates and regulatory requirements. Furthermore, if the benefits and costs of each rule in the sequence are expected to differ significantly based on the order in which they are evaluated, a sensitivity analysis that changes the order of their evaluation may provide insights into how to design each to maximize the net benefits of the rules collectively.

5.3.3 Accounting for Benefits and Costs that Accrue Across Multiple Rules

When the EPA targets the same contaminants or industries through a sequence of regulations, the benefits and costs of these actions are additive. To ensure consistency in regulatory accounting, regulatory analyses should fulfill an “adding-up condition" when comparing a single large regulation to multiple smaller regulations that imply the same requirements for the same set of entities. The adding-up condition means that the sum of the estimated incremental benefits (and costs) from a set of small regulations analyzed separately should be the same as the incremental benefits (and costs) from the same actions evaluated jointly in a single regulation. Benefits and costs from previous rules should be included in the baseline so that they are not double counted in a new regulation.

The adding-up condition was originally proposed in the context of contingent valuation studies (Diamond and Hausman 1994; Kling and Phaneuf 2018) and has been applied to valuation of water quality improvements (Newbold et al. 2018). If analysts do not impose an adding-up condition and fail to account for improved environmental quality in the baseline when valuing incremental improvements from successive regulations, then inconsistent results could arise if people value marginal improvements more when the environmental good is scarce.

Text Box 5.2 - Accounting for Other Regulations in the Baseline

Because the benefit and cost estimates of one regulation may be affected by those of others, it is important to consider if they should be incorporated into the baseline. As a rule, analysts should be transparent and use objective reasoning when deciding to account for other regulations in a baseline. Transparency requires that all assumptions are clearly stated. Objective reasoning requires that speculation be avoided. If there is uncertainty about an anticipated rule, then two baselines — one with the anticipated rule and one without — might be considered. If only one baseline is considered due to time or resource constraints, then it should be constructed using only final rules and, in some cases, imminent rules that are expected with a high degree of certainty in the absence of EPA action. General guidelines to follow are given below.

All final rules, including those that have not fully taken effect, should be included: The analysis should assume firms will comply with already promulgated rules. For example, on March 15, 2005, the EPA promulgated the Clean Air Mercury Rule (CAMR) to reduce mercury emissions from coal-fired power plants (U.S. EPA 2005b). Five days earlier, on March 10, 2005, the EPA finalized the Clean Air Interstate Rule (CAIR) (U.S. EPA 2005a). While the primary purpose of CAIR was to reduce sulfur dioxide (SO_2) and nitrogen oxides (NO_x), the control technologies necessary to achieve these reductions also lowered mercury emissions. Because the final CAIR rule had been issued, the analysis for CAMR assumed that the mercury reduction from CAIR was in the baseline. This meant that the estimated incremental reduction in mercury from CAMR was much smaller than if CAIR had not been included in the baseline.

Including imminent final rules may be appropriate if the impacts are known with a high degree of certainty: If another (final) rule is imminent and will take effect prior to the effective date of the new rule under consideration, then the imminent rule should be included in the baseline, but only if its requirements and impacts are known with a high degree of certainty. The analyst should not speculate that another rule will be implemented. In addition, the analyst should be clear as to what assumptions have been made to include the imminent rule in the baseline.

Proposed rules should not be in the primary baseline: While a proposed rule signals the intent to issue a final rule and the Agency maintains a schedule to do so, there is no guarantee that the final rule will be issued or that it will follow the planned schedule. Even if the Agency does issue a final rule, it may differ significantly from the proposed rule, which means that the assumptions embedded in a baseline using a proposed rule will not accurately reflect the likely future effects of the final rule. An alternative baseline for a proposed rule may have another proposed rule in it, however, if the two rules are expected to be finalized in the same sequence and the existence of the first rule may influence the benefits and costs of the second substantially.

Future regulatory actions of other jurisdictions should be considered carefully: Actions by state and local governments and even international organizations can affect the costs and benefits of federal rules, particularly if they are regulating the same sector or pollutant. In this case, the analyst must use professional judgment to determine what would happen in the baseline (i.e., in the absence of EPA action) and how the regulatory response of other jurisdictions may change in the policy scenario.

State regulations that have been finalized should be included in the baseline. The more difficult case occurs when a state has a legal obligation to implement a regulation but either has not done so or is in the process of doing so. For example, the EPA occasionally issues rules establishing numeric water quality standards for some states when the states themselves have not done so. One might argue that the state regulation should be in the baseline since they had the legal obligation to issue the criteria, but this is not the case. The EPA's justification for action is that it assumes the state will not act. In this example, only if the state would issue the water quality standard in the absence of EPA action can a reasonable case be made for including the state action in the baseline.

Compliance with a finalized international agreement cannot simply be assumed in the baseline, especially if some EPA action (such as codifying the international standard) is required for it to become effective. The costs and benefits associated with any behavioral response by firms to the EPA action should be part of the policy scenario. In the case where firms will meet the international standard on their own, even without EPA action, then the compliance with the standard can be included in the baseline but establishing that this behavioral response will occur requires justification.

In some cases, environmental regulations yield relatively small changes in health or the environment that may not be noticeable to the public until multiple regulations have achieved a large aggregate improvement. Just as it is important to account for small average costs imposed by regulations — which can be economically significant when aggregated over a sufficiently large population — it is conceptually correct to account for small improvements in public health and the environment. For instance, the EPA's Science Advisory Board (U.S. EPA 1998) noted that, "small effects distributed across a large population exert large total health effects," and recommended that the Agency quantify changes in IQ resulting from regulations that reduce lead exposure, including changes of less than a single IQ point on average per child.

Some benefits only occur after a threshold has been reached. However, a specific benefits threshold may not be met with a single rule. In such cases, it is reasonable to account for the benefits of making progress toward a goal, even if the threshold is not met in the rule under consideration. Otherwise, if the benefits are associated only with the rule that passes the threshold, it may be impossible to justify the previous rules that made incremental progress.

For example, the EPA has calculated the benefits associated with improving river miles for various designated uses (e.g., swimming, fishing, and boating) in several 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. Analyses of earlier rules claimed benefits only if a river mile changed its designation, implicitly giving a value of zero to partially improved river miles. More recent analyses have included estimates of the partial benefits from incremental improvements toward the threshold. Either approach can be used, but accounting for the benefits of partial gains provides useful information to decision makers and the public and allows the Agency to justify incremental progress to a threshold. Once partial gains have been valued in one rule, then subsequent rules cannot claim full credit for crossing the threshold. Doing so would double count those benefits.

In the special case when new data or methods make estimates of benefits or costs for earlier rules obsolete, the analyst should develop a baseline based on the new information and discuss all changes made since the previous regulatory analysis.

5.4. Time Horizon of Analysis

The time horizon of analysis is the period over which the baseline and policy scenarios are compared. The time horizon is defined by the starting and ending points.[16] A guiding principle is that the time horizon should be chosen to capture all the benefits and costs for the policy

16 The time horizon for analysis may also be called the "time period of analysis" or "time frame of analysis."

alternatives analyzed, subject to available resources.[17] This principle is consistent with the requirement that a BCA sufficiently reflects the welfare outcomes of those affected by the policy. If the time horizon is too short, the estimate of the net benefits will be of incorrect magnitude and perhaps of the wrong sign because benefits and costs often occur over different periods of time. The analysis should clearly describe the time horizon used for the analysis and it should be clearly identified whenever present or annualized values are reported (see Chapter 6).

The appropriate time horizon will depend on the economic and legal conditions unique to the regulatory context under consideration. In many cases, the time span of the physical effects that drive the benefit estimates, duration of market effects from compliance activities, the duration of impacts on other externalities, and the economic lifetime of any pollution control investments will be key factors in its determination. Legal conditions that affect the time horizon of analysis include the timing of compliance dates. While selecting the appropriate time horizon is challenging, the analysis should identify the time horizon chosen and explain why it is expected to capture all benefits and costs. It should also identify the extent to which the sign of net benefits or the ranking of policy options by their magnitude of net benefits may be sensitive to the choice of the analytic time horizon.[18]

The starting point for the analysis should be based on when conditions between the baseline and policy scenarios diverge, and thus benefits and costs of the regulation begin to be realized. Two possible choices for the starting point are when an enforceable regulatory requirement becomes effective or when the final rule is promulgated. These dates are convenient starting points because they are clearly defined under administrative procedures and represent specific deadlines. However, the starting point of the analysis should precede the date when regulatory requirements become effective if firms or households are expected to make anticipatory investments or other behavioral changes after the rule is finalized and leading up to the effective date.[19] Likewise, for a regulation with requirements that become effective over time, benefits and costs should be accounted for during the period prior to when the legal requirements are fully implemented. A time horizon of the analysis that begins when a regulation is fully implemented is insufficient for accounting for all benefits and costs in the case where behavior changes prior to compliance dates, and thus the starting point of the analysis should be earlier.

The duration of when costs and benefits occur should generally be used to determine the ending point for the analysis. In theory, the longer the time horizon, the more likely the analysis will capture enough of the benefits and costs of the regulation to reliably estimate net-benefits and compare alternatives. However, other factors, such as the relative uncertainty in projecting

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17 Chapter 6 provides a formal method of identifying the ending point of the time horizon of analysis. A symmetric method may be used to identify the starting point. In addition, Chapters 7 and 8 also provide detailed guidance on selecting the time horizon of the analysis for benefits and costs, respectively.

18 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 time horizon of analysis. Chapter 6 provides guidance on how to discount benefits and costs.

19 In most circumstances, a starting point that precedes final rule promulgation is unnecessary, but an earlier starting point may be desirable if significant behavioral changes were made in anticipation of the final rule. Two possible starting points that precede promulgation of the final rule and are clearly defined legal milestones are when authorizing legislation was signed into law and when the EPA formally proposed the rule. However, when using a starting point that precedes regulatory requirements, it is important for the analysis to identify which behaviors occurred specifically because of the anticipated federal rule versus those that happened for other reasons. This will likely be difficult to do.

conditions in the distant future, may also need to be considered. Forecasts of economic, demographic, and technological trends are required over the entire time horizon of the analysis. Because long term forecasts are less reliable than near term forecasts, the analyst should balance the advantages of capturing important effects against the disadvantages of decreased reliability of forecasts further out in time, although those sources of uncertainty may meaningfully affect benefits or costs and should be accounted for if so. The period in which a regulation is fully implemented should not be used as the ending point if benefits and costs will occur thereafter. Furthermore, regulated entities will consider expected future conditions when choosing their compliance strategies, and a longer time horizon will capture the information they will use when choosing their compliance approaches.

Analysts should ensure consistent accounting of benefits and costs considering differences in when they accrue over time. To ensure consistent accounting, all the costs from activities that lead to quantified benefits should be accounted for in the analysis and vice versa. Ensuring consistency implies that the ending point may differ for assessing costs than for assessing benefits when the accrual of costs and benefits does not coincide.[20] For example, the human health benefits of a policy to reduce leachate from landfills may not occur for many years after the cost of compliance is incurred either because decreases in groundwater contamination take time or because even after contamination is reduced some health improvements do not manifest immediately. In other contexts, while control costs are incurred upfront, changes in pollution may lead to health and ecological benefits that continue to accrue over time.

Generally, the analysis should account for costs until at least the end of the economic lifetime of any pollution control methods adopted for regulatory compliance.[21] Costs will then be consistent with the total abatement, and in turn benefits, achieved by these pollution control methods.[22] Similarly, the length of the cost analysis should capture any turnover in markets for regulated goods (e.g., vehicles) and the length of time those goods are in use. This guidance may be challenging to implement in an analytic framework that captures the possibility of additional regulated sources appearing in the future, but the possibility of entry and exit of sources should still be included. Again, the analysts should weigh the value of additional benefit and cost information gleaned from a longer time horizon of analysis against uncertainty about future economic conditions.

Some statutory provisions have schedules for when regulations need to be reviewed, and an ending point corresponding to this review date may be a tempting choice. However, care should be taken when using regulatory or statutory deadlines to determine the ending point of the time horizon of analysis. For example, these provisions may not envision the regulation being loosened but only tightened, and therefore the requirements under consideration are expected to persist over time, at

20 However, as explained in Chapter 6 annualized benefits and costs should be calculated using the same assumed time period over which the annualized values apply.

21 The economic lifetime is the length of time a piece of equipment is expected to be operational before it is worn out and needs to be replaced. This guidance is particularly important when compliance costs are amortized over an economic lifetime or financing period. When compliance costs are amortized the benefits during one segment of the amortization period may be notably different than over another segment of the amortization period. The analysis will be misleading if the choice of segment affects the relative benefit to cost estimate (as well as the total benefits and costs of the regulation).

22 As discussed in the previous paragraph, if the benefits from these controls do not arise until later (i.e., are latent), the end date for the benefits analysis should be later than the end date for the cost analysis.

least at the promulgated level of stringency, potentially yielding additional benefits and costs.[23] Similarly, the benefits and costs of a regulation should be evaluated beyond when a particular statutory requirement is satisfied if the regulation will continue to affect behavior. A time horizon that reflects the span over which the baseline diverges from the policy case and accounts for all the benefits and costs is appropriate even if the period extends beyond the scheduled review.

In certain circumstances where benefits and costs are not expected to notably change over time, it may be analytically convenient to estimate benefits and costs over a shorter time period (e.g., one year) if they are representative of the benefits and costs over a longer time horizon of analysis (e.g., a decade). In other cases, it may be analytically challenging to estimate benefits and costs for each period over the entire time horizon; thus, benefits and/or costs are estimated for only a few periods that are each representative of longer periods.[24] In these cases, the analysis should still identify the entire time horizon over which the representative periods of analysis are applicable and discuss any limitations or uncertainty introduced by this approach. The representative periods of analysis should be chosen such that they adequately identify the relative net benefits of the various options under consideration. Focusing on one or a subset of periods without careful consideration of whether those periods are representative of all benefits and costs over longer time periods may lead to potentially misleading findings of the magnitude, and possibly even the sign, of net benefits. For example, treating the annual benefits and costs in the year a rule becomes fully implemented as representative of the benefits and costs in all years may lead to a misleading net-benefits estimate if the annual benefits or costs incurred prior to the full implementation year are quite different.[25,26]

5.5 Representing Economic Behavior

To measure the benefits and costs of a regulation, it is important to characterize the behavior of firms and households in both the baseline and the policy scenarios. In particular, assumptions about how firms and households (1) engage in technological change, (2) comply with regulations,

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23 Furthermore, if there is a credible reason to assume that the regulation will be loosened in the future then this possibility should be acknowledged in the analysis and the compliance choices of regulated sources should reflect this possibility (e.g., regulated sources would be more likely to adopt easily reversible compliance strategies if they thought the regulation may be loosened in the future). Another reason to evaluate the benefits and costs of the rule beyond the statutory review date is if the rule currently under consideration is expected to be accounted for in the baseline of any analysis with a time frame beyond the statutory review date, including the rulemaking following the statutory review.

24 The representative periods may be chosen to characterize periods of different length. For example, if benefits and costs increase quickly in the near term and are then generally constant afterward, representative periods used to characterize the near term are applicable to short period (e.g., a couple of years), while representative periods used to characterize the long term are applicable to longer periods (e.g., a decade).

25 This outcome is possible even if the benefits and costs in the full implementation year are representative of later years. If they are not representative of benefits and costs incurred in later years then, again, the net benefits estimate may be misleading.

26 Comparing an annualized value to an annual value also may be potentially misleading. The annualization period chosen is arbitrary so long as it is long enough to accounts for all benefits and costs, and a longer annualization period would lead to lower annualized benefits or costs. For example, comparing an annual benefit to annualized costs over a long time period may give the impression that net benefits are positive when they may not be. Also, if annualized values are reported, they should be reported for both benefits and costs. See Section 6.1.6 for further discussion.

(3) participate in voluntary actions, and (4) affect levels of other contaminants in the baseline and policy scenarios can also influence costs and benefits.

5.5.1 Behavior of Households and Firms

Predicting firm, household, and other organizational responses to regulation requires a model of economic behavior. Analysts should assume behavior consistent with utility or profit maximization unless there is evidence supporting other behavioral assumptions (see Section 4.4 and Section 5.5.2 for more discussion of behavioral anomalies).

When modeling the response to regulation, it is important to capture how regulated firms may choose to comply with new requirements. For instance, firms could change production practices, output, location, or even exit the industry. Likewise, it is important to capture household responses, such as changes in the products they buy, where they live, or the types and frequency of averting behaviors (e.g., purchasing bottled water or staying inside on bad air quality days). These responses also may result in changes in market prices and externalities, which could further alter economic behavior. Behavioral response to the regulation may also precede compliance dates, which can make it difficult to disentangle how much of the behavior is attributable to the regulation.

Future economic conditions are inherently uncertain, and households, firms, and other organizations will account for these uncertainties when responding to regulations (as well as in the baseline). Their decision-making under uncertainty may differ from what would occur if future conditions were known with certainty. For example, when facing uncertain future economic conditions, regulated entities may avoid making irreversible investments, which provides them greater flexibility to adjust their future compliance strategies. This may occur even if under most likely future economic conditions an irreversible investment is the least-cost compliance strategy. Without accounting for such uncertainty, an analysis may predict greater investment in an irreversible compliance method than would be expected to occur.[27]

Capturing behavior when uncertainties are present is analytically challenging. For example, information is needed on the risk preferences of households and firms. Economic modeling tools are considerably more complex (or must sacrifice other details) to model decision-making that accounts for uncertain future conditions. When uncertain future conditions are likely to have a significant effect on the behavior of households and firms, the analysis should describe these sources of uncertainty and how they may affect estimates of benefits and costs.

Depending on the types of behavioral responses that are anticipated, the analyst will need to identify and select the most appropriate economic and environmental model(s) for the regulatory analysis. Uncertainty in model results tends to be higher when a model is either exceedingly simple (e.g., because it misses key interactions or feedbacks) or increasingly complex (e.g., due to data

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27 This behavior is an example of option value (Dixit and Pindyck 1994). An option value is the value of delaying an action to learn if it is the best choice. Regulations may impose benefits and costs by eliminating options in the future that may have value to society or private firms and households even if those options would not be exercised under likely future conditions.

requirements). Analysts should seek balance: "the optimal choice generally is a model that is no more complicated than necessary to inform the regulatory decision" (U.S. EPA 2009).[28]

Models used to inform EPA decision-making should be reliable, transparent, defensible, and useful (U.S. EPA 2009). For instance, any modeled changes in behavior should be supported by empirical estimates of demand, supply, cross-price, and income elasticities.[29] When the literature presents a range or identifies factors that could significantly affect these estimates, analysts should also examine the sensitivity of benefit and cost estimates to different elasticity assumptions. See Text Box 5.3 for a discussion of other considerations when selecting models for estimation of costs and/or benefits, including the extent to which a model adequately represents key markets of interest and the representativeness of other significant assumptions. See Section 5.6 for guidance on how to conduct uncertainty analyses for BCA and other economic analyses.

5.5.2 Potential for Cost Savings

If firms and households behave in ways consistent with profit and utility maximization, they will adopt available cost-effective technologies or practices absent regulation. Even if they are not in widespread use when a new regulation is developed, cost-effective technologies may be adopted under baseline conditions in the future as information about their effectiveness spreads. When households and firms voluntarily undertake these changes without the regulation, the regulatory action cannot be credited with any private cost savings resulting from their adoption. In cases where a regulation is estimated to result in net private cost savings, it is important to provide evidence of why these cost-saving measures would not already be undertaken in the baseline.

When evidence to explain this phenomenon is not available, analysts should consider whether the finding of private cost savings is defensible and whether all costs are being accounted for. For instance, a regulation may impose "hidden" costs that are not easily quantified in a standard engineering cost model but still represent welfare losses for firms or households that offset cost savings from adopting a technology. Lower operating expenditures from a new technology required by a regulation might be offset by increases in other costs if the new technology breaks down more frequently, requires special training to operate, or has other undesirable features. If data are available on such costs, analysts should include them in the analysis.

In some cases, evidence may suggest that firms or households do not adopt cost-saving measures because of market failures (e.g., asymmetric information). If the regulation addresses these market failures, it could lead to net private cost savings. In these instances, analysts should provide a clear description and evidence of the market failure and how the new action addresses it.

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28 "Models are constructed to provide the simplest analysis possible that allows us to understand the issue at hand […] The real world is typically much more complex than the models we postulate. That doesn't invalidate the model, but rather by stripping away extraneous details, the model is a lens for focusing our attention on specific aspects of the real world that we wish to understand" (McAfee and Lewis 2009).

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

Text Box 5.3 - Model Choice

When selecting models for use in regulatory analysis, analysts should evaluate the following:

Is the model based on sound science? Prior to use in regulatory analysis, the model should be subject to credible and objective peer review to ensure that it is consistent with scientific and economic theory and based on the best available data and empirical evidence. Many of the questions that follow can also be put to peer reviewers to evaluate the particulars of a specific model and/or appropriateness of the model within a specific policy context.

Is the model "fit for purpose"? Analysts must identify the best model(s) for the analysis and thoroughly explain why it is applicable given the features and expected effects of the rule. A model may be based on sound science but still inappropriate for evaluating the features of interest.

Is the model supported by the best available data? The suitability or representativeness of underlying data to evaluate the effects of a specific policy is also an important consideration. For instance, data quality and resolution may limit the ability to use some models in a regulatory context. For this reason, it is important to identify what data are available or can be collected to adequately parameterize the model (U.S. EPA 2009). Analysts should use assumptions and calibration/estimation of key parameters that are peer reviewed.

Does the model reasonably approximate the systems or market(s) of interest? A model should capture the most salient details of the policy and the systems or markets affected. A model selected to evaluate a regulation should be no more complicated than is necessary to inform decision-making. If model capabilities add complexity without substantially improving performance, the more transparent option is to eliminate them (NRC 2007).

Is the model transparent? In addition to model tractability, it is important that documentation of all aspects of the model be publicly available, including details about model structure, key assumptions, sources and values of key parameters, and limitations. When possible, models and their underlying data should be publicly available. When a model is not publicly available, for instance due to the confidential nature of underlying data, it is important to explain the reasons for relying on these sources of information.

Can key model assumptions or parameter values be evaluated? Analysts should use sensitivity analysis to explore the robustness of results to key input values, specifications, or assumptions, particularly when the literature is inconclusive regarding the most defensible approach or estimates. Sensitivity analysis may be application specific: parameters that may matter little in one context may be key drivers of results in other contexts (U.S. EPA 2009).

Conducting uncertainty analysis is also important, as it "investigates the effects of lack of knowledge and other potential sources of error in the model" (U.S. EPA 2009). Sensitivity and uncertainty analysis inform users about the confidence that can be placed in model results. In some cases, analysts also may need to rely on multiple models. Section 5.5 provides detailed guidance regarding when sensitivity and uncertainty analyses are appropriate.

What are the key limitations of the model? Every model has its strengths and weaknesses. It is important that decision makers and stakeholders understand a model's limitations. What does the model capture? What is not captured or only captured with large bounds of uncertainty? These should be communicated within the analysis in a way that is easy for a non-technical audience to understand.

The economics literature has also documented specific instances in which households or firms act in ways that appear to run counter to their self-interest (Shogren and Taylor 2008; Shogren et al. 2010; Croson and Treich 2014). However, research also indicates that market experience can eliminate behavior that is inconsistent with profit-maximization in certain settings (List 2003; List 2011). If estimated net private cost savings could be due to widespread suboptimal behavior, analysts should provide empirical evidence specific to the affected market. In addition, care should be taken to ensure that assumptions that underlie modeled household behavior are consistent with actual behavior.[30] In the absence of such evidence, analysts should assume rational profit- or utility-maximizing behavior by firms and households in the primary analysis, which would eliminate the possibility of estimating net private cost savings as a result of regulation.[31] Sensitivity analysis can be used to consider other behavioral assumptions if warranted.

It is also important for analysts to make consistent assumptions about firm and consumer behavior under the baseline and policy scenarios unless there is reason to believe the regulation will change underlying behavioral patterns. For example, the economics literature has found mixed evidence on whether car buyers fully account for future gasoline expenses when choosing fuel economy.[32] A fuel economy standard could reduce the impact of undervaluation of fuel economy on consumer decisions, but if such behavior occurs in the baseline, it is likely to persist regardless of regulatory requirements. Chapter 4 Section 4.4 offers more discussion about possible insights from behavioral economics for policy design.

5.5.3 Technological Change

It is important to capture future changes in production techniques or pollution control that may influence the baseline and policy scenarios and consequentially both costs and benefits. Technological change can be thought of as having at least two components: genuinely new technological innovation, such as the development and adoption of a new alternative pollution control method; and learning effects, in which experience leads to cost savings through improvements in operations, capability or similar factors. Analysts should recognize that the longer the time horizon, the greater the uncertainty regarding the potential for and characteristics of technological change (or learning) within a sector. Thus, it is important to balance the need to account for the effect of innovation on the costs and benefits of regulation against the defensibility of those analytic assumptions.

Technological change in other sectors of the economy may also be important to account for in the analysis. For example, while the cost of phasing out ozone-depleting substances has declined over time due to technological improvements in substitutes, innovation in mitigating factors, such as improvements in skin cancer treatments and efficacy of sunscreen lotions, have also occurred. Further, the analysis should include the costs associated with research and development (R&D), including the potential to crowd out other investments that would have occurred absent the

30 See Ketcham et al. (2016) for an example where the finding that consumers do not act in their own self-interest was actually driven by the inflexibility of the functional form assumed.

31 An exception would be when the regulation involves a transfer, such as a subsidy or rebate to purchase a product, that leads to a net-cost savings for the firm or household. However, absent the value of the transfer, the net-cost savings would still be negative under profit- or utility-maximizing behavior.

32 Recent studies continue to find a wide range in how consumers value future gasoline expenses in their vehicle purchase decisions (Allcott and Wozny 2014; Busse et al. 2013; Sallee et al. 2016; Gillingham et al. 2021; Leard et al. 2023).

regulation, to correctly value cost-reducing technological innovation, but only if the costs are induced by the regulation. Distinguishing R&D induced by the regulation from changes in other investment decisions is often difficult. While innovation is expected to occur in the baseline and policy scenarios, rates of technological change may differ across scenarios due to innovations that reduce the cost of compliance. 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 are, at a minimum, in development or reasonably anticipated.

In some cases, there also may be empirical evidence of reductions in costs as firms accumulate experience in production or abatement over time. Historic and projected estimates of learning are often represented by "learning rates". A learning rate is typically defined as the percentage reduction in costs for each doubling of production (or production capacity). It is not advisable to assume a constant, generic learning rate or 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 and cost reduction. Furthermore, while learning may reduce compliance costs over time, it is not widely believed that cost become negative as discussed in Text Box 5.4. Before incorporating learning effects, the analyst should carefully examine the existing evidence for relevance to the specific context. Estimated learning effects can vary due to many factors, including already accumulated experience with a technology, industry, and the length of the time period considered. Also, because estimates of learning rates are based on doubling of cumulative production, including learning effects will have a greater influence on analyses with longer time horizons. See Chapter 8 for further discussion.

5.5.4 Compliance

One aspect of analytic design that can be complex is what to assume about the extent of compliance with current and future environmental regulations. Assumptions about compliance in both the baseline and policy cases can significantly affect the results of the analysis and should be clearly described. Assumptions about compliance rates for a new regulation for a sector should generally be based on past compliance behavior for related regulations for the sector. When an industry has not been regulated before, data will not typically be available to gauge the likelihood of compliance with a new rule, but compliance should be expected to be consistent with similar regulations of similar entities. In most cases, a baseline and policy scenario that assumes full compliance should be analyzed along with evidence-backed scenarios including alternative assumptions about compliance.

Text Box 5.4 - Technological Change, Induced Innovation, and the Porter Hypothesis

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

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

The weak version of the hypothesis assumes that an environmental regulation will stimulate innovation, but it does not predict the magnitude of these innovations or the resulting cost savings. There is mixed evidence in support of the weak version of the Porter hypothesis (Ambec et al. 2013; Martinez-Zarazosa, et al. 2019). This version of the hypothesis is very similar to the induced innovation hypothesis.

The narrow version of the hypothesis predicts that flexible regulation (e.g., incentive-based) will induce more innovation than inflexible regulation and vice versa. There is empirical evidence that this is the case (Kerr and Newell 2003; Popp 2003; De Santis and Jona Lasinio 2016). Analysts may be able to estimate the rate of change of innovation under the weak or narrow version of the hypothesis, or under induced innovation. Note, however, that these types of innovation may crowd out other forms of innovation. By raising the cost of pollution, the regulation makes it profitable to find cheaper compliance strategies, but finding these strategies also has its own opportunity cost (e.g., firms use their engineers, scientists, and other experts to develop more cost-effective compliance strategies instead of developing some other technology).

The strong version of the Porter Hypothesis predicts cost savings from environmental regulation under the assumption that firms do not maximize cost savings without pressure to do so. While anecdotal evidence of this phenomenon may exist, the available economic literature has found no statistical evidence supporting it as a general claim (Jaffe et al. 1995; Palmer et al. 1995; Jaffe and Palmer 1997; Brännlund and Lundgren 2009; Ambec et al. 2013; Dechezlepetre and Sato 2017). For the strong version to be true, it requires special assumptions and an environmental regulation combined with other market imperfections that are difficult to generalize. Thus, analysts should not assume cost savings from a regulation based on the strong version of the Porter Hypothesis.

When there are significant compliance issues with an existing regulation, an assumption of under-compliance in the baseline for a new regulation should be included when supported by evidence from monitors, inspections, or enforcement actions.[33] Analysts may establish a "current practice"

33 For example, in the Lead Renovation, Repair, and Painting Program Rule (U.S. EPA 2008), the EPA assumed a 75% percent compliance rate for estimating costs and benefits based on compliance in the construction industry with previous occupational health and safety regulations.

baseline incorporating data on actual compliance rates rather than assume full compliance. Current practice baselines are particularly useful for regulations intended to address compliance problems with existing policies. Assuming a full-compliance baseline that disregards under-compliant behavior could obscure the value of these types of regulations.[34] If the policy being evaluated is not designed to address the underlying reason for non-compliance, then under-compliance data may be applicable to the policy case as well as the baseline.

If under-compliance is assumed either in the baseline or in the policy case, then identifying the reason for non-compliance is important and could affect the sign of the regulation's net benefits and the distribution of benefits and costs. For example, non-compliance could occur selectively where compliance costs are high. If compliance is not systematically correlated with costs, then the compliance assumption is less likely to change the sign of the regulation's net benefits.

When analyzing new requirements for an industry subject to existing regulations, it is important to carefully specify the assumptions about baseline compliance to avoid double counting benefits and costs. This could arise if the same set of actions occurs across multiple regulations. Assuming full compliance with existing regulations in the baseline makes it easier for analysts to focus on the incremental effects of the new regulatory action without double counting. If there is evidence of under-compliance in the baseline, analysts should consider whether the regulation is structured to reduce the compliance problem[35] or whether the problem is likely to persist in the policy case. If it will persist and this behavior is not captured, the net benefits of a regulation will not be estimated correctly. For example, if analysts repeatedly factor under-compliance into the baselines for a sequence of emissions tightening rules but assume that entities will fully comply under the policy case, inconsistent results will arise. Summing the benefits and costs from the sequence of rules will overstate the benefits and costs because each rule takes credit for a portion of the same actions.

Conversely, there may be cases in which firms over-comply with regulations. Over-compliance in the policy scenario should be assumed in limited circumstances. As with under-compliance, it is important to identify the reason for over-compliance and assure it is consistent with expected behavior. The analysis should not typically assume that a regulation will motivate abatement greater than what is legally required. However, over-compliance may occur if firms wish to reduce the risk of non-compliance (e.g., facilities may overcontrol due to local pressure) or because least-cost compliance methods achieve greater reductions than required (e.g., shifting to a different process that does not pollute rather than installing abatement equipment) among other reasons. In such cases, the benefits, and costs of over-compliance in the policy case should be accounted for. If more additional regulations are considered later, current practices can be used to define baseline conditions for the new regulation unless these practices are expected to change.

To summarize, analysts should include a baseline and policy scenario that assumes full compliance, but under-compliance in the baseline or policy scenario should also be analyzed when there is supporting evidence. Over-compliance can be assumed in limited circumstances. Whenever scenarios other than full compliance are included in regulatory analysis, the analyst should discuss the sensitivity of the results to the compliance rate assumption.

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34 For instance, banning lead from gasoline was precipitated, in part, by the noncompliance of consumers. When consumers put leaded gasoline in vehicles that required non-leaded fuel, this resulted in increased vehicle emissions (U.S. EPA 1985).

35 See Section 3.2 for a brief discussion of relevant enforcement methods to consider, Chapter 4 for some examples, and Section 8.2for a discussion of compliance assumptions in a cost analysis.

5.5.5 Voluntary Actions

Occasionally, polluting industries adopt voluntary measures to reduce emissions. Firms or sectors can undertake such actions independently, or they might participate in formal, government-sponsored programs. Such voluntary measures are adopted for a variety of reasons, including to improve public relations, to avoid regulatory controls, to reduce other legal risks or to access resources associated with joining a formal program. When this is the case, it is important to account for these actions in the baseline for new regulations and to be explicit about the assumptions of firms' future actions. If participation in voluntary programs was motivated by the threat of the regulation, then a new regulation could affect future participation in these programs.

Typically, voluntary emission reductions that are expected to occur without a new regulation may be included in the baseline consistent with the guidance on over-compliance above. This is not always possible, however, as voluntary actions are often difficult to measure (Brouhle et al. 2005). It can be difficult to determine whether pollution reduction measures that precede compliance dates represent anticipatory effects that are attributable to a regulation or if they are voluntary measures that would have occurred without the regulation. Sensitivity analysis could shed light on the importance of assumptions about voluntary emission reductions under the baseline if this is a significant source of uncertainty.

5.5.6 Changes in Other Environmental Contaminants

It is common for EPA regulations to cause decreases and increases in environmental contaminants that are not the subject of the regulation. These changes may occur for a variety of reasons that the analyst should consider. This section provides guidance specifically on identifying and accounting for changes in these other contaminants by drawing out the implications of properly accounting for the baseline and behavior discussed above. Projections of changes in the levels of environmental contaminants should be consistent with expected economic behavior. These changes should be based on expected outcomes of least cost compliance, existing economic relationships, and continued compliance with existing regulations. The analysis should take a balanced approach to identifying increases and decreases in other contaminants that may be affected by the regulation relative to efforts to account for other welfare changes that may result.[36] Any benefits or costs from these changes in other contaminants should be accounted for in a BCA.

As mentioned in Section 5.1.3, changes in environmental contaminants other than those subject to the regulation may result from the compliance approaches used by regulated entities.[37] For example, the use of an abatement technology to reduce one air pollutant may simultaneously

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36 The benefits from changes in environmental contaminants other than those related to the statutory objective of the regulation have sometimes been called "co-benefits" and these contaminants sometimes called "co-pollutants". However, these terms are imprecise and have been applied inconsistently in past practice, and as such should be avoided (unless these terms are used explicitly in statutes). Similarly, benefits from changes in environmental contaminants other than those related to the statutory objective of the regulation are sometimes referred to as "ancillary benefits,". This term should be used cautiously in an analysis because it may be interpreted as having economic, legal or policy meaning that is unintended.

37 Section 5.1.3 also emphasizes that changes in externalities other than those due to changes in environmental contaminants should be accounted for in a BCA. These other externality changes are not as common across regulations as changes in other contaminants but may be particularly important in certain regulatory contexts such as changes in transportation externalities from emissions standards on vehicles (e.g., congestion or safety) or changes in ambient conditions such as temperature and noise.

reduce or increase other air pollutants from the same source, and/or could change the emissions of the same or another pollutant into a different medium (e.g., water). It is also possible for changes in other environmental contaminants to occur as a result of market interactions. For example, a regulation may cause consumers or firms to substitute away from one commodity towards another, whose increased production may be associated with additional emissions of an environmental contaminant as well as the costs of abating it. Other examples include when a regulation induces beneficial reuse of a waste product and thereby reduces production and the associated emissions and costs of the product that the waste replaces; a controlled pollutant might be a precursor to multiple secondary pollutants; or when the use of a hazardous product is banned, and its replacement also poses a hazard.

Care should be taken when estimating changes in other contaminants to ensure they are consistent with expected market behavior and technological change. For example, consider an abatement technology that may potentially reduce emissions of multiple pollutants. The analyst should consider whether the technology will achieve similar reductions in all of these pollutants in a new application as it had in previous applications, or if the regulated entities will tailor it to control the regulated pollutant(s) in the new application so as to reduce the technology's cost.

As with estimating changes in contaminants subject to a regulation, analysts should also consider the implications of existing pollution control regulations on other contaminant levels and costs. For example, consider the case where a regulation on one pollutant leads to installations of a technology that reduces a second pollutant, and that second pollutant is subject to an allowance trading program with a cap that is economically binding (i.e., there is a positive allowance price). In this case, the regulation may not ultimately lead to reductions in the second pollutant. Instead, reductions in the second pollutant at regulated entities that install the new technology may be offset by reductions in abatement activities by entities subject only to the cap.[38] To the extent that any new regulation affects the cost of complying with an existing regulation, as would occur in this example, these changes in cost should be accounted for in the analysis.

If a regulation is expected to increase environmental contaminants not subject to the regulation, they should be accounted for in a BCA even if an anticipated future regulation is expected to mitigate them. This guidance follows directly from establishing the baseline and accounting for all benefits and costs. It is important to account for these changes for completeness, such that the sum of the benefits and costs of rules evaluated in sequence should sum to the costs and benefits of the rules if evaluated collectively as discussed in Section 5.3.3.

Finally, as discussed in Chapter 3, if the regulation is expected to induce large benefits from changes in contaminant(s) beyond those arising from its primary statutory objective, an analysis of a policy option where those contaminant(s) are regulated, either separately or simultaneously with the contaminants that are the primary statutory objective of the regulation, may be useful to determine whether there are more economically efficient ways of obtaining these benefits.

38 There may still be benefits (or negative benefits) from changes in the timing and location of emissions of these environmental contaminants even if the cap continues to bind. Chapter 4 describes how allowance trading programs work.

5.6 Uncertainty

Uncertainty is inherent in BCAs, particularly when estimating and valuing environmental benefits for which there are no existing markets.[39] The primary issue is often not how to reduce uncertainty, but how to account for it and present useful conclusions to inform policy decisions. While households and firms can be expected to incorporate uncertainty in decisions and responses will reflect their risk preferences (see Section 5.5.1), BCA itself should not adopt any particular risk stance because the Potential Pareto criterion requires BCA to reflect the values of those affected. An additional imposition of risk preferences in the BCA itself is, therefore, inconsistent with the underlying basis of BCA.

BCAs should present information on the expected or most plausible outcomes and associated uncertainty (Dudley et al. 2017). It is important to recognize that point estimates alone cannot provide policy makers with information about whether these estimates are robust to alternate assumptions, nor can they convey the full range of potential outcomes. Treatment of uncertainty is an essential component of analysis that enhances the communication process between analysts and policy makers.

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

- Present outcomes or conclusions based on expected or most plausible values;
- Provide descriptions of all known key assumptions, biases, and omissions;
- Perform sensitivity analysis on key assumptions;
- Include sensitivity analyses that examine both higher and lower values rather than only one or the other;
- Justify the assumptions used in the analyses; and
- Make full use of available probability distributions of key parameters.

Sensitivity analysis on key assumptions may be all that is needed for an uncertainty analysis, or it may be only the initial assessment. Statistical confidence intervals and probability distributions, if available, are used to describe the statistical uncertainty associated with specific variables and to provide a more complete characterization of uncertainty. The outcome of the initial assessment may be sufficient to understand the influence of key parameters on outcomes and to inform the policy decisions. If, however, the implications of uncertainty are not adequately captured in the initial assessment then a more sophisticated analysis should be undertaken when the data allow. The need for additional analysis should be clearly stated, along with a description of the methods used for assessing uncertainty.

Probabilistic methods such as Monte Carlo analysis can be particularly useful because they explicitly characterize analytical uncertainty and variability (e.g. Brandimarte, 2014). Where probability distributions of relevant input assumptions are available and can be feasibly and

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39 Stemming from definitions given in Knight (1921). economists have often distinguished risk and uncertainty according to how well one can characterize the probabilities associated with potential outcomes. Risk applies to situations or circumstances in which a probability distribution is known or assumed, while uncertainty applies to cases where knowledge of probabilities is absent. However, these definitions are not always adhered to in economics. Also, note that the economic definitions for these terms may differ from those used in other disciplines.

credibly combined, BCAs should characterize how the probability distributions of the relevant input assumptions would, on net, affect the resulting distribution of benefit and cost estimates. In this case, the analysis would consider sources of uncertainty jointly rather than singly.

However, probabilistic methods can be challenging to implement when data needed to characterize distributions are limited.[40] In the absence of data to specify distributions for specific parameter values, it is more transparent and defensible to use simpler sensitivity analysis. Note that for some rules OMB Circular A-4 requires a formal quantitative uncertainty analysis that provides some estimate of the probability distribution of regulatory effects.[41]

The analysis should make clear that the statistical uncertainty captured by the Monte Carlo or other probabilistic analysis generally does not account for model uncertainty, the degree to which mathematical models represent real-world systems. For example, when quantifying changes in a specific health effect from a reduction in an environmental contaminant, the statistical uncertainty analysis assumes that a particular dose-response model is the "true" model; that is, as if we are certain there is a causal relationship and that the dose-response function used in the analysis is the truth. There are some approaches to incorporating model uncertainty in probabilistic analyses, such as model averaging.[42] More often, model uncertainty (including uncertainty over whether an environmental contaminant causes a specific type of health impact) will need to be captured and described independent of the statistical uncertainty analysis. When possible, alternative model specifications that are supported by or consistent with underlying biological, engineering, or economic evidence or theory should be used to illustrate the consequences of assuming a different model.

It is important to recognize that there may be cases where there are competing assumptions, estimates or models considered as equally plausible that cannot be combined or weighted probabilistically. In these cases, it can be appropriate for the results driven by these factors to be presented co-equally in the BCA. However, the number of outcomes will generally grow multiplicatively with the number of inputs treated equivalently. For example, if there are three alternative inputs that are evaluated and treated separately and equivalently, and each of these can take two values, then there would be eight co-equal net benefits estimates to present, making it difficult to interpret BCA results. Therefore, the presentation of co-equal results in BCA should be done sparingly, with sensitivity analysis as the preferred treatment where possible. Presenting co-equal results should be reserved for particularly important analytic inputs and should always be fully described and justified whenever it is done.

5.6.1 Performing Sensitivity Analysis

Sensitivity analysis is a systematic method for describing how net benefit estimates or other outputs of the analysis change with assumptions about input parameters. Some basic principles for sensitivity analysis include:

- **Identify key parameters**. For most applied analyses, a full sensitivity analysis that includes every variable is not feasible. Instead, the sensitivity analysis will often need to be limited to

40 Jaffe and Stavins (2007) provide a useful overview of probabilistic analysis of uncertainty in regulatory analysis, including challenges and limitations.

41 See Circular A-4 (OMB 2023) for additional details on this requirement.

42 Moral-Benito (2015) provides an overview of model averaging in economics.

those input parameters considered to be key or particularly important, which may be economic parameters (e.g., valuation estimates) or inputs from other disciplines that feed into the benefits analysis (e.g., dose-response, exposure). A determination about which parameters are key should be informed by the range of possible values for input parameters and each one's functional relationship to the output of analysis. The analyst should specify a plausible range of values for each key variable and describe the rationale for the range of values tested.

- **Vary these key parameters**. The most common approach is a partial sensitivity analysis that estimates the change in net benefits or other economic outcomes while varying a single parameter, leaving other parameters at their base value. A more complete analysis will present the marginal changes in the economic outcome as the input parameter takes on progressively higher or lower values. When an input has known or reasonably determined maximum and minimum values, it can be informative to investigate if outcomes are robust to these alternative input values.

  Varying two parameters simultaneously can often provide a richer picture of the implications of base values and the robustness of the analysis but can be more difficult to communicate effectively. Analysts should consider using graphs to present these combined sensitivity analyses by plotting one parameter on the x-axis, the economic outcome on the y-axis, and treating the second parameter as a shift variable.[43] Results of the sensitivity analysis should be presented clearly and accompanied with descriptive text.

- **Identify switch points.** Switch points are defined as those conditions under which the economic analysis would recommend a different policy decision. For BCA, the switch point would typically be the input parameter value where estimated net benefits changes sign. Switch point values for key input parameters can be very informative. For instance, they can be compared to the available literature to assess whether the values are plausible or well outside known distributions or observations. While switch points are not tests of confidence in the statistical sense, they can help provide decision-makers with an understanding of how robust the analytic conclusions are.

- **Assess the need for more detailed analysis**. Finally, sensitivity analyses may be used as a screening tool to determine where more extensive effort may be needed. For example, the plausible range of values for an influential uncertain parameter may be narrowed with further research or data gathering, which can be used to better characterize the parameter's uncertainty. If several parameters independently have a large influence on the results of the analysis when they are varied, then a more sophisticated treatment of uncertainty that allows for joint consideration of their effects may be necessary. One option is to combine alternative values for multiple parameters into a scenario that differs from the primary analysis. It is important that the selected values be consistent with one another and that choices are explained and well-documented. It is also important to consider that combining extreme values for multiple inputs (e.g., minimum values) can produce a scenario that is unlikely so the analysis should include some description of the plausibility of the combination of values.

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

5.6.2 Approaches to Consider When Data Are Missing

When key data elements are unavailable in an analysis it will not be possible to estimate central values or perform sensitivity or quantitative analysis around those values. In these cases, it is important to assess and qualitatively characterize the importance of the missing information in the analysis. There are also analytic approaches to consider when data are missing.

- **Break-even analysis**. Break-even analysis can be used when one element is missing in an analysis. Essentially, break-even analysis identifies the switch point value for the missing element where the net benefits change sign.[44] Unlike the case above, however, the switch point value cannot be associated with any point an underlying distribution. Break-even analysis may best be explained by example. Suppose a BCA shows that net monetized benefits are negative, but there is a key health endpoint with an established per-unit estimate of economic value but without risk estimates that would allow quantification of the health endpoints. In this case it is possible to estimate the number of cases of the health endpoint avoided (each valued at the per-unit value estimate) at which overall net benefits become positive, or where the policy action will "break even."

  The same sort of analysis can be performed when analysts lack valuation estimates, producing a break-even value that should again be assessed for credibility and plausibility. Break-even analysis can also be used for missing costs when net quantified benefits are positive, shedding light on what the value of the missing cost estimate would need to be for net benefits to be negative.

  Break-even estimates can be assessed for plausibility either quantitatively or qualitatively. For example, the break-even unit value estimate for a specific health endpoint may be compared to values for effects considered to be more or less severe than the endpoint being evaluated. For the break-even value to be plausible, it should fall between the estimates of these more and less severe effects. Policy makers will need to determine if the break-even value is acceptable or reasonable.

  Break-even analysis is most useful when there is only one missing value in the analysis, or when it is applied to a large or important missing value. For example, an analysis missing risk estimates for two different health endpoints (but with valuation estimates for both), would need to consider a "break-even frontier" that allows the incidence of both effects to vary. While it is possible to construct such a frontier, it may be difficult to determine which points on the frontier are relevant for policy analysis.

- **Expert elicitation.** Expert elicitation is a formal process for obtaining and combining judgments from experts on missing inputs in the economic analysis.[45] The values elicited, and the uncertainty around these values if characterized in the elicitation, can then be used in the economic analysis for those missing inputs. Typically, expert elicitations include multiple experts to capture a range of backgrounds and diversity of knowledge, but ultimately the responses of these experts are combined into a single estimate or probability distribution for the input of interest. There are established approaches for the elicitation, including how to define the target questions, conduct expert interviews and analyze the

44 Boardman et al. (2018) describes determining break-even points under the general subject of sensitivity analysis and includes empirical examples.

45 OMB Circular A-4 (2023) suggests analyses consider drawing upon expert judgment using Delphi methods, a form of expert elicitation.

responses. Expert elicitations conducted for BCA should follow best practices and carefully document the elicitation process, results and how those results are applied in the BCA.

Formal expert elicitation can be time-consuming and require substantial resources so it should be reserved for those cases where its value, in terms of improving the BCA for decision-making, merits the resources needed. Less formal approaches for drawing upon expert judgments for missing values may also be useful if those approaches are clearly identified and described in the economic analysis, including any known limitations. See Colson and Cooke (2018) for an overview of expert elicitation.

5.6.3 Other Considerations Related to Uncertainty and Risk

There are additional issues related to uncertainty that may merit consideration, including how to account for responses to risk information, how to evaluate policies or regulations that provide information, and how to consider the value of information that may become available later.

- **Uncertainty may affect private decisions.** Households and firms can be expected to incorporate uncertainty when making decisions such as what actions to take to reduce their own exposures to risk or what investments to make in response to regulation. As with other aspects of behavioral responses, to measure benefits and costs of a regulation it is important to clearly characterize the behavior of firms and households. As described in Section 5.5.1, analysts should generally assume utility or profit maximization under uncertainty, taking preferences over uncertainty as given. For households, for example, this generally means a model of expected utility maximization with whatever degree of risk aversion best represents the affected households. In practice it also means that existing estimates of willingness-to-pay will reflect the risk preferences of the populations analyzed. See Section 5.5.1 for more information on representing economic behavior and behavioral responses.
- **Lay and expert risk perceptions**. Lay perceptions of risk may differ significantly from scientific assessments of the same risk. An extensive literature has developed on the topic.[46] Because individuals respond according to their own risk perceptions, it is important for the analyst to be attentive to situations where there is an obvious divergence in these two measures. In such cases, analyses should clearly state the basis for the economic value estimates used in their analysis and should also consider describing the known differences between public risk perceptions and scientific risk assessments. It may also be useful for the regulation to provide information to the public that may reduce these differences and that may allay public concerns.
- **Provision of information**. Some policy actions focus on providing information to individuals on risks to health and welfare. If this information allows them to make better decisions that improve household welfare, there is an economic benefit to providing this information. When this is the case, revealed preference approaches can make new information appear to have a net negative effect on household welfare because households may undertake new (and costly) activities in response. For example, information on drinking water quality may lead consumers to buy and use costly filtration systems at home, which could be misconstrued to mean that providing the information diminished consumer welfare. An appropriate framework for evaluating the benefits of information provision

46 For a general overview see Renner et al. 2015.

under these circumstances is to assess the costs of sub-optimal household decisions under the less complete information.[47] Analysts should carefully consider these issues when they evaluate policies that focus on information provision.

- **Option value**: Some environmental policies involve irreversible decisions made in the face of uncertainty. If information that reduces this uncertainty can be expected to develop over time, then there is a positive value to waiting until this information becomes available.[48] In this case, the value originates from the option to hold off making the decision until uncertainties are resolved or reduced. An analysis can show the potential costs of making a decision without this new information. The potential gains from waiting may best be evaluated in a value-of-information framework where the gains in net benefits from having better information can be compared to the costs associated with gathering this information, which includes any forgone benefits due to postponing environmental protection.[49]

  Generally, it is difficult to quantitatively include these option values in an analysis, but the concept is useful and may be highlighted qualitatively if circumstances warrant. Further, this is an important concept to keep in mind when considering policy approaches. As described in Section 4.6.5 it may be useful to examine approaches such as voluntary programs or pilot projects designed to gather information to make a more informed analysis of the benefits and costs of regulatory approaches.

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

48 This is sometimes known as quasi-option value, starting with the seminal work of Arrow and Fisher (1974). A slightly different framing is "real options" analysis following Dixit and Pindyck (1994). These approaches yield option values that differ slightly but capture the same concept. Traeger (2014) describes the precise relation between the two and how they can be considered in benefit-cost analysis.

49 Examples of value of information analysis include Marchese et al. (2018) and von Winterfeldt et al. (2020).

Chapter 5 References

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Chapter 6 - Discounting Future Benefits and Costs

Discounting allows for economically consistent comparisons of benefits and costs that occur in different time periods. In practice, it is accomplished by multiplying changes in future consumption (including market and non-market goods and services) by a discount factor. Discounting reflects that (1) people prefer consumption today over consumption in the future, and (2) invested capital is productive and provides greater consumption in the future. Properly applied, discounting can tell us how much future benefits and costs are worth today.

Social discounting is the main type of discounting discussed in this chapter. This is discounting from the broad society-as-a-whole point of view embodied in benefit-cost analysis (BCA). *Private discounting*, on the other hand, is discounting from the specific, limited perspective of private individuals or firms. This distinction is important to maintain.

This chapter addresses discounting over the relatively near term, called *intragenerational discounting,* as well as discounting over much longer time horizons, or *intergenerational* discounting. Intragenerational (a.k.a., *conventional*) discounting applies to contexts that may have decades-long time frames, but where the timeframe of analysis is within the lifetime of current generations. Intergenerational discounting addresses very long time horizons in which the discounted effects will impact generations to come.

This chapter focuses on the most important discounting issues for applied policy analysis, beginning with practical, basic mechanics and methods for discounting. It then turns to the theory and foundational logic for discounting and the different approaches to estimating discount rates. The presentation of the results should include the full stream of the benefits and costs over the time horizon of analysis both without discounting and appropriately discounted. Analysts should present results using both a consumption rate of interest and, if appropriate, a sensitivity analysis reflecting the shadow price of capital approach.[1]

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1 This chapter summarizes some key aspects of the core literature on social discounting, but it is not a detailed review of the vast and varied social discounting literature. Excellent sources for additional information are: Lind

6.1 Mechanics and Methods for Discounting

The most common methods for discounting involve estimating either net present values or annualized values.[2] An alternative method is to estimate a net future value. Net present value, annualization, and net future value are different ways to express and compare the costs and benefits of a policy in a consistent manner. These three methods will be discussed below.

6.1.1 Net Present Value (NPV)

The net present value (NPV) of a stream of benefits and costs in the future is the value that those benefits and costs provide to society today. The NPV at time 0 (the year to which values are discounted) of a projected stream of current and future benefits and costs is calculated by multiplying the benefits and costs in each year by a time-dependent weight, or discount factor, *d*, and aggregating all of the weighted values. This can be done by discounting the benefits and subtracting the discounted costs, which is equivalent to discounting the net benefits over all *n* years, (*n* is the number of years in the future until the last year of the time horizon of the analysis) as shown in the following equation:

$$\begin{aligned} NPV &= B0 + d1B1 + \ldots + dn{-}1Bn{-}1 + dnBn \\ &\quad - C0 - d1C1 - \ldots - dn{-}1Cn{-}1 - dnCn \\ &= NB0 + d1NB1 + d2NB2 + \\ &\quad \ldots + dn{-}1NBn{-}1 + dnNBn \end{aligned} \qquad (1)$$

where

B_t are the benefits in year t,

C_t are the costs in year t, and

NB_t are net benefits, the net difference between benefits and costs (B_t - C_t) in year t.

Alternatively, NPV can be calculated by estimating the present value (PV) of costs and the PV of benefits separately and then subtract the PV of costs from the PV of benefits:

$$NPV = \left(B_0 + \sum_{t=1}^{n} d_t B_t\right) - \left(C_0 + \sum_{t=1}^{n} d_t C_t\right). \qquad (2)$$

In either case, the discounting weights, d_t, are given by:

$$d_t = \frac{1}{(1+r)^t} \qquad (3)$$

where r is the discount rate and t is the year.

As shown in equation (1), the benefits and costs should be discounted to the same year to appropriately calculate net benefits. This is because both future benefits and costs should be

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(1982a, b; 1990; 1994), Lyon (1990, 1994), Pearce and Turner (1990), Pearce and Ulph (1994), Arrow et al. (1996), Portney and Weyant (1999), Frederick et al. (2002), Moore et al. (2004), Spackman (2004), Groom et al. (2005), Cairns (2006), Burgess and Zerbe (2011a), Moore et al. (2013a), Harberger and Jenkins (2015), Li and Pizer (2021), and Newell et al. (2022, 2024)

2 Note that discounting is distinct from inflation, although observed nominal market rates of return reflect expected inflation. While most of the discussion in this chapter focuses on real discount rates and values, benefits and costs should also be adjusted for inflation when relevant.

evaluated from the perspective of the same year to provide them equal consideration.[3] Also, as discussed in Section 6.1.6.1, the same rate should be used to discount benefits and costs in a given year. However, in some analyses with very long time horizons there may be reasons to use different discount rates in different future years, as discussed in Section 6.3.

6.1.1.1 Beginning-of-Year versus End-of-Year Discounting

In the NPV equation, B_0, C_0, and NB_0 are the benefits, costs, and net benefits incurred immediately (when t=0), so they are not multiplied by a discount factor. This makes sense when time is continuous, but what is "immediate" becomes less clear when time, t, is an entire year. For example, if a rule is finalized at the beginning of a year and costs and benefits will be realized throughout that year, are these values "immediate" or should they be discounted one period? If costs and benefits incurred throughout the year are considered immediate, then they would be B_0 and C_0 in equation (1) above. This is known as beginning-of-year discounting because all intra-annual effects in the current year are treated as if they occur at the beginning of the year, when t=0. The alternative is to treat all intra-year effects in the current year as if they occur at the end of the year, when t=1, and discount them back one period. Effects in the next year would then be discounted back two periods. This is known as end-of-year discounting. The choice between beginning- or end-of-year discounting does not generally have a large quantitative effect on the analysis. Whichever approach is adopted should be explicitly stated and applied to both benefits and costs so that the analysis is internally consistent.[4]

6.1.1.2 Time Periods of Less than One Year

When estimating the NPV, it is important to explicitly state how time periods are designated and when costs and benefits accrue within each time period. Typically, time periods are in years, but alternative time periods can be justified if costs or benefits accrue at irregular or non-annual intervals. To correctly discount intra-year effects, the annual discount rate, r, must be adjusted to an “effective rate,” $\tilde{r}$, which produces the same result as the annual discount rate if compounded for one year. The effective discount rate for any non-annual period is:

$$\tilde{r}_t = (1 + r)^{1/(\#\ of\ periods)} - 1 \qquad (4)$$

For example, if the annual discount rate is 7% and costs are incurred on a quarterly basis (i.e., there are four periods in a year), then the effective quarterly discount rate, $\tilde{r}$, is approximately 1.7%. The formula for discounting weights, d_t, given above, can be used with this effective rate, but t is measured in quarters rather than years.

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3 As discussed in Section 5.2, the analysis should identify the year to which benefits and costs are discounted and the dollar year used to report them. It is important to identify and distinguish the reporting and dollar years of the analysis when they differ.

4 Three common Excel functions used for discouning -- PMT and PV, and NPV -- use end-of-year discounting by default. The PMT and PV functions include a 0/1 "type" argument indicating if the discounting is done at the end or the beginning of the year. The default is 0, and therefore needs to be changed to 1 to do beginning-of-year discounting. The NPV function implicitly assumes end-of-year discounting. To use the NPV function to calculate the net present value using beginning-of-year discounting, the solution to the NPV calculation must be multiplied by the expression (1+r), where r is the discount rate. Analyses that use the PMT, PV or NPV functions without making these adjustments are implicitly assuming end-of-year discounting.

While the discounting formula can be adjusted to account for intra-annual discounting periods, it may not be necessary unless exact values are required. The NPV generated by an intra-annual effective discount rate, $\tilde{r}$, will be between the NPV using beginning-of-the-year discounting and the NPV using end-of-the-year discounting using the annual discount rate, r. These NPVs don't usually differ by much in a typical economic analysis.

6.1.1.3 Continuous Discounting

Costs and benefits may also be discounted on a continual basis during the year. In this case, benefits or costs occurring at the end of a future year (or period), *t,* are discounted by the weight:

$$d_t = e^{-\tilde{r}t} \tag{5}$$

Where e is Euler's number, or 2.718, when rounded to three decimal places, and is the base of the natural logarithm. This is a commonly used expression in economics and finance. Furthermore, continuous discounting provides a convenient way to represent a discount weight for some theoretical economic concepts related to discounting. Note equation (5) uses a discount rate appropriate for continuous discounting, $\tilde{r}$. As with intra-annual discounting discussed above, the effective discount rate, $\tilde{r}$, should produce the same result as the annual discount rate. The effective discount rate for continuous discounting is:

$$\tilde{r} = \ln(1 + r) \tag{6}$$

In this case, t=1 represents one year, but the discounting weight is assumed to be applied to every moment, continuously throughout the year.

6.1.2 Annualized Values

An annualized value is an illustrative cost or benefit that, if incurred every year over the entire time horizon of the analysis, would produce the same net present value (NPV) as the original time-varying stream of costs, benefits, or net benefits. In some cases, annualized values are easier to understand than NPV.

Because the annualized value is constructed to generate the same net present value as the actual stream of values, comparing annualized values is equivalent to comparing net present values. That is, one can use either the NPV or the annualized values to determine whether benefits exceed costs or which option produces the highest net benefits. As with NPV, benefits and costs may be annualized separately and compared, or the stream of net benefits can be annualized.

The formulas below illustrate the estimation of annualized costs; the formulas are identical for benefits.[5] The exact equation for annualizing depends on whether there are any immediate costs (i.e., any costs at time zero, t=0).

Annualized costs when there is no cost at $t=0$ (e.g., no C_0 in equation (1)) are estimated using the equation:

$$Annualized\ Cost = PVC * \frac{r*(1+r)^n}{(1+r)^{(n+1)}-1} \tag{7}$$

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5 Variants of these formulas may be common in specific contexts. See, for example, the Equivalent Uniform Annual Cost approach in the EPA's Air Pollution Control Cost Manual (U.S. EPA 2017).

where

Annualized Cost = annualized cost accrued at the end of each of *n* years,

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

r = the discount rate per year, and

n = the length of the time horizon over which costs are annualized.

Annualized costs when there is initial cost at $t=0$ are estimated using a slightly different equation:

$$Annualized\ Cost = PVC * \frac{r*(1+r)^n}{(1+r)^n - 1} \tag{8}$$

The annualization approach in equation (7) is generally consistent with end-of-year discounting because the first cost value, C_1, is discounted one period. Equation (8) is more consistent with beginning-of-year discounting because there is a value, C_0, which is not discounted one period.[6] Note that the numerator expression is the same in both equations, although the PVC is calculated differently depending upon whether there are costs at $t=0$. The only difference is the "n+1" and "n" terms in the denominator of (7) and (8).

Some important caveats are associated with the use of annualized values. First, they are generally illustrative; the annualized value is not the actual value that will manifest every year. Second, the annualized value changes with the timeframe of the annualization. This means that the annualized value will be *different* for each value of *n,* even for the same discount rate, *r*. The longer the timeframe assumed for the annualization, the lower the annualized value. Third, annualization formulas assume that the timeframe of *n* periods begins immediately. If the actual stream of costs being annualized does not occur immediately, the timeframe for the annualized value and the actual stream will not be the same.

One special case of equation (7), the annualization formula when there is no cost at $t=0$, is when $n=\infty$. In this case, the annualized cost is simply:

$$Annualized\ Cost = PVC * r \tag{9}$$

For example, suppose an action permanently eliminates the use of an environmental amenity (e.g., a wetland), and the estimated present value of that amenity is $1 million at a discount rate of 2%. The cost of this policy is the lost value of the amenity in perpetuity -- the period of the analysis is effectively infinity. The annualized cost of that policy - that is, the cost that if lost every year, *forever*, would be equivalent to $1 million in present value today - is $1 million ∗ 2% = $20,000 per year.

The corollary to equation (9) is:

$$PVC = \frac{Annualized\ Cost}{r} \tag{10}$$

Thus, if an environmental amenity is estimated to be worth $20,000 per year, its present value using a %2 discount rate is $1 million, assuming that it provides benefits into perpetuity.

6 The default PMT function in Excel (with "type" equal to 0) will produce the same answer as equation (7). Setting the "type" variable to 1 will produce the result from equation (8).

6.1.3 Net Future Value (NFV)

Instead of discounting all future values to the present using the NPV, it is possible to estimate the stream of values from the perspective of some future year, for example, at the end of the last year of the policy's effects, *n*. This would be the net future value (NFV). This might be particularly useful when conducting a retrospective analysis.

The net future value for net benefits (NB_t) is estimated using the following equation:

$$NFV = a_0 NB_0 + a_1 NB_1 + a_2 NB_2 + \ldots + a_{n-1} NB_{n-1} + NB_n \qquad (11)$$

Where, as before, NB_t are net benefits, (B_t - C_t), in year *t*. This formula can also be used for either benefits or costs alone.

In the NFV equation, the *accumulation* weights, a_t, are different from the discounting weights in equation (3) used for NPV, and are given by:

$$a_t = (1 + r)^{(n-t)} \qquad (12)$$

where *r* is again the annual discount rate. The net future value for year *n* can be expressed in relation to the net present value for *t*=0, as follows:

$$NPV = \frac{NFV}{(1+r)^n} \qquad (13)$$

The NFV can be modified for intra-annual values using an effective discount rate described in the NPV section above. It can also be calculated assuming continuous accumulation using the effective discount rate in equation (6) and accumulation weights:

$$a_t = e^{\tilde{r}t} \qquad (14)$$

The only difference between equation (14) and equation (5) is the use of $\tilde{r}$ rather than -$\tilde{r}$ in the exponent.

6.1.4 Comparing the Methods

NPV represents the value of a stream of costs and benefits from some point in time (often the present moment) going forward. NFV represents the value of the stream of costs and benefits at some future time. Annualization is the calculation of a constant, annual value for costs and benefits that would produce the same NPV as the actual stream of costs and benefits.

Depending on the circumstances or application of the analysis, one method might have certain advantages over the others. Discounting to the present to get an NPV is likely to be the most informative for the standard economic analysis of a policy that will generate future benefits and costs. NFV may be more appropriate for evaluating the cumulative impacts of regulation or when conducting a retrospective analysis. The difference between the two is simply the choice of the reporting or perspective year for the analysis. Annualized values may be used in conjunction with the NPV to communicate the result or compare options when the costs or benefits are highly variable over time. It is important to remember, however, that annualized values assume that the annualization period begins immediately and that the results are sensitive to the annualization period – the annualized value will be lower the longer the annualization period -- so analysts should be aware of potentially different annualization periods when comparing annualized values from one analysis to those from another.

The choice of discount rate affects the values generated by these discounting methods. 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 either higher or lower depending on the time at which impacts occur and the length of time over which the values are annualized. However, the ranking of monetized net-benefits among regulatory alternatives is unchanged across these three methods for any discount rate.

6.1.5 Sensitivity of Net Present Value Estimates to the Discount Rate

Both the size and sign of NPV can be sensitive to the choice of discount rate when there is a significant difference in the timing of costs and benefits. This is the case for policies that require large initial outlays or have long delays before benefits are realized, as do many EPA policies. Text Box 6.1 illustrates how discount rates affect NPV.

In other cases, the discount rate is not likely to affect the sign of the NPV estimate. Specifically, the NPV will not be affected by the discount rate when:

- All effects occur in the same period. In this case, discounting may be unnecessary or superfluous because net benefits are positive or negative regardless of the discount rate used.
- Costs and benefits of a policy occur consistently over the period of the analysis, or their relative values do not change over time.

In these cases, whether the NPV is positive does not depend on the discount rate, but the discount rate can still affect how the present value compares to another policy.

6.1.6 Issues in Discounting Applications

Several important analytic components need to be considered when discounting costs and benefits.

6.1.6.1 Consistent Use of the Discount Rate

The same discount rate must be used for both benefits and costs occuring in a given year, as the discount rate reflects society's intertemporal preferences for trading off consumption over time independent of the sign of the change in consumption. This is necessary for a consistent comparison of net-benefits across policy alternatives and helps prevent discounting from being used to justify a particular policy. A high discount rate reduces the weight given to costs and benefits in the future and minimizes their impact on the NPV, whereas a low discount rate weights future impacts more heavily and increases their impact on the NPV. Therefore, almost any policy can be arbitrarily justified by using separate discount rates for benefits and costs.

Text Box 6.1 - Potential Effects of Discounting

To illustrate how different discount rates affect net present value, consider an example where the benefits of a given program occur 30 years in the future and are valued (in real terms) at $5 billion at that time. The rate at which the $5 billion future benefits is discounted can dramatically alter the economic assessment of the policy: $5 billion 30 years in the future discounted at 1% is worth $3.71 billion in the present, 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, changing the discount rate from 1% to 10% generates more than an order of magnitude of difference in the present value of benefits. Longer time horizons will produce even more dramatic effects of discounting on a policy's NPV. After 100 years, the present value of $5 billion is $260 million at 3% and only $5.8 million at 7% (see Section 6.3 on intergenerational discounting). Particularly in the case where costs are incurred in the present and therefore are not affected by the discount rate, it is easy to see that the choice of the discount rate can determine whether a policy has positive or negative net benefits.

6.1.6.2 Future Value of Environmental Effects and Uncertainty

There are two issues that are sometimes confounded with social discounting and the choice of social discount rate, but should be treated separately: how the value of environmental impacts change over time, and when future benefits and costs may be uncertain. While these issues are important, particularly in an intergenerational context, they both should be addressed separately in the economic analysis rather than adjusting the discount rate to account for them.[7]

First, the future value of environmental effects (i.e., their "current price" in future years) depends on many factors, including the availability of substitutes and the level of wealth in the future. For example, the relative price of environmental goods in the future will rise if those environmental goods become scarcer over time. These changes in relative prices should be applied to future effects and the associated values discounted, but the discount rate should not be adjusted to incorporate a change in relative prices.

Second, uncertainty or risk in future benefits and costs resulting from the policy should not be incorporated into the social discount rate. While it is technically possible to adjust the discount rate to account for uncertainty, doing so may hide important assumptions and information about the relative effects of discounting and uncertainty from decision-makers. Uncertainty about future values should be treated separately when discounting. However, uncertainty about the discount rate itself is different from uncertainty about future benefits and costs and can affect discounting as discussed in Section 6.3.3.

6.1.6.3 Placing Effects in Time

Placing effects properly in time is essential for all calculations involving discounting. As discussed in Section 5.4, analyses should account for implementation schedules and the resulting changes in emissions or environmental quality, including possible changes in behavior that occur between the announcement of policy and compliance deadlines. Additionally, a lag may occur between changes

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7 See, for example, Moore et al. 2017.

in environmental quality and the corresponding change in welfare. It is the change in welfare which defines economic value, and not the change in environmental quality itself. The EPA's Science Advisory Board addressed this issue (U.S. EPA 2001a) for the 2001 Arsenic Rule (U.S. EPA 2001b). If exposure to arsenic in drinking water is reduced, the number of cancer cases is expected to decline over time to a lower, steady-state level. How fast this reduction in risk occurs depends on the "cessation-lag" following reduction in exposure. Whenever values are estimated for future periods, the analysis should also report those values discounted to the present to allow for a proper comparison across periods and avoid a potential misunderstanding regarding the magnitude of a future period's benefits and/or costs relative to those in the current period, all else equal.

6.1.6.4 Period of the Analysis

As described in Section 5.4, the guiding principle is that the time horizon should be sufficient to capture all of the welfare effects from policy alternatives, subject to available resources. This principle is based on the requirement that BCAs reflect the welfare outcomes of those affected by the policy. A complete BCA accounts for all welfare changes over the entire time period that an action is expected to yield benefits and costs.

Analysts should avoid presenting the net benefits or effects of a regulation for a single snapshot period (e.g., a year), as it is likely incomplete and, therefore, cannot be used to draw clear conclusions about the overall impact of the regulation. For example, consider the case where a regulation requires capital expenditures in the first year and has no subsequent costs, but benefits are realized over many years following the initial investment. Presenting the benefits and costs of the rule in only the first year or only one of the subsequent years would misrepresent the regulation's expected net benefits.

Previously, n was defined as the final period in which the policy is expected to have impacts. While a complete BCA should account for impacts expected to occur in *all* future years, it may be impractical to do so. One solution is to analyze a time horizon that ends in $T<n$, such that:

$$\sum_{t=T}^{n}(B_t - C_t)\frac{1}{(1+r)^t} \leq \varepsilon, \qquad (15)$$

where ε is an acceptable estimation tolerance for the NPV of the policy's net benefits. That is, the time horizon should be long enough that the net benefits for all future years beyond T are expected to be negligible when discounted to the present. In practice, however, it is not always obvious when this will occur. For instance, it is not necessarily possible to anticipate whether or when the policy will become obsolete or “non-binding” due, say, to exogenous technological changes or how long the capital investments or displacements caused by the policy may persist.

A symmetric approach may be used to identify the appropriate starting point of the time horizon of the analysis. For example, the time horizon of analysis may begin in year τ, rather than in the year the regulation is promulgated, 0, if in the years from 0 to τ-1 the net benefits are zero or sufficiently small. Note that this guidance should be used cautiously. Even if the net benefits are expected to be negligible in early or later periods, if the benefits or costs are large during those periods relative to total benefits or costs, then the analysis should account for them to be clear about the impacts over time.

As a practical matter, other than identifying points in time before and after which benefits or costs are negligible, a reasonable time horizon of the analysis may, for example, be informed by:

- The expected life of capital investments required by, or expected from, the policy (e.g., when the emissions of flow pollutants are affected);

- Statutory or other requirements for the policy or the analysis; or
- The extent to which benefits and costs are allocated to future generations.

Section 5.4 elaborates on how these first two bullets may influence the time horizon of analysis, while Section 6.3 elaborates on the third.[8]

The choice of time horizon for the analysis should be clearly explained and well-documented, 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. If annualized values are reported — then both annualized benefits and costs should be be reported and the time horizon over which benefits and costs are annualized should be the same and clearly documented. Furthermore, an annual value of benefits (or costs) should not be compared to an annualized value of costs (or benefits) because, as discussed in Section 6.1.2, annual and annualized values represent different time scales of analysis.

6.1.6.5 Discounting Non-Monetized Effects

A common criticism of discounting for environmental policies is that health impacts such as "lives saved" or physical impacts such as "improved water quality" are not like money flows. They cannot be deposited in a bank and withdrawn after earning interest. This criticism does not appreciate that the valuation approaches are designed to estimate the amount of money that is as valuable to individuals as the environmental or health effects being examined. If all environmental and health impacts have been appropriately valued (monetized), then those money-equivalent flows can be discounted like real money flows over time.

However, some effects cannot always be monetized. In this case, the undiscounted stream of the non-monetized effects should be presented as they occur over time. As a general matter, these non-monetized effects should still be discounted in benefit-cost analysis and cost-effectiveness analysis if they are aggregated over time. This is because they are assumed to hold some value, albeit unspecified, and discounting assumes that individuals prefer the benefit of that value today over the future. This is the usual practice in cost-effectiveness analysis (Section 7.5.2.1), where monetized costs and non-monetized effectiveness measures are both discounted. OMB Circular A-4 (2023) recommends discounting non-monetized health effects.

For some effects, however, the (unknown) marginal value of a change in the non-monetized effect might be dependent upon the level and timing of that change. That is, marginal values are not constant over time. For example, suppose there are annual emissions thresholds below which environmental effects are negligible, but above which lead to major environmental damages. The economic value of emissions depends upon whether those emissions are above or below this threshold, and discounting these economic values would be appropriate. If we lack these values, however, and discount the effects themselves, we are treating all changes as if they had the same value. Here, it would be preferable to display the undiscounted stream of non-monetized effects with an appropriate justification and explanation.

6.2 Background and Rationales for Social Discounting

The goal of social discounting is to compare benefits and costs that occur at different times based on the rate at which society is willing to make such trade-offs. The analytical and ethical foundation

8 Section 8.2.3.1 provides additional guidance on the appropriate time horizon of analysis specific to accounting for costs of compliance.

of the social discounting literature rests on the traditional test of a *potential* Pareto improvement in social welfare, whereby those who, on net, benefit from a policy could potentially compensate those who, on net, experience costs, such that everyone is at least as well off as they were before (see Chapter 1 and Appendix A). This framework casts the consequences of government policies in terms of individuals contemplating changes in their own consumption over time.[9] In this context, trade-offs (benefits vs. costs) reflect the preferences of those affected by the policy, and the time dimension of those trade-offs should reflect the intertemporal preferences of those affected. Thus, social discounting should seek to mimic the discounting practices of the affected individuals. Simultaneously, social discounting must reflect social trade-offs in consumption over time, which may differ from trade-offs from a private, individual perspective.

The literature on discounting often uses a variety of terms to describe identical or very similar key concepts. For the purposes of the *Guidelines*, the following fundamental concepts are used in defining a social discount rate:

- The **social rate of time preference** is the discount rate at which society is willing to trade consumption in one period (usually year) for consumption in the next period.
- **Consumption rate of interest** is the rate at which an individual is willing to trade consumption in one period for consumption in the next period. This rate reflects the individual's rate of time preference and, following the potential Pareto principle, the social rate of time preference should be based on this individual rate.
- The **social opportunity cost of capital** is the consumption allowed in the next period due to private investment in the prior period. This is the rate at which society can trade consumption over time due to productive capital. Benefits and costs should account for future consumption changes due to changes in private investment.
- **Market interest rates** are what we observe in markets for loanable funds. There are several real market interest rates which, to varying extents and accounting for tax distortions, can be taken as estimates for the individual rates of time preferences and the social opportunity cost of capital.

Social discounting is primarily concerned with the relationships among these concepts and how they are measured.

6.2.1 Consumption Rate of Interest and Social Opportunity Cost of Capital

If capital markets were perfect and complete with no distortions or uncertainties, the market interest rate would equal both the consumption rate of interest and the social opportunity cost of capital since it reflects both how individuals value present versus future consumption and how productive capital can be transformed into future consumption. Following the potential Pareto principle and valuing future costs and benefits in the same way as the affected individuals, this market rate would be the appropriate social discount rate.

However, perfect and complete markets do not exist. Private sector returns are taxed (often at multiple levels), capital markets are not perfect, and capital investments often involve private (and not necessarily social) risks. These factors cause a divergence in the consumption rate of interest

9 The term consumption is broadly defined to include both the use of both private and public goods and services by households in BCA and includes the intergenerational nature of this change in consumption.

and the social opportunity cost of capital. That is, there is a divergence between the rates at which *individuals* and *society* can trade consumption over time. Text Box 6.2 illustrates how these rates can differ.

A large body of economic literature analyzes the implications for social discounting of divergences between the consumption rate of interest and the social opportunity cost of capital. Most of this literature is based on the evaluation of public projects, but many of the insights still apply to regulatory BCA, and the dominant approaches from the literature are briefly outlined here. More complete recent reviews can be found in Spackman (2004), Burgess and Zerbe (2011a), Moore et al. (2013a, 2013b), and Harberger and Jenkins (2015). Section 6.2.2 discusses social discounting using the consumption rate of interest as the social rate of time preference, whereas Sections 6.2.3 and 6.2.4 discuss methods for discounting when investment changes.

6.2.2 Social Rate of Time Preference as the Social Discount Rate

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 for evaluating public policy decisions.

But the social rate of time preference differs from individual rates of time preference. An 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 are routinely observed to have several different types of savings, each possibly yielding different returns, while simultaneously borrowing at different market interest rates. For these and other reasons, the social rate of time preference is not directly observable and may not equal any particular market interest rate. Generally, there are two primary approaches to deriving the social rate of time preference.

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

One common approach to estimate the social rate of time preference is to use 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 the government should value policy-related consumption changes as individuals do. In this approach, the social discount rate should equal the consumption rate of interest found in the market.

In principle, estimates of the consumption rate of interest could be based on after-tax interest rates consumers face for either saving (i.e., lending) or borrowing. Because individuals have different marginal tax brackets, different levels of assets, and different opportunities to borrow and invest, the type of market interest rate that best reflects the consumption rate of interest 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 available to the public will provide a reasonable estimate of the consumption rate of interest for society.

The historical rate of return on long-term government bonds, after-tax and in real terms, is a useful measure as it is relatively risk-free, maintaining the distinction between risk and social discounting described in Section 6.1.6. Also, as long-term instruments, they provide more information on how individuals value future benefits over time frames more relevant for environmental policy analysis.

Text Box 6.2 - Social and Consumption Rates of Interest

The following example illustrates how the return on private sector investments may differ from the consumption rate of interest. Suppose a private sector investment for one period is returned as consumption, the real pre-tax market rate of return on those investments is 5% and that taxes on capital income amount to 40% of the rate of return. In this case, the private investment yields a 5% return, 2% is paid in taxes to the government and individuals receive the remaining 3%. From a social perspective, current consumption - if it were instead invested in capital - can be traded for future consumption at a rate of 5%, with 3% going to individuals and 2% going to the government. But from the individuals' perspective, they are effectively trading consumption through time at a rate of 3%. Therefore, the consumption rate of interest is 3% and the social rate of return on private sector investments (also known as the social opportunity cost of capital) is 5%.

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

A second option is to construct the consumption rate of interest as the social rate of time preference in a framework attributed to Ramsey (1928), which explicitly reflects: (1) preferences for utility in one period relative to utility in a later period; and (2) the value of additional consumption as income changes. These factors are combined in the equation:

$$r = \rho + \eta g \qquad (16)$$

where

r = the consumption rate of interest,

ρ = the pure rate of time preference,

η = the elasticity of marginal utility with respect to consumption, and

g = the consumption growth rate.

The pure rate of time preference, ρ, is the rate at which the representative individual discounts utility in future periods due to a preference for utility sooner rather than later. The elasticity of marginal utility with respect to consumption, η, defines the rate at which the well-being from an additional dollar of consumption declines with the total level of consumption. The consumption growth rate, g, defines how consumption is expected to grow over time. For example, it may be expected to increase because incomes are expected to increase over time. Estimating a social rate of time preference in this framework requires information on each of these arguments. While η and g can be derived from data, ρ is unobservable and must be assumed or calibrated.[10] Text Box 6.3 provides a more detailed discussion of the Ramsey equation, and Section 6.3.1 discusses using the Ramsey framework to guide intergenerational discounting.

10 The Science Advisory Board defined discounting based on a Ramsey equation as the "demand-side" approach, noting that the value judgments required for the pure rate of time preference make it an inherently subjective concept (U.S. EPA 2004). However, recent research has developed methodologies to calibrate the pure rate of time preference using a descriptive approach (Newell et al., 2022).

Text Box 6.3 – The Ramsey Discounting Framework

The Ramsey discounting framework provides an intuitive approach to thinking about, and potentially calibrating, the social discount rate. It can be derived by considering a representative individual with utility $u(c_t)$ in period t, where c_t denotes consumption. The agent is assumed to make choices to maximize lifetime welfare, $\int_0^T e^{-\rho t} u(c_t) dt$, where ρ is the pure rate of time preference (i.e., the rate at which the agent discounts utility) and $e^{-\rho t}$ is the discount factor. Suppose the agent is considering a one period investment of one dollar in consumption at time t for additional consumption at time $t+1$. The minimum investment rate of return, r, required for the individual to find the investment desirable is defined by the equation:

$$\frac{du}{dc_t} = e^{-\rho}\left(e^{r}\frac{du}{dc_{t+1}}\right) = e^{r-\rho}\frac{du}{dc_{t+1}} \qquad (6.3.1)$$

That is, to be worthwhile the increased utility of consumption in the second period, $\left(e^{r}\frac{du}{dc_{t+1}}\right)$, discounted back at the pure rate of time preference, ρ, must be at least equal to the forgone utility of consumption forgone in the first period to fund the investment, which could be induced by a regulatory action. The rate r defines the additional return, beyond recovering the initial investment, required for the agent to be just as well off as before. So, r represents the discount rate appropriate for comparing a future change in consumption with a change in present consumption.

If it is assumed, as is common, that the utility function has an iso-elastic form, such that $u(c_t) = \frac{c_t^{1-\eta}}{1-\eta}$, where η is the absolute value of the elasticity of marginal utility, the Ramsey formula can be recovered. Substituting this utility function into equation (6.3.1), taking the natural log of both sides of the equation, applying the relationships $ln(a) - ln(b) = ln(a/b)$ and $ln(a^m) = m\, ln(a)$, and solving for r produces:

$$r = \rho + \eta\, ln\left(\frac{c_{t+1}}{c_t}\right) = \rho + \eta g \qquad (6.3.2)$$

where g is the rate of growth of consumption between t and $t+1$.

This definition highlights two reasons that future changes in consumption should be discounted (as described in section 6.2.2.2).

1. The pure rate of time preference, ρ, captures the general preference by individuals for utlity sooner rather than later and measures the rate at which individuals discount their own future utility.
2. The term, ηg represents that a marginal change in consumption in the future may not have the same value as a marginal change in consumption today**.** For example, if consumption increases over time, the marginal utility of consumption will decrease over time, implying that a marginal change in future consumption is valued less (and discounted more) than a marginal change in current consumption.

As shown by Ramsey (1928), in an economy with no taxes, market failures or other distortions, the social discount rate r, as defined in equation (6.3.2), would be expected to equal the market interest rate. The market interest rate, in turn, would be equal to the social rate of return on private investments and the consumption rate of interest. However, distortions and market failures cause these rates to diverge in practice. As such, r represents the consumption rate of interest.

6.2.3 Social Opportunity Cost of Capital as the Social Discount Rate

The social opportunity cost of capital recognizes that the social return to private investments may exceed the private returns. Therefore, if funding for government projects or capital investments required to comply with government regulations displace total private investment in the economy the opportunity cost of those forgone investments may exceed the private returns. In other words, if a regulation displaces private investments, society will lose the total returns from those forgone investments, including the tax revenues generated.

Private capital investments might be displaced if public projects are financed with government debt and government borrowing crowds out private investment. In a regulatory context, private investment might be displaced if regulated firms cannot pass through capital expenses to households and the supply of investment capital is relatively fixed. In these cases, demand pressure in the investment market will tend to raise market interest rates and reduce private investments that would otherwise have been made.[11] A BCA should account for the full social cost of any declines in private capital investments due the policy being evaluated (and similarly the social benefits of any increase in private capital investments induced by the policy), as appropriate.

In principle, the social opportunity cost of capital can be estimated by a pre-tax, marginal, risk-free rate of return on private investments, but this rate is not observed in the marketplace. As a result, these values are sometimes derived by using National Accounts data to estimate rates of return on reproducible capital (e.g., Burgess and Zerbe 2011b; Harberger and Jenkins 2015), though there are some differences in the exact accounts included and their relative weights across these analyses. In practice, average returns that are likely to be higher than the marginal returns are typically observed, given that firms will make the most profitable investments first. This leads to uncertainty as to how marginal returns can be estimated. Observed rates also reflect an unknown risk premium faced in the private sector, which causes them to be higher than a risk-free rate.

In very specific circumstances using the social opportunity cost of capital as the social discount rate in a BCA for an environmental policy can account for the social costs of displaced capital investments. In particular, it requires that the policy costs fully crowd out private investments.[12] Harberger (1972) recognized this is unlikely to be the case and derived a generalized version of this approach, assuming that policies displace a mix of consumption and investment. In this case, the social discount rate is a weighted sum of the net pre-tax marginal return to capital (i.e., the social opportunity cost of capital) and the after-tax marginal return to capital (i.e., the consumption rate of interest). Sandmo and Drèze (1971), Drèze (1974), and Burgess (1988) extended this approach to include the marginal cost of foreign financing in an open economy. In practice, this weighted sum is likely to be closer to the consumption rate of interest than the opportunity cost capital for a number of reasons. First, the United States is a large, open economy with a high capital mobility.

11 Another justification for using the social opportunity cost of capital argues that the government should not invest (or compel investment through its policies) in any project that offers a rate of return less than the social rate of return on private investments. While it is true that social welfare will be improved if the government invests in projects that have higher values rather than lower ones, it does not follow that rates of return offered by these alternative projects define the level of the social discount rate. If individuals discount future benefits using the consumption rate of interest, the correct way to describe a project with a rate of return greater than the consumption rate is to say that it offers substantial present value net benefits.

12 The terms "displacement" and "crowding out" refers to how total private investment in the economy is reduced due to new investment in response to the environmental policy. That is, how compliance costs in response to the policy displace investment that would have occurred without the policy. An environmental policy has fully crowded out private investment if private investment is reduced by the full compliance cost of policy.

Most regulatory costs are not expected to result in a substantial displacement of capital investment, which can be funded through an increase in financing by foreign lenders. Second, the benefits of regulation could induce capital investment (e.g., by increasing productivity or reducing depreciation), which is unaccounted for in the social opportunity cost of capital approach.[13] As such, the shadow price of capital approach is the analytically preferred approach to account for the full social cost of any changes in private capital investment expected in response to the policy being analyzed.

6.2.4 Shadow Price of Capital Approach

As noted above, because capital markets are taxed and experience other market distortions, the consumption rate of interest and the social opportunity cost of capital are not equal. This means that while individuals are indifferent between consumption and the returns to risk-free private investment on the margin, society is not. The shadow price of capital approach accounts for this by adjusting the costs and benefits that affect investment into equivalent consumption impacts (i.e., their shadow values) that reflect the social value of altered private investments.[14] All impacts—the costs and benefits that affect consumption and the shadow costs and benefits of affected investments—are then discounted using the social rate of time preference that represents how society trades and values consumption over time.[15] Many sources recognize this method as the preferred analytic approach to social discounting for public projects and policies.[16]

The shadow price (or social value) of private capital investment captures the perspective that a unit of private capital produces a stream of social returns at a rate greater than that at which individuals discount them due to distortions in the capital market noted in the discussion of the social opportunity cost of capital. This is because a capital investment produces a rate of return for its owners equal to the consumption rate of interest (post-tax) plus a stream of tax revenues for the government (generally considered to be used for consumption). Text Box 6.4 illustrates this idea of the shadow price of capital.

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13 This approach has been used by the Federal government for many years and was recommended in previous EPA Guidelines (2016) and OMB Circular A-4 (2003), but it is technically incorrect and can produce NPV results substantially different from the shadow price of capital approach. For an example of these potential differences, see Spackman (2004).

14 A "shadow price" can be viewed as a good's true opportunity cost, which may not equal the market price. Adjusting the cost and benefits of investment to reflect their consumption equivalent impact is, essentially, reporting their shadow values. Lind (1982a) remains the seminal source for this approach in the social discounting literature.

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

16 See OMB Circular A-4 (2023), Freeman (2003) and the report of the EPA's Advisory Council on Clean Air Compliance Analysis (U.S. EPA 2004).

Text Box 6.4 – Calculating and Applying the Shadow Price of Capital

A highly stylized example illustrates the shadow price of capital concept. Suppose that the real pre-tax annual rate of return on private investments (i.e., the social opportunity cost of capital) is 3.5% and the post-tax consumption rate of interest is 2%. Under these conditions, $1 in private investment will produce a stream of private consumption of $.02 per year, and tax revenues of $.015 per year. Further assume that the $1 investment does not depreciate (i.e., it exists in perpetuity), the annual $0.02 net-of-tax earnings from this investment are consumed each year and the $.015 annual tax-revenue is also used for consumption in each year. The present value of the perpetual stream of constant investment income to individuals is the stream of income divided by the discount rate (Equation (10)), $0.02 / 2% = $1. The present value of the $0.015 per year stream of tax revenues discounted at 2% is $0.015/0.02 = $0.75. This is the present value of the additional benefits to society (via the government). Thus, the full social value of this $1 private investment – the shadow price of capital – is $1.75, greater than the $1 private value that individuals place on it.

This example is a highly stylized case where changes in the productive capital stock persist in perpetuity and all income from capital assets funds only consumption. A more complete derivation of the shadow price of capital, as given by Li and Pizer (2021), takes into account depreciation and the savings rate (i.e., the rate at which individuals invest income):

$$Shadow\ price\ of\ capital = \frac{(1 - savings\ rate)(gross\ rate\ of\ return\ on\ capital)}{[consumption\ rate\ of\ interest + depreciation\ rate - (savings\ rate)(gross\ rate\ of\ return\ on\ capital)]}$$

The gross rate of return on capital is the net rate of return on investments before depreciation (i.e., the social opportunity cost of capital plus the depreciation rate). Maintaining the assumptions of a 3.5% social opportunity cost of capital and a 2% consumption rate of interest, and assuming a depreciation rate of 10% and an equilibrium savings rate of 25% would yield an estimate of 1.17 for the shadow price of capital.

To apply the shadow price of capital estimate in a BCA, one needs additional information about how much investment is displaced and induced. For example, assume a large public project is financed with 75% as additional government debt and 25% through increased taxes. Further supposed that the increase in government debt displaces an equal amount of private investments and the increase in taxes reduces individuals' current consumption by an equal amount.

The shadow price of capital approach would be applied to the cost estimate in the following steps:

1. Separate the costs that displace capital investment from the costs that displace consumption.
    a. $0.75 of every $1 in costs financed through debt displaces investment.
    b. $0.25 of every $1 in costs financed through taxes displaces consumption.
2. Apply the shadow price of capital (1.17 from the example above) to the $0.75 of costs that displace private investment. This yields a shadow cost of $0.88 which accounts for the impact of these costs on investments.
3. Add to this the remaining current cost ($.25) that displaces current consumption, which is not adjusted for the shadow price of capital.
4. The total social cost of this public project is $1.13 for every $1 spent.

The same steps should be followed for the benefits estimate, separating the benefits that induce capital investment from those that directly increase consumption, to determine the total social benefits. The total social cost would then be compared to the social benefits of the project.

When compliance with environmental policies displaces private capital investments (e.g., machinery and equipment), the shadow price of capital approach adjusts any capital-displacing project or policy cost upward by the shadow price of capital (i.e., the effect of displacing capital on consumption society-wide). This calculation effectively converts changes in private investment into consumption equivalents, such that all costs and benefits can then be discounted using a social discount rate equal to the consumption rate of interest. The most complete frameworks for the shadow price of capital also recognize that while the costs of regulation might displace private capital, the benefits could induce additional private investments in capital. In principle, a complete analysis using the shadow price of capital would treat capital adjustments from costs and benefits in the same fashion.

Policies analyzed in a general equilibrium framework (Chapter 8) will implicitly apply a shadow price of capital approach. In the case of partial equilibrium analyses, additional steps are necessary to apply the shadow price of capital approach. The first step is determining whether a policy will alter private investment flows. Next, the altered private investment flows (positive and negative) are multiplied by the shadow price of capital to convert them into consumption-equivalent units. All flows of consumption and consumption equivalents are then discounted using the consumption rate of interest. A simple illustration of this method applied to the costs of a public project is shown in Text Box 6.4.

6.2.4.1 Estimating the Shadow Price of Capital

While the shadow price of capital approach provides a theoretically sound method for measuring the impact of changes in private capital investment, it is challenging to implement in practice. The Li and Pizer (2021) specification described in Text Box 6.4, requires estimates of the social rate of time preference, the social opportunity cost of capital, a depreciation rate, a savings rate, and, in particular, the extent to which regulatory costs displace private capital investment and benefits stimulate private capital investment. The first two components can be estimated as described earlier, and the depreciation rate and savings rate can be estimated from empirical data, but information on how regulation affects capital formation is more difficult to obtain, making the approach difficult to implement.[17]

How policies affect capital investment depends on whether the economy is assumed to be open or closed to trade and capital flows, and on the magnitude of the policy intervention relative to the flow of investment capital from abroad. Some argue that early analyses implicitly assumed that capital flows into the nation were either nonexistent or very insensitive to market interest rates, known as the “closed economy” assumption.[18] However, if an economy has highly mobile capital flows, including from international sources, that are sensitive to market interest rate changes (the “open economy” assumption), then total investment in private capital is likely to be less sensitive to regulatory policy interventions, and there will be little, if any, crowding out.[19] If there is no

17 In addition to Li and Pizer (2021), Lyon (1990) and Moore et al. (2004) provide reviews of how to calculate the shadow price of capital and possible settings for the various parameters that determine its magnitude. Boardman et al. (2011) contains a textbook explanation as well as empirical examples. Depending on the magnitudes of the various factors, shadow prices from 1 to infinity can result according to Lyon (1990), but the ratio of the social opportunity cost of capital to the social rate of time preference is an upper bound in the Li and Pizer (2021) specification.

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

19 See, for example, Warnock and Warnock (2009).

crowding out of private investment, then no adjustments using the shadow price of capital are necessary; benefits and costs should be discounted using the consumption rate of interest alone. The economic literature is not conclusive on the degree of crowding out and there is limited empirical evidence of a relationship between the nature and size of projects and capital displacement. This presents challenges to implementing the shadow price of capital approach outside of a general equilibrium framework.

6.2.5 Evaluating Alternative Social Discount Rate Estimates

The empirical literature for choosing a social discount rate focuses on estimating the consumption rate of interest at which individuals trade off consumption through time. Historical real rates of return on “safe” assets (post-tax), such as U.S. Treasury securities, are normally used to estimate the consumption rate of interest. Some studies and reports have found government borrowing rates range between 1.5-4%, with long-term interest rates declining for the last two decades.[20] Other studies have expanded this portfolio to include other bonds, stocks and even housing. This generally raises the range of rates slightly. It should be noted that these rates are *realized* rates of return, not anticipated, and they are somewhat sensitive to the choice of time period and the class of assets considered.[21]

Other economists have constructed a social discount rate by estimating the individual parameters in the Ramsey equation. These estimates necessarily require judgments about the pure rate of time preference. Moore et al. (2013a) and Boardman et al. (2011) estimate the social discount rate to be 3.5% under this approach. The Ramsey equation has been used more frequently for intergenerational discounting, which is addressed in the next section.

Using the social opportunity cost of capital as the social discount rate requires a situation where private investment is fully crowded out by the costs of environmental policies. This is an unlikely outcome, but it can be useful for sensitivity analysis and special cases. Estimates of the social opportunity cost of capital typically range from 4.5% to 8%, depending upon the type of data used.[22]

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20 Newell and Pizer (2003) find a 200-year average (1798-1999) rate of 4% for long-term (30-year) U.S. government bonds. According to the U.S. Congressional Budget Office (CBO) (2005), funds continuously reinvested in 10-year U.S. Treasury notes from 1789 to 2004 would have earned an average inflation-adjusted return of slightly more than 3% a year. OMB (2003) reported a 30-year average (1973-2002) pre-tax rate for 10-year U.S. Treasury notes of 3.1%. U.S. CBO (2016) estimated that the average real rate for 10-year Treasury notes was 2.9% between 1990 and 2007. Boardman et al. (2011) suggests 2.71% as the 1953-2001 average real rate of return on 10-year U.S. Treasury notes. However, the Council of Economic Advisers (CEA 2017) notes a decades-long downward trend in real rate of return for U.S. Treasury notes. Bauer and Rudebusch (2020, 2023) found that the decline in real interest rates reflects a reduction in the equilibrium real interest rate, suggesting that lower real interest rates are expected to persist. OMB Circular A-4 (2023) states that the more recent 30-year average (2003-2022) rate for 10-year Treasury marketable securities was 2.0%. U.S. EPA (2023) reported a 1991-2020 average real rate of return on 10-year Treasury securities of 1.5 to 2.0%, based on the inflation measure used. U.S. CBO (2023) projects a real interest rate on 10-year Treasury notes of 1.5% in 2033, rising to 2.2% by 2053.

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

22 OMB (2003) estimated a real, pre-tax opportunity cost of capital of 7%. Harberger and Jenkins (2015) estimate an average rate of 8% for "advanced countries." Burgess and Zerbe (2011a) estimate a rate of 6% to

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 or induce private investment, then a shadow price of capital adjustment is necessary before discounting consumption and consumption equivalents using the consumption rate of interest. Estimates of the shadow price of capital in the academic literature range from 1.1 to 2.2 (Boardman et al. 2011, Moore et al. 2013a, Li and Pizer 2021). The economic literature does not provide clear guidance on the likely scale of this displacement, but it has been suggested that if a policy is relatively small and capital markets fit an "open economy" model, there is probably little displaced investment.[23] 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 will not likely result in significant crowding out of U.S. private investments. For these reasons, some argue that for most environmental regulations, it is sufficient to discount costs and benefits with an estimate of the consumption rate of interest with sensitivity analysis as appropriate.[24]

6.3 Intergenerational Social Discounting

Policies designed to address long-term environmental problems such as global climate change, radioactive waste disposal, groundwater pollution, or biodiversity present unique challenges because they can involve significant economic effects across generations. Often, costs are imposed mainly on the current generation to achieve benefits that will accrue primarily to unborn, future generations. Discounting in this context is generally referred to as intergenerational discounting.

This section discusses the main issues associated with intergenerational social discounting using the Ramsey discounting framework as a convenient structure for considering how the "conventional" discounting procedures might need to be modified for policy analysis with very long, multi-generational time horizons. This discussion presents alternative modeling approaches to estimate the term structure, or the sequence of discount rates over time, along with important caveats when using these approaches.

Intergenerational discounting is complicated by at least three factors: (1) the "investment horizon" is longer than what is reflected in observed market interest rates representative of intertemporal consumption trade-offs made by the current generation; (2) intergenerational investment horizons involve greater uncertainty than intragenerational time horizons; and (3) future generations without a voice in the current policy process are affected. These complications limit the utility of using observed market rates to evaluate long-term public investments. The leading alternative is to use model-based approaches to forecast a discount rate representative of expected household preferences. These models suggest using a social discount rate lower than one based on recently observed market rates and conditions, especially when uncertainty over the future state of the world is taken into consideration.

The problem of comparing benefits borne by future generations to costs experienced by the current generation involves both economic and ethical questions. Therefore, the normative choice of how a

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8%, and Moore et al. (2013b) estimate a rate of approximately 5% using the same model but with different inputs. Using an approach similar to OMB (2003), CEA (2017) estimated real rates of return to capital to be around 7% based on National Accounts data but noted that approach may be subject to measurement error leading to an overestimate.

23 Lind (1990) first suggested this.

24 See Lesser and Zerbe (1994), Moore et al. (2004), and OMB (2023).

decision maker should weigh the welfare of present and future generations, along with the preferences of the current generation regarding future generations, cannot be made on economic grounds alone. Nevertheless, economics offers important insights concerning intergenerational discounting, the implications and consequences of alternative discounting methods, and the systematic consideration of uncertainty.

6.3.1 The Ramsey Framework in an Intergenerational Context

The Ramsey framework introduced in Section 6.2.2 is one of the most commonly used approaches for modeling consumption discount rates.[25] It is based on fundamental economic theory and provides an intuitive organizing framework for thinking about consumption discount rates over long time horizons. If per capita consumption grows over time — as it has since the Industrial Revolution (Valdés 1999) — then future generations will be richer than the current generation. Due to the diminishing marginal utility of consumption, increases in consumption will be valued less in future periods than they are today. In a growing economy, changes in future consumption would be given a lower weight (i.e., discounted at a higher rate) than changes in present consumption in the Ramsey framework, even setting aside discounting due to the pure rate of time preference, ρ.

This framework can be viewed in positive terms as a description (or first-order approximation) of how the economy works in practice. It can also be considered in normative terms to define how individuals should optimally consume and reinvest economic output over time. As a result, the individual parameters of the Ramsey equation can be specified using two approaches: the descriptive (or positive) approach and the prescriptive (or normative) approach.

- **The descriptive (positive) approach** attempts to calibrate the parameters of the Ramsey equation by using estimates from observed behavior. The resulting consumption discount rate reflects society's observed preferences for trading off consumption over time and the best available information on the future growth rate of consumption. Advocates of the descriptive approach generally call for inferring the discount rate from market rates of return "because of a lack of justification for choosing a social welfare function that is any different than what decision-makers [individuals] actually use" (Arrow et al. 1996). However, this can be difficult to do in practice.
- **The prescriptive (normative) approach** is based on defining a social welfare function that formalizes the normative judgments that the decision-maker wants to explicitly incorporate into the policy evaluation. In the case of the Ramsey equation, parameters would then be chosen to match these desired normative judgments.[26,27] The main argument against the prescriptive approach is that it may not be consistent with individuals' preferences for inter-temporal trade-offs revealed by their market behavior.

While the Ramsey framework is commonly used and is based on an intuitive description of the general problem of trading off current and future consumption, it has limitations. Arrow (1996)

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25 Text Box 6.3 provides a derivation of the Ramsey framework. Key literature on this topic includes Arrow et al. (1996), Lind (1994), Schelling (1995), Solow (1992), Manne (1994), Toth (1994), Sen (1982), Dasgupta (1982), Pearce and Ulph (1994), Gollier (2010), and Arrow et al. (2013).

26 Arrow et al. (1996).

27 For instance, there has been a long debate, starting with Ramsey himself, on whether the pure rate of time preference, which shows a general preference for consumption by the current as opposed to future generations, should be greater than zero when evaluating public policy decisions.

contains a detailed discussion of descriptive and prescriptive approaches to discounting over long time horizons, including examples of rates that emerge under various assumptions about components of the Ramsey equation.

6.3.2 Efficiency and Intergenerational Equity

A principal concern when policies span long time horizons is that future generations affected by the policy are not yet alive. Therefore, they cannot participate in the decision-making process and their preferences are uncertain. This is not always a severe problem for practical policy analysis. Many policies impose relatively modest costs and benefits, or have costs and benefits that begin immediately or occur in the not-too-distant future. In most cases, it suffices to assume future generations will have preferences like those of present generations. However, for policies where the costs and benefits are large and distributed asymmetrically over large expanses of time, the choice of discount rate may involve both efficiency and ethical considerations.

6.3.2.1 Efficiency Considerations

As discussed in Chapter 1 and Appendix A, the BCA efficiency test is grounded in the notion of a potential Pareto improvement, whereby those who benefit from a policy, on net, could potentially compensate those who experience costs, on net, such that everyone is at least as well off as they were before. The potential for this compensation to occur across generations hinges on the interest rate at which society can transfer wealth across long time horizons. The choice of social discount rate, therefore, contains an implicit assumption about whether, and at what price, the distribution of wealth across generations could be adjusted to compensate those who bear costs, on net. Some have argued that in the U.S. context, the federal government's borrowing rate is a good candidate for this rate, while others have argued that practical difficulties associated with implementing intergenerational transfers suggest that the Kaldor-Hicks potential compensation test is limited in its ability to assess policies affecting multiple generations.[28,29] Still others argue that the discount rate should be below market rates to correct for market distortions, and uncertainties or inefficiencies in intergenerational transfers of wealth.[30] The role of uncertainty is discussed in more detail below.

6.3.2.2 Equity Considerations

Because future generations cannot participate in decisions made by current generations, social discounting may raise ethical issues regarding the intertemporal distribution of wealth. This concern does not suggest forgoing the use of a positive discount rate but has led to suggestions that the discount rate used in intergenerational contexts should be below market rates to ensure that generations are treated equally based on ethical principles (e.g., Arrow et al. 1996, Portney and Weyant 1999).[31] One interpretation of this idea is to forgo discounting the utility of future

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

29 For more information and theoretical foundations of the Kaldor-Hicks test for potential Pareto improvements, see Appendix A.

30 Arrow et al. (1996); Weitzman (1998).

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

generations by setting the pure rate of time preference in the Ramsey framework to zero. These suggestions are for using a prescriptive (i.e., normative) approach for discounting.

6.3.3 Declining Discount Rates

Theoretical and empirical support is growing for discount rates that decline over time for intergenerational discounting (Arrow et al. 2014). That is, the appropriate rate to use in discounting effects in year 101 to year 100 will be lower than the appropriate rate to use in discounting effects in year 2 to year 1. Multiple rationales support a declining discount rate, most notably slowing consumption growth rates and uncertainty about economic growth.

6.3.3.1 Rationales for Declining Discount Rates

A slowing of consumption growth rates leads to declining discounting, as is evident from the Ramsey framework. Using a constant discount rate in BCA is technically correct only if the rate of economic growth per capita remains fixed over the time horizon of the analysis. In principle, any 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. If economic growth per capita changes over time, the discount rate will also fluctuate. In particular, an assumption that the growth rate is declining systematically over time (perhaps to reflect some physical resource limits) will lead to a declining discount rate. This is the approach taken in some models of climate change.[32]

Uncertainty about future consumption growth can also lead to a declining discount rate. The longer time horizon in an intergenerational policy context 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). These feedbacks further increase uncertainty when attempting to estimate the social discount rate. This additional uncertainty implies effective discount rates lower than those based on observed average market interest rates (Weitzman 1998, 2001; Newell and Pizer 2003; Arrow et al. 2013; Cropper et al. 2014).[33,34]

The effect of uncertainty on discount rates is a result of the fact that discounting is a non-linear operation, such that the average discount factor (i.e., $E[e^{-rt}]$) is not equal to the discount factor calculated at the average discount rate (i.e., $e^{-E[r]t}$). As an alternative to estimating the average discount factor, one can calculate the certainty equivalent discount rate schedule, which is the discount rate schedule that yields the same discount factor in any time period as the average discount factor across the possible discount rates. Uncertainty about future consumption growth will cause this certainty equivalent discount rate schedule to decline over time as the potential for low discount rates will increasingly dominate the expected NPV calculations for benefits and costs

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32 See, for example, U.S. EPA (2023).

33 This holds regardless of whether or not the estimated investment effects are predominantly measured in terms of private capital or consumption.

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

far in the future (Weitzman 1998). Text Box 6.5 provides a simple example highlighting how declining discount rates arise in this fashion.[35]

6.3.3.2 Approaches to Estimate Declining Discount Rates under Uncertainty

Declining discount rate schedules can be derived from specifications of the Ramsey formula or from historically estimated stochastic models of interest rates.

If there is uncertainty in the consumption growth rate, then the standard Ramsey formula may need to be adjusted. Incorporating uncertainty in consumption growth results in a third term being subtracted from the Ramsey formula to account for the potential of low growth futures (Gollier 2002; Arrow et al. 2014). If the shocks to consumption growth are independent and identically distributed, then the precautionary term will cause the discount rate to be lower but not decline. However, if the shocks are positively correlated over time, then the precautionary term will grow over time and cause the discount rate to decline (Goiller 2014). If there is parametric uncertainty regarding the process underlying consumption growth or the other values in the Ramsey formula, this can also lead to a declining discount rate. However, if the uncertainty in the growth rate is endogenously incorporated in the benefits or costs calculations using Monte Carlo simulations, this adjustment is unnecessary.[36]

The use of historical data to estimate a declining discount rate schedule is shown by Newell and Pizer (2003). They use 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. This is attractive as a descriptive approach because it does not require specifying uncertainty over the consumption growth rate and parameters of the Ramsey formula, but its results are sensitive to the selection of a model to represent the stochastic interest rate process (Groom et al. 2007).

Some modelers and government bodies have used fixed step functions for the discount rate term structure to approximate more rigorously-derived declining discount rate schedules and to reflect non-constant economic growth, intergeneration equity concerns, and heterogeneity in future preferences.[37] 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, no empirical evidence suggests the point(s) at which the discount rate declines, so any year selected for a change in the discount rate will be ad-hoc.

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35 While this explanation is motivated by uncertainty over long-term consumption growth, a similar result arises when there is persistent uncertainty about preferences or heterogeneity in preferences. See Heal and Millner (2014).

36 For example, see the approach taken in Newell et al. (2022).

37 For instance, in the United Kingdom, the Treasury recommends the use of a 3.5% discount rate for the first 30 years followed by a declining rate over future time periods until it reaches 1% for 301 years and beyond. The guidance also requires a lower schedule of rates, starting with 3% for zero to 30 years, where the pure rate of time preference in the Ramsey framework (the parameter r in our formulation) is set to zero. For details, see Lowe (2008). Additionally, Weitzman (2001) presents a novel approach to calibrating a fixed step discount rate schedule based on uncertainty using survey data.

Text Box 6.5 – Declining Discount Rates from Uncertainty

The term structure for the certainty equivalent discount rate may decline over time due to uncertainty about future economic conditions or social preferences. Consider a simple example where one is attempting to evaluate the net present value of a policy that yields $1 in net benefits every year, and there is uncertainty as to whether the discount rate is 2% or 4%, with each rate equally likely. Because discounting is a nonlinear operation, using the average discount rate of 3% will not provide the same result as calculating the expected net present value of the two equally likely rates. Figure 6.1a presents the present value of this stream of net benefits for time horizons from 1 year to 300 years. Using the average discount rate of 3% underestimates the average present value of the payments for long time horizons. However, the plot shows that, the difference is relatively small over short time horizons.

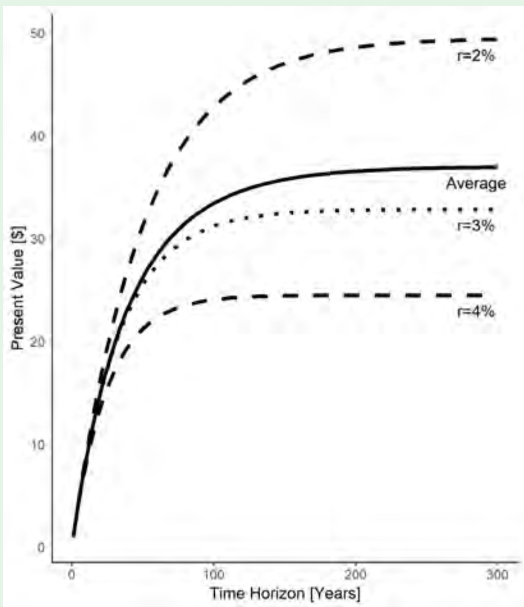

Figure 6.1a: Net Present Value

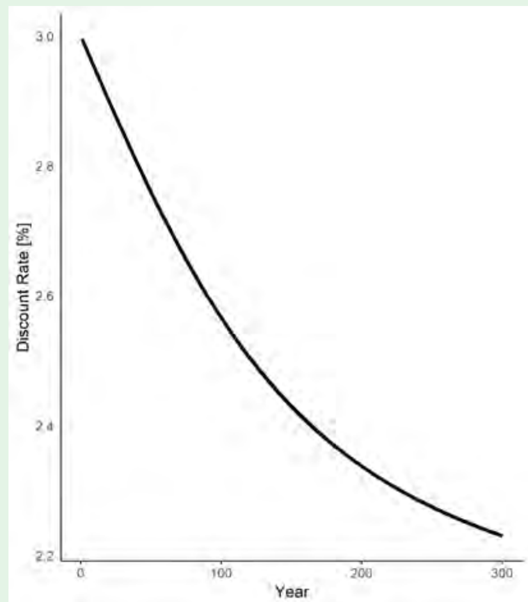

Figure 6.1b: Certainty Equivalent Discount Rate

As opposed to calculating the average net present value, one could solve for the discount rate schedule that, when applied to the problem as if there were certainty about the discount rate, yields the same present value for a particular time horizon as when explicitly accounting for uncertainty. This discount rate schedule is referred to as the certainty equivalent discount rate. Figure 6.1b presents the certainty equivalent discount rate for this example. The discount rate schedule begins close to, but below, the average discount rate of 3% and so for short time horizons the 3% and certainty equivalent discount rates have approximately equal impact on the present value. However, as one moves further out in time, the certainty equivalent discount rate declines and becomes much lower. This effect may be seen in Figure 6.1a. At the 4% discount rate, after approximately 100 years, future payments do not appreciably affect present value. However, at the 2% discount rate, extending the time horizon past 100 years appreciably increases the present value. Therefore, in terms of calculating the average present value it is the possibility of the discount rate being 2% that matters more (i.e., it dominates). This is the general effect that causes the certainty equivalent discount rate in Figure 6.1b to decline.

6.3.3.3 Consistency Issues and Declining Discount Rates

Another concern regarding declining discount rates is the potential for time inconsistency in policy recommendations over time (Arrow et al. 2014). Time inconsistency means that a net-beneficial policy today may not be net-beneficial if evaluated in the future, even when nothing has changed except for the date of the evaluation. The use of fixed step functions can exacerbate the problem. Therefore, whether an analysis shows the policy to be net-beneficial will be sensitive to the point in time the analysis is conducted. Text Box 6.6 provides an illustration of this time consistency problem.

If the analyst obtains new information between the time the original and updated analysis are conducted, the results of the analysis may have changed. However, if a fixed declining discount rate schedule is adopted and not updated between analyses to reflect the arrival of new information, that could lead to a potential time inconsistency problem (Arrow et al. 2014).

6.3.3.4 Calibration and Challenges

A wide range of potential approaches for calibrating a discount rate or a schedule of declining discount rates is available for discounting intergenerational costs and benefits. More complex analysis is justified when the proportion of costs and benefits occurring far out on the time horizon and the temporal separation of costs and benefits is large. While strong theoretical and empirical evidence shows that a declining discount rate schedule is appropriate when considering effects over long time horizons, calibration complications and concerns with time inconsistency remain notable challenges.

One possible response to such challenges is to select a constant but slightly lower discount rate when discounting costs and benefits expected to occur far out in the time horizon, reflecting a certainty equivalent discount rate. Independent of the approach or rate selected, the same discount rate should be applied to all benefits and costs that occur in the same year for both intra- or intergenerational consequences to ensure consistency in the analysis (Arrow et al. 2013).

6.4 The Role of Private Discounting in Economic Analysis

This chapter focuses on social discounting, which is discounting from the broad society-as-a-whole perspective embodied in BCA. By contrast, private discounting is the discounting of expected future benefits or costs (e.g., revenues or expenditures) from the perspective of private individuals or firms. Private discount rates reflect the preferences of specific individuals for consumption over time, as well as the prices that individuals and firms pay to borrow and lend money. These rates vary among firms, industries, and individuals due to differences in preferences, tax treatments, and costs of borrowing. Section 6.2.1 describes why market interest rates differ from the consumption rates of interest.

As previously stated, private discount rates should not be used to estimate the NPV of the social net benefits of policies and projects because the intertemporal preferences of society as a whole (as measured by the social rate of time preference) are not likely to be equal to private market lending rates or individual or firm preferences.

Text Box 6.6 – Time Inconsistency and Declining Discount Rates

Time inconsistency means that a net-beneficial policy today may not be net-beneficial if evaluated in the future, even when the only change is the date of the evaluation.

Consider the following stylized example of a declining discount rate used to analyze a policy. The discount rate schedule is a step function with 3% for benefits and costs that occur one period in the future and 0% in each period thereafter. The policy will cost $1,000 in the second period from today and will provide benefits of $1,003 in the third period. If evaluated today, the policy has positive net benefits of $e^{-0.03}$ ($e^{-0.00}$$1,003 - $1000) = $3.

However, a reevaluation of the policy in the second period would have negative net benefits of $e^{-0.03}$ $1,003 - $1000) = $-27, because costs are not discounted while the benefits in period three are discounted to period two at 3%. Therefore, whether an analysis shows the policy to be net-beneficial will be sensitive to the point in time the analysis is conducted. This is a time-inconsistent approach to discounting.

6.4.1 Predicting Private Behaviors and Choices

Private discounting should be used to predict behaviors and choices of individuals and firms in response to policy, and how investment in the economy and consumption (broadly defined) are expected to change as a result.[38] Individuals and firms can be expected to make decisions based on their own opportunity costs rather than those of society as a whole. For example, from the viewpoint of a private firm, the change in a stream of future profits due to the adoption of a pollution abatement project would be evaluated at the rate at which the firm can borrow. Similarly, the expected consumption behavior of individuals and households should be modeled consistently with how they make purchasing decisions. To predict the purchase of durable goods, for example, private evaluation and perception of the consumer's benefits and costs from using these goods over time should be used. Failure to account for choices based on appropriate private discount rates will lead to inconsistencies between the behavior of individuals and firms in the analysis and their expected behavior in the real world.[39] Therefore, private discount rates should be used to evaluate how firms and individuals will respond to policy.

6.4.2 Treatment of Interest Payments

Any changes in the amount of interest paid for borrowing (e.g., loans) resulting from a potential regulation should not be included in the calculation of its estimated social benefit or cost. Interest payments do not reflect the use of real resources such as labor, capital, and materials in an economy. Rather, the interest payment is a transfer between the borrower and lender and would net out of a social benefit-cost analysis. Private interest rates, in part, reflect the opportunity cost to society of any changes in the timing of consumption as a result of a regulation, but this opportunity

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38 This guidance applies both the regulated sources and any individuals and firms meaningfully affected by the behavior of the regulated sources.

39 For this same reason, using a social discount rate to model how firms and individuals evaluate private benefits and costs can lead to misspecification of the baseline over time and/or a mistaken projection of their responses to a policy.

cost is already accounted for in social discounting, as discussed above.[40] However, interest payments should be accounted for when evaluating the incidence and economic impacts of a regulation. For example, if a firm must take out a loan to comply with a regulation, the interest payment on that loan should be accounted for when estimating the effect of the regulation on the firm's profits.[41] See Chapter 9 for further discussion of how to determine the incidence of a regulation.

6.4.3 Selecting Private Discount Rates

Selecting which discount rate best represents household or firm behavior is a challenge. An appropriate discount rate may be observed from market behavior, but different households and firms borrow at different interest rates, and even within a household or firm, borrowing (and lending) occurs at different rates.[42] For example, firms may borrow at different rates depending on whether they are financing investments through debt or equity. Therefore, the choice of discount rate used to represent private behavior should be explained and, if necessary, sensitivity analyses using different rates should be considered.

6.5 Recommendations and Guidance

The following recommendations are intended as practical and plausible default assumptions rather than comprehensive and precise estimates of social discount rates that apply in all situations. In some analyses, there may be compelling reasons to gather data and develop a realistic model with precise empirical estimates for the factors most relevant to the specific circumstances. In such cases, these estimates should be presented along with the rationale in the description of the

40 Administrative charges on a loan (e.g., origination fees) may include the cost of preparing and administering any loans. Changes in these costs, if they can be determined, should be accounted for in a benefit-cost analysis.

41 When evaluating the incidence of a regulation over time, it may also be important to recognize the annualization of any capital investment. However, when estimating net-benefits, costs should be discounted from the period they are realized and not necessarily when they are paid for by the regulated source (or other economic actor). The private amortization schedule of financed costs should not be used.

42 As discussed in the behavioral economics literature, individual behavior is not always consistent with the conventional discounting framework. For example, households may consume and save different sources of wealth differently, and therefore are applying different discount rates to those sources of wealth, even when the sources of wealth are fungible (Thaler 1990). There is also evidence that discount rates for individuals decline over time, are lower the larger the magnitude of the future value, are higher for gains than for loses and that individuals may prefer a stream of benefits that increase over time over one that is constant over time despite each having the same nominal values (Fredrick et al. 2002). Alternative behavioral frameworks have been proposed that are consistent with these observed patterns of discounting (e.g., Loewenstein and Prelec 1992; Laibson 1998). Conventional discounting should be used to represent individual, household or firm behavior in the economic analysis, although alternative discounting frameworks to represent the behavior of individuals or households may be provided in a sensitivity analysis, provided the alternative framework is well-studied in the literature in settings comparable to that of the regulation. Care should be taken when applying alternative discounting models to predict behavior, as observed behavior that at first appears inconsistent with the conventional framework may actually be consistent with the perceived inconsistency due to omitted considerations. For example, an individual's discount rate may appear to change over time due to perceived uncertainty about future outcomes being valued, even though their strict rate of time preference may not be changing (Fredrick et al. 2002).

methods and any appropriate peer review. Results based on default assumptions should also be included for comparison purposes and consistency with OMB guidance, as appropriate. With this caveat in mind, recommendations for discounting are below.

- Display the full time paths of benefits and costs as they are projected to occur over the time horizon of analysis both without discounting and appropriately discounted.
- When determining the net benefits of a regulation, the analysis should compare the discounted value of the entire time horizon of benefits and costs. It is inappropriate to characterize the effect of a regulation with only the costs or benefits for a limited period of time, e.g., a single year, when benefits and costs may occur during other periods. Similarly, it is inappropriate to compare an annual value to an annualized value.
- Calculate the present or annualized value of social benefits and costs 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 and inducement effects are negligible). OMB (2023) recommends a real consumption rate of discount of 2% based on empirical estimates.
- To the extent that a regulation is expected to displace or induce short-term or long-term capital investment, then the shadow price of capital should be applied to the components of benefits and costs impacting this investment to convert all effects into consumption equivalents.
    - In general, there is uncertainty as to the extent to which private capital is displaced or induced by regulatory requirements. If the shadow price of capital approach is not applied explicitly or implicitly using a general equilibrium framework, then analysts should consider a sensitivity analysis consistent with OMB (2023) to understand the potential effect of capital investment changes on the discounted benefits and costs. OMB recommends considering a range of 1.0 to 1.2 as the shadow cost of capital. The sensitivity analysis should be presented separately and not part of the primary estimates of benefits, costs, or net benefits, and should be considered as a check on the robustness of the relative net benefits of the analyzed options.
- If the policy has costs or benefits that extend over a long time horizon (e.g., most benefits accrue to one generation and most costs accrue to another), then a constant consumption rate of interest may not be appropriate. The analysis should also present the net benefits under an additional approach whose rationale is clearly explained. These approaches may include:
    - Calculating the expected present value using a Monte Carlo simulation which explicitly accounts for uncertainty in the growth rate of consumption and the correlation between the growth rate and the benefits and costs.[43]
    - Calculating the expected present value of net benefits using a schedule of declining discount factors (Newell and Pizer 2003, Groom et al. 2007, Hepburn et al. 2009, OMB 2023).
- Regardless of the approach or rate selected, the same discount rate should be applied to *all* benefits and costs that occur in the same year to ensure consistency in the analysis, and

43 For example, see Newell et al. (2023).

benefits and costs should be discounted to the same year when calculating net benefits. In addition, assumptions that may influence the discount rate (e.g., the gross domestic product (GDP) growth rate) should be consistent with assumptions made elsewhere in the analysis when feasible. In cases where this is not possible (e.g., because a valuation estimate has a discounting assumption embedded in it that cannot be disentangled), the analysis should clearly explain the limitation, why it cannot be resolved, and its implications for the analysis.

When discounting future benefits and costs, the following principles should be kept in mind:

- Private discount rates should be used to predict the behavior of individuals and firms and to evaluate economic impacts and incidence, but they should not be used in place of the social discount rate to assess the social benefits and costs of a policy.
- The discount rate should reflect marginal rates of substitution between consumption in different time periods. It should not be confounded with factors such as uncertainty in benefits and costs or the value of environmental goods or other commodities in the future (i.e., the "current price" in future years).
- The economic analysis should account for the lag time between a change in regulation and the resulting welfare impacts. This includes accounting for expected changes in human health, environmental conditions, ecosystem services, and other related factors.

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Chapter 7 - Analyzing Benefits

This chapter provides an overview of the benefits analysis process, discussing the quantification of benefits first and then their monetization. The aim of a benefits analysis of an environmental policy is to describe the changes resulting from that policy and to estimate the social benefits that ensue. Willingness to pay (WTP) is the preferred measure for benefits.[1] WTP provides a full accounting of individual preferences across trade-offs between wealth and benefits and is measured in monetary terms to allow the calculation of net benefits. Net benefits are used to compare policy options and assess the magnitude of expected net improvements in societal welfare.

When analyzing benefits and costs, the *Guidelines* assumes the policy under review improves environmental quality, but at a cost, as noted in Chapter 1. Benefits, then, are reduced risks to human health and increased welfare from environmental improvements. This chapter provides tools and methods for estimating these benefits. However, these same tools and methods are equally applicable to valuing changes in environmental quality, regardless of the direction of those changes (e.g., for deregulatory policies where declines in environmental quality are assessed as a cost).[2]

Note that a benefits analysis may contain negative elements. For example, there may be increases in human health risks due to increases in emissions of a pollutant other than the one being regulated. These risk increases are costs but may be presented as negative benefits in the benefits analysis, sometimes described as “disbenefits “or “countervailing risks.” Similarly, there may be negative costs (i.e., benefits) that appear on the cost side of the benefit-cost ledger. In this way, similar kinds of effects are kept together, which is appropriate so long as it does not change the conclusion (i.e., the net benefits of various options are not affected) and the analysis is internally consistent.

This chapter highlights the benefit transfer approach, the chief method for monetizing benefits in economic analysis of regulatory actions and reviews available options for more fully incorporating endpoints that are not

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1 As described elsewhere in the Guidelines we use “willingness to pay” to refer to both willingness to pay and willingness to accept compensation concepts. Compensation that falls between the willingness to pay of those who gain and the willingness to accept of those bearing costs would also be compatible with the potential Pareto criterion.

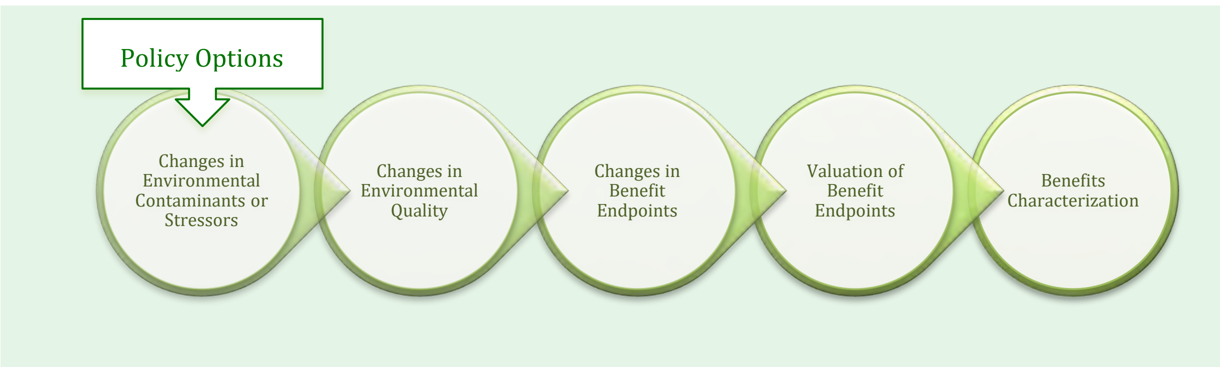
2 While this chapter focuses on the anticipated social benefits of regulation, the same approach also applies in a retrospective setting. See Chapter 5, Text Box 5.1, for more discussion of retrospective analysis.

monetized due to lack of existing values or quantification. Chapter 11, “Presentation of Analysis and Results,” presents ways to convey information on non-monetized benefits to help inform policy-making.[3]

7.1 The Benefits Analysis Process

Figure 7.1 presents a conceptual model for benefits analysis. After policy options have been identified, the first step is to identify the changes in environmental contaminants or stressors that are likely to result from policy options relative to the baseline. These may be measured as changes in emissions or in concentrations of contaminants, but they can also be considered more broadly. For example, “stressors” can be the number of hazardous waste sites, and the benefits analysis may be built upon changes in this metric.

Figure 7.1 - A Conceptual Model for Benefit Analysis



Changes in contaminants or stressors often lead to changes in environmental quality such as a change in ambient air quality. Environmental quality should be interpreted broadly for this conceptual model, including exposure to contaminants. Often, a great deal of analysis is required to project how changes in contaminants or stressors affect environmental quality, including modeling 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). In many cases, explicit modeling of human intake or exposure might be another intermediate step in the conceptual model that precedes quantifying changes in benefit endpoints.

The next step is to identify the benefit endpoints that may be affected by changes in environmental quality. Benefit endpoints are organized in the *Guidelines* into broad categories: human health

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

improvements, ecological improvements, aesthetic improvements, and reduced materials damages (Section 7.2). Table 7.1 lists examples of benefit endpoints in each of these categories. Once changes in benefit endpoints are identified, valuation follows well-defined economic principles (Section 7.2) using well-established economic methods (Section 7.3). Commonly used methods for each type of benefit are also described in Table 7.1.

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

Human Health Improvement	Examples	Commonly Used Valuation Methods
Mortality risk reductions	Reduced risk of: Cancer fatality Acute fatality	Averting behaviors Hedonics Stated preference
Morbidity risk reductions	Reduced risk of: Cancer Asthma Cognitive Impairment	Averting behaviors Cost of illness Hedonics Stated preference

Ecological Improvements	Examples	Commonly Used Valuation Methods
Market products	Food; Fuel; Timber; Fish	Production function Demand analysis for consumer benefits
Recreation activities and aesthetics	Wildlife viewing Fishing and hunting Boating Swimming Hiking Scenic views	Production function Averting behaviors Hedonics Recreation demand Stated preference
Valued ecosystem services	Climate moderation Flood moderation Groundwater recharge Sediment trapping Soil retention Nutrient cycling Pollination by wild species Biodiversity, genetic library Water filtration Soil fertilization	Production function Averting behaviors Stated preference
Non-use values	Relevant species populations, communities, or ecosystems	Stated preference

Other Benefits	Examples	Commonly Used Valuation Methods
Aesthetic improvements	Visibility Taste Odor	Averting behaviors Hedonics Stated preference
Reduced materials damages	Reduced soiling Reduced corrosion	Averting behaviors Production / cost functions
Other market and non-market goods	Reduced fuel expenditures Reduced infrastructure expenditures Enhanced energy security	Demand analysis for consumer benefits Production/cost functions Other methods as needed

Finally, the aggregate value for all benefits, including benefits arising from the primary statutory objective of the regulation as well as other benefits, provides the basis for characterizing the benefits of each policy option. Ideally, the benefits analysis would comprehensively assess all welfare-improving effects — all benefit endpoints — attributable to a rule or policy decision, including potential interactions and feedbacks between effects. This may be possible to an extent with the use of integrated assessment models (IAMs) (see Text Box 7.1 for background on IAMs). However, the modeling and data required for such a comprehensive assessment make it difficult to do so in most circumstances.

Benefits analysis need not proceed by enumerating all benefit endpoints separately or follow the specific sequence described in Figure 7.1, particularly if valuation estimates are linked to effects further upstream in the model. For example, rather than monetizing enumerated health benefit endpoints, the hedonic property method (Section 7.3.1.3) may estimate the total value to residents of changes in the presence of hazardous waste sites – a change in a stressor in Figure 7.1 – by linking policy changes to changes in property values. This valuation estimate could then be used in benefits analysis. This method of assessing benefits can be viewed as a reduced form approach to the modeling.[4] Even when viewed as a reduced form approach, however, it is important to think through the conceptual model to assess whether there are benefit endpoints not reflected in the reduced form valuation estimate that should be included through additional analysis.

A General Approach to Benefits Estimation

Ultimately, benefits analysis should link policy changes to the value of all benefits that can be meaningfully attributed to those changes. This is most often done using a pragmatic, general approach aligned with the conceptual model in Figure 7.1 by tracing policy-related changes through a set of models to predict changes in specific benefit endpoints, then valuing each endpoint, or sometimes sets of endpoints, separately. An overall estimate of total benefits is the sum of these separate components.

4 There are many other ways this type of reduced form approach may be appropriately used, sometimes including estimates of the benefits per unit of environmental contaminants that are reduced.

Text Box 7.1 - Types of Benefits Associated with Environmental Policies: Categories, Examples and Commonly Used Valuation Methods

Integrated assessment models (IAM) are sometimes used to estimate the benefits of a policy. In the broadest sense, IAMs are "approaches that integrate knowledge from two or more domains into a single framework" (Nordhaus 2013), and this class of model has been used in many disciplines, including earth sciences, biological sciences, environmental engineering, economics and sociology. In environmental economics, IAMs combine natural processes and economic systems into a single modeling framework. These models "connect economic activity with environmental consequences, and ultimately, with valuation" (Keiser and Muller 2017). A full IAM will capture four components — behavior that generates emissions/pollutant loadings, pollution fate and transport, environmental and human outcomes, and valuation — as well as feedbacks within and across these components. It also aims to capture the importance of these consequences in a transparent, reproducible way.

IAMs have been used in environmental economics to study stock pollutants, primarily greenhouse gases (GHGs) (Nordhaus 1993), and flow pollutants, e.g., air pollution (Mendelsohn 1980) and water pollution (Freeman 1979, 1982). Current IAMs vary in structure, geographic resolution and the degree to which they capture feedbacks and valuation of changes in physical endpoints and regulatory compliance costs, with research often focused on improving these representations. IAMs have been used to study the interaction between GHG mitigation and urban and regional air pollution policies (Reilly et al. 2007), the dynamic economic and ecosystem general equilibrium effects of fisheries management policy (Finnoff and Tschirhart 2008), and linkages in the food-water-energy nexus affecting policy outcomes (Kling et al. 2017). The choice of IAM will depend on the research or policy question.

IAMs are used in BCA in the valuation of changes in GHG emissions. IAMs that combine representations of climate and economic systems are used to develop monetized estimates of the damages associated with incremental emissions of carbon dioxide (CO_2), denoted as the social cost of carbon dioxide (SC-CO_2), allowing the inclusion in a BCA of social benefits of actions expected to change these. Specifically, the SC-CO_2 is the present value of the stream of future economic damages associated with an incremental increase (by convention, one metric ton) in CO_2 emissions in a particular year. It is intended to be a comprehensive measure and includes economic losses due to a wide range of anticipated climate impacts, such as net changes in agricultural productivity, human health risks, property damages from increased flood frequencies, the loss of ecosystem services, etc. Analogous metrics estimate the monetary value of climate impacts associated with other non-CO_2 GHGs, such as methane and nitrous oxide. In January 2017, the National Academies of Sciences, Engineering, and Medicine issued recommendations for research and a regularized process for updating the SC-CO_2 estimates used in federal regulatory BCA to ensure that estimates reflect the best available science. Since the framework used to estimate the social cost of non-CO_2 GHGs is the same as that used for SC-CO_2, the Academies' recommendations also apply to the estimates of the social cost of non-CO_2 GHGs. See U.S. EPA (2023) for a detailed discussion of EPA's implementation of the Academies' near-term recommendations.

The development of IAMs for use in other aspects of regulatory BCA is an emerging area of research. For example, the EPA is developing an IAM for broad-scale water quality benefits analysis by integrating hydrological and water quality modeling (HAWQS/SWAT) with an economic valuation model (BenSPLASH) (Corona et al. 2020). The Hydrologic and Water Quality System (HAWQS) is a water quantity and water quality modeling system using the Soil and Water Assessment Tool (SWAT) as its core engine. The <u>Ben</u>efits <u>S</u>patial <u>Pl</u>atform for <u>A</u>ggregating

Socioeconomics and H2O Quality (BenSPLASH) is an open-source analytical tool used to quantify the economic benefits of changes in key water quality parameters and value the surface water quality benefits of regulatory actions.

In short, the goal is to monetize those benefit endpoints that can be monetized, to quantify those that can be quantified but not monetized and to provide qualitative characterizations of what cannot be quantified. The results should then be described in a manner useful for policy makers.[5]

This general approach can be divided into three steps.

Step 1: Identify relevant benefit endpoints associated with the policy.

Step 2: Quantify significant changes in these benefit endpoints to the extent feasible.

Step 3: Monetize the changes using appropriate valuation methods or by drawing on values from existing studies.

Each step in this approach is discussed in more detail in the sections that follow. Collaboration with appropriate experts often will be necessary to execute these steps.[6]

Step 1: Identify Relevant Benefit Endpoints

The first step is to conduct an initial assessment of the types of benefits associated with the policy options being considered. This requires evaluation of how conditions and ultimately benefit endpoints differ between each policy option and baseline conditions (Chapter 5), including the current and future state of relevant economic and regulatory variables (Section 5.2). The goal for this step is to enumerate the full set of benefit endpoints and to identify those that should be further developed for quantification and valuation. In this assessment, analysts should, to the extent feasible:

- **Develop an understanding of the changes in environmental contaminants or stressors resulting from policy options.** Initially, the range of policy options being considered may be very broad. Collaboration among all analysts and policy makers involved in the policy analysis can help ensure that all potential effects are recognized. It is important to account for both contaminant or stressor changes directly targeted by the policy options and those that will occur even if not directly targeted.[7]
- **Identify the benefit endpoints likely to be affected by policy options.** This step often requires considering the transport of contaminants through the environment along many pathways, including movement through the air, surface water and groundwater. Along these pathways, the pollutants can have detrimental effects on natural resources, such as affecting oxygen availability in surface water or reducing crop yields. Pollutants can also have direct or indirect effects on human health, for example, affecting cancer incidence through direct inhalation or through ingestion of contaminated food. This step is inherently

5 See Chapter 11 for more detail on presenting qualitative, quantified and monetized benefits.

6 A summary of a large-scale benefits exercise that followed these steps is described later in Text Box 7.5.

7 See Chapter 5, Section 5.1 for additional discussion of considering benefits that arise from changes in pollutants other than those that would be directly regulated by the policy.

multi-disciplinary and will include consulting with risk assessors and other experts involved in the rule or policy, sometimes as part of a formal workgroup (U.S. EPA 2014).

- **Evaluate the potential changes in benefit endpoints resulting from each policy option**. If policy options differ only in their level of stringency, then each option may have an impact on all identified endpoints. Where policy options are more complex, however, the options may have an impact on some endpoints but not on others.
- **Determine which benefit endpoints warrant further investigation** in the overall benefits analysis using at least the following framing questions:
    - *Which benefit endpoints are likely to be large relative to total benefits or are otherwise important for informing policy decisions?* This determination should be based on an assessment of the importance of each benefit endpoint to the benefits analysis, including its potential magnitude, the extent to which it can be quantified and the extent to which it can be monetized. Preliminary assessments should be made using the best, readily available quantitative information; however, as a practical matter, these decisions are often based on professional judgment.
    - *Which benefit endpoints should be included even if they may not be large relative to total benefits?* Some benefit endpoints may not be captured by the first criteria but are important and informative for other reasons. For instance, benefit endpoints necessary to evaluate how minority, disadvantaged or susceptible groups are affected in distributional analyses (Chapter 10) may not be large at a national level but may be very important at a smaller scale. Benefit endpoints may also be important because they reflect Agency priorities, are closely related to the underlying motivation for the rule or are otherwise of particular interest to decision makers.
    - *Which benefit endpoints are likely to differ across policy options?* Analysts should assess how the effects of each policy option will differ. Benefits categories should be meaningfully attributed to policy with some degree of confidence, while recognizing that there will always be uncertainty and that this uncertainty can be characterized in the benefits analysis. Again, this may be done as part of an interdisciplinary team working on the rule or policy.
    - *How much uncertainty is associated with the benefit endpoint?* All endpoints have some uncertainty. For example, toxicologic and epidemiologic evidence may be insufficient to fully determine the likelihood that a contaminant causes a particular health effect. That said, it is important not to limit benefits endpoints at this stage even in the presence of substantial uncertainty. Highly uncertain benefits may still be important in the net benefits calculation and should generally be carried through the analysis. Willingness to pay to avoid a very uncertain probability of a severe effect could be larger than willingness to pay to avoid a more certain probability of a less serious effect (McGartland et al. 2017). At a minimum, assessing uncertainty early can inform what additional analysis is needed to effectively characterize benefits.
    - *What are the costs of undertaking analysis to characterize the benefit endpoint?* The costs of quantifying and monetizing benefit endpoints may be minimal if existing data and models can be applied. If existing data and models are insufficient, value of information considerations are important. A benefit endpoint may not be worth a great deal of further investigation if the costs to quantify and monetize it exceed its

informational value. However, consideration should also be given to other current and future rulemaking efforts that would rely on this endpoint for benefits.

The outcome of this step can be summarized in a list or matrix that describes the changes expected from the policy options being considered, defines associated benefit endpoints and identifies the endpoints that warrant further investigation.

The list of benefit endpoints should be as comprehensive as possible and may be lengthy at first, encompassing all of those that reasonably can be expected to occur regardless of whether they can be quantified and/or put in dollar terms. Analysts should preserve and refine this list as the analysis proceeds. Maintaining the full list of potential effects facilitates later revisions if new information warrants it. Equally important, benefits that can only be characterized qualitatively should be presented along with quantitative information in the benefits analysis (see Chapter 11).

Step 2: Quantify Changes in Significant Benefit Endpoints

Next, the analysis should quantify changes in the benefit endpoints identified in Step 1 as warranting further investigation, focusing on changes attributable to each policy option relative to the baseline. Expertise from a wide array of disciplines in addition to economics is usually needed in this step, including human health and ecological risk assessment, engineering and natural sciences. Quantifying endpoints generally requires a function relating changes in emissions, concentrations and/or exposure to changes in specific ecological services, health effects or risks. Data are usually needed on the magnitude, duration, frequency and severity of the endpoints. For example, changes in cancer risks typically come from human health risk assessments, and the benefits analysis will need information on baseline risks, risk changes associated with each option, the timing of the risk changes, fatality rates and the size and age distribution of affected populations. If visibility is the attribute of concern, needed information includes the geographical areas affected, the baseline visibility and the change in visibility resulting from each policy option.

Sometimes data or modeling constraints will prohibit quantifying significant benefit endpoints. In these cases, it is useful to quantify changes in environmental stressors or measures of environmental quality that would lead to benefits. These changes can be informative in the overall characterization of benefits even if they cannot be aggregated with benefit endpoints.

Analysts should consider the following recommendations when quantifying changes in benefit endpoints.

- **Ensure endpoints are appropriate for benefits evaluation.** A principal role of the economist at this stage is to ensure that the endpoints are characterized in ways that are consistent with principles of economic analysis and the specific models used for benefits analysis. They should also be characterized in a manner that avoids double-counting. Focus on the needs of economic analysis is particularly important at the early stages of ecological or human health risk assessments, and it is generally useful for economists to be part of a cross-disciplinary team for planning and scoping these assessments.[8] The ability to monetize or even quantify benefits analysis may be limited if effects are described too broadly, overlap with other benefits categories, cannot be linked to human well-being, or

8 See, for example, the EPA's Framework for Human Health Risk Assessment to Inform Decision Making (U.S. EPA 2014).

are otherwise incompatible with economic analysis. Text Box 7.2 provides a more detailed discussion on integrating risk assessment and economics.

- **Consider how behavior affects benefit endpoints.** One area where economists may lend unique insights at this stage is in assessing how endpoint quantification is affected by behaviors in the baseline and potential behavioral changes from the policy. These behaviors often drive, for example, how and how much individuals are exposed to environmental contaminants. Changes in behavior due to changes in environmental quality (e.g., staying indoors to avoid detrimental effects of air pollution) can be significant, and economists need to ensure they are considered in benefits analysis.
- **Emphasize quantification over qualitative description.** Qualitative descriptions are useful, but benefits endpoints that are not first quantified can generally not be monetized. The result is that these endpoints have an effective weight of zero in the total benefits calculation. Even for highly uncertain benefits zero is not usually the best quantitative weight, and available evidence can often be used to produce some estimate that is more accurate than assuming these effects do not occur (McGartland et al. 2017, OMB 2023).

Step 3: Estimate the Monetary Value of the Endpoints

The next step is to estimate the monetary value to all affected individuals of the quantified benefit endpoints to obtain the total social benefits of each policy option. This starts with identifying valuation estimates for quantified benefit endpoints. Importantly, it may not be sufficient to multiply a change in endpoint by a single value for that endpoint, particularly in the presence of uncertainty or nonlinearities; valuation must be guided by economic theory (Section 7.2). For estimating total benefits, it is typical to use a representative agent approach, where values are calculated for an "average" or representative individual in the relevant population and then multiplied by the number of individuals in that exposed population.[9]

When estimating monetary value of effects, analysts should:

- **Determine which valuation methods are best suited for each endpoint**. When possible, the value estimate should be based on willingness to pay (WTP), but other measures (e.g., cost of illness) may be used when there are no available WTP estimates. Valuation methods are not unique to specific endpoints, and often a given endpoint can be valued through several methods. Table 7.1 shows general benefit categories, examples of specific benefit endpoints and associated valuation methods commonly used. Sometimes time and resources may be available to conduct original research using these methods, but more often the analysis will need to draw upon existing value estimates in a process called benefit transfer. Section 7.3 provides details on valuation methods. Benefit transfer is described in Section 7.4.

9 Though a representative agent approach is often used, models may be available to incorporate heterogeneity. This can be especially useful for distributional analyses.

Text Box 7.2 - Coordinating Economic Analysis and Risk Assessment

Because economists rely on risk assessment outcomes as key inputs into benefits analysis, it is important to coordinate risk assessment and economic valuation. Health and ecological risk assessments are designed to support the setting of standards or to rank the severity of different hazards. However, measures from these assessments can be difficult or impossible to incorporate into benefits analyses for several reasons. First, the measures may not be probabilistic expressions of risk, but instead indicate how exposures compare to reference levels that are not associated with any quantitative level of risk. It may be that the modeled endpoints cannot be directly related to health outcomes or ecological services that can be valued using economic methods. Also, risk assessments sometimes focus on outcomes near the tails of the exposure and/or risk distribution for highly sensitive endpoints, leading to biased benefits estimates if extrapolated to the general population.

As described in the EPA's *Ecological Benefits Assessment Strategic Plan* (U.S. EPA 2006) and *Framework for Human Health Risk Assessment* (U.S. EPA 2014), coordination between economic analysis and risk assessment should begin early in the planning process for any risk assessments, starting with the Planning and Scoping and Problem Formulation stages where a conceptual model is developed specifying key factors for the assessment including specific endpoints to be addressed. The EPA's *Generic Ecological Endpoints for Ecological Risk Assessment* (U.S. EPA 2016) contains specific guidance to assist ecological risk assessors and economists in identifying ecological services that are amenable to economic analysis (U.S. EPA 2016).

Throughout the risk assessment process, 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 the probability that individuals will seek preventive care. Even in cases where the economists' contribution to the risk characterization is not direct, economists and risk assessors should communicate frequently to ensure that economic analyses are complete.

Specifically, risk assessors and economists should strive to:

Identify a set of human health and ecological endpoints that are economically meaningful, linked to human well-being and are monetizable using economic valuation methods. Risk assessors may be required to model more or different outcomes than if they were attempting to capture only the most sensitive endpoint. This also may require risk assessors and economists to convert human health or ecological endpoints or indicators measured in laboratory or epidemiological studies to effects that can be valued in the economic analysis.

Estimate changes in outcome probabilities (human health or ecological) or changes in continuous outcomes (e.g., IQ) as exposure changes, rather than safety assessment measures (e.g., reference doses) when possible. For human health, probabilistic dose-response assessment may be useful for estimating outcome probabilities (WHO 2017, Chiu and Slob 2015).

Work to produce expected or central estimates of risk, rather than bounding estimates as in safety assessments. Any expected bias in the risk estimates should be clearly described.

Attempt to estimate the timing between changes in emissions or exposures and associated changes in health and ecological risks or outcomes. For health outcomes these time lags are referred to as cessation lag (the time between reduced exposure and reduced health risks) or latency (the time between increases in exposure and increased health risks.)

Attempt to characterize the full uncertainty distribution associated with risk estimates. This will contribute to a better understanding of potential regulatory outcomes and will enable economists to include risk assessment uncertainty in a broader uncertainty analysis uncertainty. The EPA's guidance and reference documents on Monte Carlo methods and probabilistic risk assessment, including the EPA's *Policy for Use of Probabilistic Analysis in Risk Assessments* (U.S. EPA 1997e), and the *1997 Guiding Principles for Monte Carlo Analysis* (U.S. EPA 1997d) may be of interest.

- **Identify valuation estimates and how they are to be used.** Valuation estimates available for benefits analysis will not always perfectly match the policy context being considered. Benefit transfer is the exercise of both identifying valuation estimates that sufficiently relate to the policy context and then transferring the results to the policy analysis. It is important that this is done in ways that are consistent with economic reasoning and theory, and it is not always sufficient to simply apply a single, fixed value. Section 7.4 contains information on both general steps for benefit transfer and specific transfer methods to consider.
- **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 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 no directly comparable study, to draw from. In all cases, the benefits analysis should clearly describe the sources of the valuation estimates and provide a qualitative discussion of the reliability of those sources.
- **Avoid double-counting to the extent possible.** Double-counting may arise for at least two reasons. First, different valuation methods often incorporate different subsets of total benefits, so some types of benefits may be counted twice when aggregating across values. Second, endpoints may be defined in ways that overlap. For example, a human health endpoint of avoided "emergency room visits" is likely to overlap with an endpoint of avoided heart attacks, so valuing these endpoints separately and aggregating them would introduce double-counting. It is important to avoid double-counting when possible and to clearly acknowledge any potential overlap when presenting the aggregated results.
- **Characterize uncertainty.** The analysis should include a quantitative uncertainty assessment when possible using sensitivity analysis or other methods. As with other aspects of the analysis, the depth and scope of this assessment should be commensurate with the scale of the benefits analysis. In some cases, it may be sufficient to focus on a few

key parameters.[10] Important considerations for analysis of uncertainty are provided in Chapter 5, and principles for presenting information on uncertainty are in Chapter 11.[11]

The analysis should ultimately present both the aggregate monetized values as well as the value of each specific benefit endpoint. The monetized benefits estimate should be supplemented by displaying benefits that could be quantified but lack valuation estimates, and a characterization of benefits that can only be qualitatively described. When data or modeling limitations prevent quantitative characterization of benefits endpoints, it can be useful to provide quantitative data related to benefits (e.g., changes in stressors or environmental quality). Chapter 11 discusses the presentation of information on benefits. When the policy or regulation under consideration is expected to result in important feedbacks and interactions between various physical and economic endpoints, analysts should consider whether available integrated approaches for analyzing the specific policy are more appropriate than quantifying each specific endpoint in a separate analysis.

7.2 Economic Value and Types of Benefits

Economic valuation is based on the traditional economic theory of human behavior and preferences, centered on the “utility” (or “welfare”) that people realize from consumption of goods and services, both in market and non-market settings. Core to this approach is the principle of consumer sovereignty, in which values used for benefit-cost analysis (BCA) respect the preferences individuals have for these goods and services rather than being based, for example, on the preferences of the analyst or policy maker. Different levels and combinations of goods and services provide different levels of utility for any one person. Also, because people have different preferences, utility derived from sets of goods and services will vary across people.

Economic theory suggests that when goods and services are bought and sold in competitive markets, optimizing consumers maximize their level of utility subject to constraints on their budget by equating the ratio of the marginal utility (the utility afforded by the last unit purchased) of any two goods that a person consumes with the ratio of the prices of those goods. If it were otherwise, that person could reallocate their budget to buy a little more of one good and a little less of the other good to achieve a higher level of utility.[12]

Utility is inherently subjective and cannot be measured directly; however, to assign “value” an operational definition in benefits analysis, it must be expressed in a quantifiable metric. Dollars

10 If the benefits analysis applies statistical relationships to derive a change in health or ecological outcomes in response to a given change in pollutant exposure (e.g., dose-response functions), the standard errors of the relevant coefficients can be used to derive confidence intervals to help characterize uncertainty. If the statistical relationship is characterized by a continuous function, then the confidence interval is proportional to the magnitude of the change in exposure.

11 Uncertainty in benefits may affect valuation. For example, uncertainty about magnitude of a given risk reduction, may affect WTP for that benefit. It may be possible to elicit WTP that reflects uncertainty about risk, however it may be more pragmatic to apply uncertainty, i.e., a probability of the outcome, to WTP elicited under certainty. In principle, the difference between these two approaches is likely to be small (SAB, 2024).

12 Behavioral economics studies situations in which individuals’ behavior is inconsistent with the standard economic model (assuming rational choice). The implications of irrational behavior and inconsistent preferences for welfare analysis are still an emerging area of economics (Just and Just 2016; Shogren and Taylor 2008). Therefore, our discussion of benefits analysis adheres to the standard economics model of rational, utility-maximizing behavior and consistent preferences. Chapters 4 (Section 4.4) and 5 (Section 5.5) provide more discussion of behavioral economics and its implications for environmental policy design.

conveniently allow direct comparison of benefits to costs and summing of benefits across different effects,[13] but this choice for the unit of account has no theoretical significance. Table 7.1 summarizes the types of benefits most often associated with environmental protection policies and provides examples of each benefits types as well as valuation methods often used to monetize the benefits for each type.

The benefits of an environmental improvement are illustrated graphically in Figure 7.2 which shows marginal abatement costs (MACs) and marginal damages (MDs) of emissions. Reducing emissions from e_0 to e_1 produces benefits equal to the shaded area under the marginal damages curve. Many environmental goods and services, such as air quality and biological diversity, are not traded in markets. The challenge of valuing non-market goods that do not have prices is to relate them to one or more market goods that do. This can be done either by determining how the non-market good contributes to the production of one or more market goods (often in combination with other market good inputs), by observing the trade-offs people make between non-market goods and market goods, or by asking people directly about the tradeoffs they are willing to make. Section 7.3 provides a discussion of the various revealed and stated preference valuation methods. Of course, some methods will be more suitable than others in a given scenario for a variety of reasons, and some will be better able to capture certain types of benefits than others.

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.[14] 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. (Freeman et al. 2014).[15]

Economists generally expect that the difference between WTP and WTA will be negligible, provided the values are small relative to household wealth and substitutes are available for the market or

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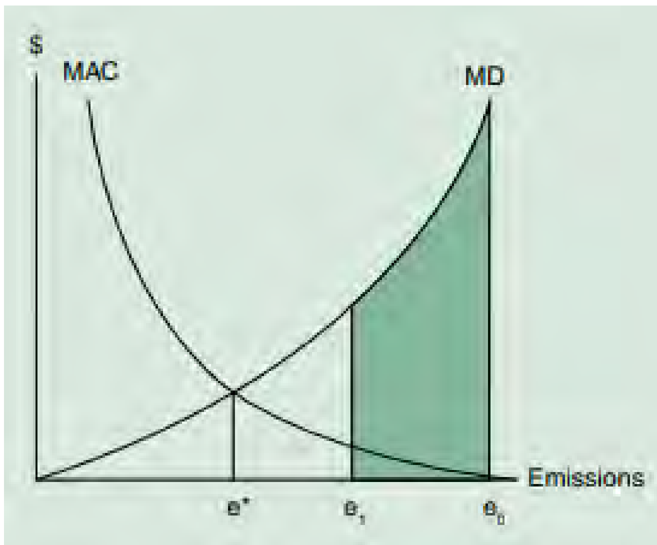
13 Because an individual's utility is unobservable, it cannot be measured in a cardinal sense. However, economists assume that consumers make choices based on whether one bundle of goods is preferred to another, but not necessarily by how much. In other words, consumers respond to changes in prices and income by ordinally ranking consumption bundles using preference relationships. If an individual's preference relationships display nonsatiation (more is better) and substitutability (if one good in a bundle is decreased, it is possible to increase another good to make the consumer indifferent) then they can be represented by an ordinal preference function, or "utility function." While economists cannot observe changes in an individual's utility directly, they can observe income and consumption decisions at various prices that reflect changes in the ordinal utility function and can then compute a money-based measure of these utility changes. This money-based measure is the individual's "willingness to pay" or "willingness to accept" described below. More detail on the development of this utility function can be found in Just et al. (2005) and Freeman et al. (2014).

14 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 et al. (2014), Just et al. (2005), and Appendix A of these Guidelines.

15 See Freeman et al. (2014) for a discussion of how WTP and WTA may be associated with property rights. OMB Circular A-4 also suggests that WTP and WTA are associated with different views of property rights and notes associated issues for benefit transfer.

non-market goods in question (Willig 1976; Hanemann 1991). However, there may be instances in which income and substitution effects are important (such as for some environmental goods) and lead to large disparities between WTP and WTA.[16] Ultimately, economists use the valuation estimates to assess policy outcomes by applying the Kaldor-Hicks compensation test (see Appendix A). In short, the test asks whether hypothetically the gainers from a policy could fully compensate the losers and still be better off — and conversely whether the losers could pay the winners to avoid the change altogether and still be as well off. Since WTP is a consistent measure for this test and 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.

Figure 7.2 - Benefits of an Environmental Improvement



WTP for environmental quality can also be non-linear. For example, Figure 7.2 illustrates a case in which marginal damages increase with emissions. When this occurs, it is important to account for baseline environmental quality when valuing the benefits of incremental improvements. Otherwise, inconsistent results can occur when estimating the benefits from a series of separate actions. In addition, sometimes environmental regulations yield relatively small average changes in health or the environment that may not be noticeable to the public until multiple regulations have achieved a large aggregate improvement. Just as it is important to account for small average costs imposed by regulations — which can be economically significant when aggregated over a sufficiently large population — it is conceptually correct to account for even very small improvements in environmental quality. Chapter 5 provides more discussion of analyzing multiple related rules. Text Box 7.6 in Section 7.4 discusses the issue of estimating multiple improvements in environmental quality using benefit transfer.

16 For more information see Appendix A and Hanemann (1991). Also, Kim et al. 2015, Freeman et al. (2014), and Horowitz and McConnell (2003) discuss and evaluate various explanations for the disparity between WTP and WTA, and other studies have estimated the size of the disparity, e.g., Tuncel and Hammitt (2014), and Kniesner, et al. (2014).

The types of benefits that may arise from environmental policies can be classified in multiple ways (Freeman et al. 2014). As shown in Table 7.1, these *Guidelines* organize benefits into the following categories: human health improvements, ecological improvements and other benefits.

In addition, commonly used valuation methods are provided for reference. The list is not meant to be exhaustive, but rather to provide examples and commonly used methods for estimating values. The sections below provide a more detailed discussion of each of the benefit categories listed in Table 7.1.

7.2.1 Human Health Improvements

Human health improvements from environmental policies include effects such as reduced mortality rates; decreased incidence of non-fatal cancers, chronic conditions and other illnesses; and reduced adverse reproductive or developmental effects. These categories of outcomes are discussed separately below.

Generally, it is good practice to fully characterize both the nature of the risk and the affected populations in benefits analysis, including the age distribution of the affected population. This is helpful not only to evaluate the best approach for valuing health benefits, but to communicate clearly with decision makers about who is affected and how they are affected.

7.2.1.1 Mortality

Some U.S. Environmental Protection Agency (EPA) policies will lead to reductions in human mortality risks due to health conditions such as cancer or cardiovascular disease. In considering the impact of environmental policy on mortality risk, it is important to remember that environmental policies do not protect specific, identifiable individuals from death due to environmental causes. Rather, they generally lead to small reductions in the probability of death for many people.

The value of the mortality risk reductions reflects estimates of individuals' WTP for these small reductions in the risk of dying. When aggregated over the affected population, this value has typically been referred to as the "value of statistical life" (VSL) although other terms have been used (Simon et al. 2019). Regardless of terminology, it is important to recognize that it represents the tradeoff between wealth or income and small changes in mortality risks and is not the value of life itself.

For consistency and added transparency across analyses, EPA policy is to apply a single VSL estimate for the calculation of benefits of mortality risk reductions experienced by all affected populations associated with all EPA programs and policies. Appendix B describes this recommended value, its distribution and derivation, and details its application. To reduce public confusion and misunderstanding, analysts should not use the misleading term "value of life" in Agency analyses as that term does not accurately describe what the VSL represents.

As discussed in Appendix B, analysts should address the impact of risk and population characteristics on the VSL qualitatively. In addition, analysts should account for timing considerations, including:

- The effects of latency: delayed manifestation of health or other effects;
- Cessation lags: time frame between a reduction in exposure to an environmental contaminant and the reduced risk to health; and

- Income growth over time, discounting appropriately where warranted.[17]

Valuing mortality risk changes in children is particularly challenging. The EPA's *Handbook for Valuing Children's Health Risks* (U.S. EPA 2003) provides some information on this topic, including key benefit transfer issues to consider when using adult-based studies. In addition, OMB's Circular A-4 advises:

> "For regulations where health gains are expected among both children and adults (…) 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 2023, p.51).

Reviews of the literature by Gerking and Dickie (2013) and Robinson et al. (2019) provide support for this position.

Methods for Valuing Mortality Risk Changes

Because individuals make risk-wealth trade-offs in different contexts, the value of mortality risk changes can be estimated using a variety of data sources and modeling approaches. The estimate recommended in Appendix B is derived from a combination of hedonic-wage and stated preference studies. In the hedonic wage or wage-risk method, value is inferred from the income-risk trade-offs made by workers for risks faced on the job. Stated preference studies, in which income-risk trade-offs are solicited directly through surveys, are also used to estimate WTP for reduced mortality risks. Key considerations in 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.

There are additional methods that can be used to derive information on risk trade-offs. Averting behavior studies value risk changes by examining purchases of goods that can affect mortality risk (e.g., bicycle helmets). However, isolating the portion of the purchase price associated with mortality risk reductions from other benefits or joint products provided by the good is a challenging hurdle for this literature. Also, of potential importance is short term avoidance behavior — altering one's activities, including the timing and frequency of activities, to reduce exposure.[18] Another approach is to examine trade-offs between types of risks to estimate relative preferences for risk reduction. This approach may make the valuation task more manageable for the respondent but requires multiple steps to obtain a risk-dollar tradeoff.[19]

Important Considerations

The analyst should keep three important considerations in mind when estimating mortality benefits (each described in more detail below):

- Characterizing and measuring mortality effects;

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17 Assumptions about income growth should be consistent throughout the economic analysis. This includes, to the extent feasible, consistency between income growth assumptions and discounting. See Chapter 6 for more information on discounting, generally, and Section 6.5 for consistency between income growth and discounting.

18 See Blomquist (2004) for a review of averting behavior studies and Graff Ziffin and Neidell (2013) for a discussion of short-term averting behavior.

19 See Nielsen, et al. (2019) for an overview and application of risk-risk trade-off method.

- Heterogeneity in risk and population characteristics; and
- The timing of health risk changes.

Characterizing and Measuring Mortality Effects

Although reduced mortality risks associated with an environmental policy are typically small, they are generally aggregated over the affected population and reported in terms of "statistical lives." Suppose, for example, that a policy affects 100,000 people and reduces the risk of premature mortality in the coming year by 1 in 100,000 for each individual. Summing these individual risk reductions across the entire affected population shows that the policy leads to the equivalent of one premature fatality averted, or one statistical life "saved," in the coming year.

An alternative metric seeks to capture the remaining life expectancy, or "quantity of life" saved associated with the risk reductions (Moore and Viscusi 1988) and is typically expressed as "statistical life years." Looking again at the policy described above for reducing risk in the coming year, suppose the risks were spread over a population in which each individual had 20 years of remaining life expectancy. The policy would then "save" 20 statistical life years (1 statistical life x 20 life years). In practice, estimating statistical life years saved requires risk information for specific subpopulations (e.g., age groups or health status). Statistical life years may be used as an outcome measure in cost-effectiveness analysis (Institute of Medicine (IOM) 2006). However, consistent with past Science Advisory Board (SAB) advice, the use of a constant monetized value for a statistical life year is not supported by the literature and is not recommended for benefits analysis (U.S. EPA 2007).

Heterogeneity in Risk and Population Characteristics

The WTP to avoid 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 versus man-made) and the causative event (e.g., cancer or trauma). Population characteristics include those generally expected to influence WTP for any good (e.g., income and education) as well as those more closely related to mortality risks such as baseline risk or remaining life expectancy, health status, risk aversion and familiarity with the type of risk. The empirical and theoretical literature on the effect of many of these characteristics on WTP is incomplete or ambiguous. For example, some studies suggest that older populations are willing to pay less for risk reductions (Viscusi and Aldy 2007); others find this effect to be small if it exists at all (Alberini et al. 2004). Still others suggest older populations have higher WTP (Kniesner, Viscusi, and Ziliak 2006; Smith et al. 2004). Similarly, some studies have found that reductions in fatal cancer risks garner a higher WTP than other kinds of fatal risks (e.g., Viscusi et al. 2014) while others do not find evidence of a "cancer premium" (e.g., Hammitt and Haninger 2010). Appendix B contains a more complete discussion of risk and population characteristics and how they may affect WTP.

Although mortality risk valuation estimates used in economic analyses could reflect these differences in WTP quantitatively with sufficient empirical evidence, Agency policy is to apply a single VSL estimate to all populations and mortality risks and to qualitatively describe population characteristics and risk attributes. One reason for this position is that the empirical evidence in the literature on the relationship between WTP and the various population and risk characteristics is inconclusive. In addition, population characteristics become less relevant for applications of VSL in benefits assessments of national regulations affecting broad spectrums of the population.

Timing of Health Risk Changes

Environmental contamination can cause immediate or delayed health effects. If individuals prefer health improvements earlier in time rather than later, all else equal, then the WTP for reductions in exposure to environmental pollutants will depend on when the resulting health risk changes will occur.[20]

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

If the basis for benefit transfer is a marginal WTP for contemporaneous risk reductions, then calculating the benefits of a policy with delayed risk reductions requires three steps:

1. Estimating the time path of future mortality risk reductions;
2. Estimating the annual WTP for all future years; and
3. Calculating the present value of these annual WTP amounts.

The first step should account for all the factors that ultimately relate changes in exposure to changes in mortality risk as defined by shifts in the survival curve.

7.2.1.2 Morbidity

Morbidity benefits consist of reductions in the risk of non-fatal health effects ranging from mild outcomes, such as headaches and nausea, to very serious illnesses such as cancer (see Table 7.1). Non-fatal health effects also include conditions such as birth defects, low birth weight and reduced cognitive function. Morbidity outcomes need not be so severe to prevent affected individuals from participating in normal activities but are expected to affect quality of life and labor productivity or earnings for workers (Graff Zivin and Neidell 2013, 2018). Availability of existing valuation estimates for morbidity outcomes varies considerably, and the WTP to avoid many health outcomes do not yet exist.

WTP to reduce the risk of experiencing an outcome is the preferred measure of value for morbidity effects. As described in Freeman et al. (2014), this measure consists of four additive components:

- "Averting costs" to reduce the risk of illness;
- "Mitigating costs" for treatments such as medical care and medication;
- Indirect costs such as reduced earnings from paid work, or lost time maintaining a home and pursuing leisure activities; and
- Monetary equivalent of the disutility of illness (e.g., costs of discomfort, anxiety, pain and suffering.)

Methods used to estimate WTP vary in the extent to which they capture these components. For example, cost-of-illness (COI) estimates generally only capture mitigating and indirect costs and omit averting expenditures and lost utility associated with pain and suffering. Consequently, COI

20 The description here focuses on mortality risk, but the same principles apply to non-fatal health risks.

estimates generally understate WTP to reduce a risk or avoid a given health effect. Some studies have estimated that total WTP can be two to four times as large as COI even for minor acute respiratory illnesses (Alberini and Krupnick 2000). Still, no broadly applicable "scaling factor" exists that relates COI to WTP.

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 (e.g., cost of illness) do not measure WTP but can provide useful data; however, those data must be interpreted carefully if they are to inform economically meaningful measures. Methods also differ in the perspective from which values are measured (e.g., before or after the incidence of morbidity), whether they control for the opportunity to mitigate the illness (e.g., before or after taking medication) and the degree to which they account for the four components of total WTP set out above. The three primary methods most often used to value morbidity in an environmental context are stated preference (Section 7.3.2), averting behavior (Section 7.3.1.4) and COI (Section 7.3.1.5). Hedonic methods (Section 7.3.1.3) are used less frequently to value morbidity from environmental causes.

Benefits analysis may also be informed by approaches that do not estimate WTP of reduced morbidity directly. As noted above, risk-risk trade-offs, for example, do not directly estimate dollar values for risk reductions, but rather, provide rankings of relative risks based on consumer preferences. Risk-risk trade-offs can be linked to WTP estimates for related risks.[21]

Other methods for valuing morbidity outcomes suffer from certain methodological limitations and are therefore generally less useful for policy analysis. For example, health-state indices, such as Quality Adjusted Life Years (QALYs) are composite metrics that combine information on quality and quantity of life lived under various scenarios often used for cost-effectiveness (CEA) or cost-utility analyses (CUA) (see Section 7.5.2.1). While appropriate for use in CEA or CUA, these measures are consistent with WTP measures only under very strict conditions that generally do not hold in practice and should ***not*** be used for deriving monetary estimates for use in BCA (Blechrodt and Quiggin 1999; Hammitt 2003; IOM 2006). Another commonly suggested alternative is jury awards; these also generally should ***not*** be used in benefits analysis, for reasons explained in Text Box 7.3.

Important Considerations

Two factors to consider, in addition to the heterogeneity in risk and population characteristics and the timing of health risk changes discussed above, when estimating morbidity benefits are:

- Characterizing and measuring morbidity effects; and
- Third party costs.

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

Characterizing and Measuring Morbidity Effects

Key characteristics that will influence the valuation 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 daily limitations such as “restricted activity,” “bed disability” and “lost work."[22] Severity can also be described using health state indices that combine multiple health dimensions into a single measure.[23] For duration, the primary distinction is between acute effects and chronic effects. Acute effects are discrete episodes usually lasting for a limited time period, 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, such as some gastrointestinal illness are one-time events that are unlikely to recur. Other effects, such as asthma, do recur or can be exacerbated regularly, causing disruptions in work, school or recreational activities.

For chronic conditions or more serious outcomes, morbidity effects are usually measured in terms of the number of expected cases of illness. Given the risks faced by each individual and the number of people exposed to this risk, an estimate of “statistical cases” can be defined analogously to “statistical lives.” In contrast, morbidity effects that are considered acute or mild in nature can be estimated as the expected number of times a particular symptom associated with an illness occurs within an affected population over a period of time (e.g., annually), where individual members of the population may experience the effect more than once. These estimates of “symptom days” may be used in benefits analysis when appropriate estimates of economic value are available, although a richer characterization of combinations of symptoms, severity, duration and episode frequency would be an improvement over much of the existing literature. Some studies have attempted to address these complexities in a more systematic manner (Cameron and DeShazo 2013).

Third Party Costs

The widespread availability of health insurance and paid sick leave shifts some of the costs of illness from individuals to others. While this cost-shifting can be addressed explicitly in COI studies, it may lead to problems in estimating total WTP especially in stated preference studies. If the researcher does not adequately address these concerns, individuals may mis-state their WTP, assuming some related costs will be borne by others. Some stated preference studies are designed to avoid capturing third party or insurance costs in which case the results would be additive to COI. Regardless, to the extent third party costs represent diversions from other uses in the economy, they represent real costs to society and should be accounted for in the analysis.

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22 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.

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

Text Box 7.3 - Non-Willingness to Pay Measures

As described earlier, WTP is a valid measure of economic value because it can be used in potential compensation tests of Kaldor and Hicks. Sometimes, however, other measures are suggested for use in benefits analysis that are not appropriate. Measures of economic value that do not measure WTP and cannot be related to changes in utility are not valid for use in benefits analysis. Three common examples of such values are replacement cost, proxy cost and jury awards.

Replacement cost. A common consequence of environmental deterioration is damage to assets. Some analysts suggest that the economic value of the damage is the cost of replacing the asset. In the context of BCA, this is not generally true. It is only true if: (1) damage to the asset is the only cost incurred; and (2) the least expensive way to achieve the level of satisfaction realized before the deterioration would be to replace the asset (Freeman, Herriges, and Kling 2014). If the first condition is not met, consideration of replacement costs may be useful but should be combined with assessments of other costs. If the second condition is not met, replacement costs might overestimate the consequences. Suppose that water pollution kills fish in a pond. Replacing those fish with healthy, edible ones might prove extremely expensive; the pond might need to be dredged and restocked. However, people who are no longer able to catch fish in the pond might be compensated simply by giving them enough money to purchase substitutes in the market.

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

Jury awards. Attempts are sometimes made to value environmental improvements using jury awards. Using jury awards in this way may prove problematic for several reasons. First, cases only go to trial if both sides prefer the expected value of an adjudicatory outcome to the certainty of a pre-trial settlement. Cases that go to juries are "atypical" by definition. Second, since adjudication does not always occur and can never be infallible, jury awards often do, and arguably should (Shavell 1979), embody "punitive" as well as "compensatory" elements. Juries make examples of guilty defendants to try to deter others from committing similar offenses. For this reason, jury awards may overstate typical damages. Finally, jury awards are ex-post measures based on a known outcome, not the probability of experiencing an adverse event. These estimates are not appropriate for application to ex-ante evaluation of the value associated with a statistical probability.

7.2.2 Ecological Benefits

Many EPA policies will produce ecological benefits by enhancing the delivery of ecosystem services, defined here as "direct or indirect contributions that ecosystems make to the well-being of human populations" (Thompson et al. 2009). Examples of EPA policies affecting ecosystem services include: reducing acid precipitation that may acidify forests and freshwater ecosystems; controlling pesticides and other environmental contaminants that affect pollinators such as bees, as well as predators of pests; reducing nutrient pollution from municipal wastewater treatment plants, septic systems, fertilizer and manure runoff, and atmospheric deposition that may lead to changes in the composition and attributes of receiving water bodies.

In each of these examples, environmental regulation may not directly affect goods or services in the household utility function. Instead, they affect ecological inputs into the processes that generate such goods and services. The valuation of ecosystem services is not fundamentally different than the valuation of other productive assets on which our economy depends (Polasky 2008; Barbier 2012, OMB 2024). The relevant endpoints are goods or services that enter the household utility function directly. Ecosystem services that contribute to the production of those assets, but are not directly valued by households, should be recognized as inputs in an ecological production function (EPF) and monetization should value their marginal product. Making the distinction between final ecosystem services and ecological inputs and identifying the relevant linkages is a challenging task facing analysts of environmental policy (Boyd and Banzhaf 2007).

7.2.2.1 Ecological Production Functions

An ecological production function is a description of how ecosystems combine inputs to produce ecosystem services that consumers enjoy directly or are used in the production of goods or services that are enjoyed by consumers. The natural science literature provides guidance for some cases on the form of the ecological production function (MacArthur and Wilson 1967; Kingsland 1985) and numerous examples (Hamel et al. 2015; Reddy et al 2015; Kremen et al. 2007; Jaramillo et al. 2010).

Knowledge of the relevant ecological linkages is essential to predicting the effects of environmental policies on ecosystem service provision and an economic analyst will likely benefit from collaborating with ecologists or other natural scientists to predict the effects of the proposed policy. The Agency's *Ecological Benefits Assessment Strategic Plan* describes an interdisciplinary approach for conducting ecological benefits assessments (U.S. EPA 2006). To familiarize themselves with "benefit relevant indicators" of ecological endpoints that may be affected by policy measures, analysts may also wish to consult National Ecosystem Services Partnership (2016) or National Ecosystem Services Classification System (U.S. EPA 2015a).

There are several sources an analyst might consult for potentially useful ecological production functions. In addition to searching the scientific literature on the topic of interest, some large-scale research ventures maintain suites of models of pollination, storm protection, pollution treatment, groundwater recharge and other phenomena. The Natural Capital Project, for example, maintains 19 models of Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST). Another suite of such models is the Artificial Intelligence for Ecosystem Services platform (ARIES). Bagstad et al. (2012) review and compare InVEST and a number of other models that might be used to model the generation of ecosystem services. In addition, the EPA's Office of Research and Development has compiled lists of potentially useful models in its Ecoservice Models Library (ESML 2024).

Using an "off-the-shelf" ecological production function in benefits analysis may have the disadvantage of not being tailored to the specific application, but the advantages could include:

- Some EPFs have been extensively peer-reviewed and found to be of practical value;
- Using an existing EPF may save considerable time and effort;
- In some instances, it may be possible to calibrate the parameters of an ecological production function, even if the parameters are unknown, using readily available data or summary statistics for the policy case (e.g., Massey et al. 2017);
- Alternatively, it may be possible to develop useful bounding results that hold regardless of particular parameter values (e.g., Simpson et al*.* 1996; Simpson 2016).

Even when using established and tested ecological production functions, there is no substitute for substantive familiarity with the subject matter. The analyst should understand the logic behind the form of the ecological production function and consult with an expert on the subject (often a biologist or other natural scientist) before deciding to adopt one in the work. Another consideration is that arguments of the production function should be relevant to the problem the analyst is addressing. Many ecosystem service models relate habitat area to the provision of a service; this may not inform questions of how pollutants affect the provision of the service.

Ecological production functions sometimes exhibit what may seem to be counterintuitive effects. Consider as an example, nutrients (primarily reactive nitrogen and phosphorus) from municipal, agricultural or other sources that may enter water bodies. The term “eutrophication” describes the consequences of *excessive* delivery of nutrients. Marine biologists have documented that, in some circumstances, increased nutrient availability may enhance some desirable endpoints, such as the support of larger populations of fish caught by commercial and recreational fishers (e.g., Breitburg et al. 2009). In another example, bees that carry pollen between orange groves are a necessary component of orange productions, however, carrying pollen between different groves may at times hybridize fruits resulting in a lower value crop (Sagoff 2011). These examples are unusual, but they underscore a point: preserving systems in, or restoring them to, a more “natural” state may not always enhance the value of the services they provide.

7.2.2.2 Estimating Benefits Using Ecological Production Functions

Knowing *only* the ecological production function is generally insufficient to conduct economic valuation. While ecological production functions are analogous to production functions that are a staple of textbook microeconomics, they often differ in one important respect: the inputs and outputs of ecological production functions are often not traded in markets (U.S. EPA 2009b). Consequently, rather than being able to observe prices, we must infer them using the tools of nonmarket valuation. Massey et al. (2006), Newbold and Massey (2010), Smith and Crowder (2011), and Finnoff and Tschirhart (2011) exemplify how these linkages can be made for commercial fisheries and recreational anglers.

Monetizing the ecological benefits of environmental regulations using ecological production functions proceeds in three phases (Bateman, 2012). The first is to project changes in the ecological inputs caused by the regulation. This phase may require its own extensive modeling effort such as hydrological models that predict the effect of land use changes on nutrient and sediment loadings to lakes, rivers and streams. The second phase employs the ecological production function to project how the changes in those inputs affect the provision of final ecosystem services. To use the nutrient and sediment pollution example again, this would require a model of aquatic ecosystems to project changes in environmental goods that people value such as fish to catch and water amenities like clarity and odor. Finally, changes in final ecosystem services are valued using nonmarket valuation methods. Revealed and stated preference approaches to nonmarket valuation are described in detail in Sections 7.3.1 and 7.3.2. When resource constraints prevent an original nonmarket valuation study, benefit transfer can be used to apply values estimated in other contexts; See section 7.4 for a detailed discussion with caveats.

7.2.2.3 Benefits Estimation When the Ecological Production Function is Not Known

When ecological production functions are not known, it may be easier and/or more defensible to infer ecosystem service values from other relationships. If only the changes in the ecological inputs are known, these can be used in revealed preference approaches to valuation by observing their impact on complementary market behaviors.

Fundamental results in economics establish that these production relationships may be equivalently expressed as profit functions and that profits may be capitalized into the value of assets such as advantageously located property. As such, hedonic valuation methods are frequently proposed for ecosystem service valuation (see, for example, Swinton et al. 2007; Bishop and Timmins 2018). Several researchers have conducted hedonic property value studies to estimate values of assets such as forest cover (Kim and Johnson 2002; Tyrvainen and Miettinen 2000; Mansfield et al. 2005; Sander et al. 2010), wetlands (Tapsuwan et al. 2009; Mahan et al. 2000; Woodward and Wui 2001; Bin and Polasky 2005), or other varieties of "open space" (Sander and Polasky 2009; Cho et al. 2006; Irwin and Bockstael 2002; Irwin 2002; Thorsnes 2002).

The estimation of recreational demand, or, more generally, locational choice models (e.g., Kuminoff et al. 2013) are based on similar underlying principles: choices of where to visit or live are made to maximize utility (or profits) and the ecological attributes of an area affect such choices (McConnell 1990; Parsons 1991; Phaneuf et al. 2008). Hedonic price, recreational demand or locational choice models may be regarded as "reduced form" representations of ecological production from which the analyst can infer the values individuals ascribe to ecosystem services by observing the choices they make, provided that the analyst can adequately control for potentially confounding factors. These approaches are discussed further in Sections 7.3.1.2 and 7.3.1.3.

7.2.3 Other Benefits

Other types of potential benefits from environmental policies include aesthetic improvements, reduced material damages and other benefits resulting from changes that occur in response to a regulation.

Aesthetic improvements include effects such as the improved taste and odor of tap water resulting from water treatment requirements and enhanced visibility resulting from reduced air pollution. Increased visibility due to improved air quality can be divided into two types of benefits: residential visibility benefits and recreational visibility benefits. Improvements in residential visibility are generally assumed to only benefit residents living in the areas in which the improvements are occurring, while all households in the United States are usually assumed to derive some benefit from improvements in visibility in areas such as national parks. The benefits received, however, may decrease with the distance from the recreational area in which the improvements occur.[24]

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

24 Boyle et al (2016) estimate WTP for shifts in the distribution of visibility in national parks.

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.

Some positive welfare effects resulting from regulation do not fit into the previous categories. Specific examples include lower consumer expenditures on fuel or electricity from regulations that improve vehicle fuel economy or appliance energy efficiency, or reduced infrastructure expenditures from regulations that encourage green infrastructure for stormwater management. Whether these effects are presented as cost savings or benefits is not important for the calculation of net benefits. Section 5.5.2 discusses issues for analysts to consider when they estimate that a regulation that strengthens environmental protection results in net private cost savings, which would not typically be expected.

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

When other benefits result from changes in market goods, then demand analysis of the affected market can be a useful approach. Non-market valuation approaches such as those discussed in the remainder of this chapter may be required to measure welfare effects from changes in non-market goods.

7.3 Economic Valuation Methods for Benefits Analysis

For goods bought and sold in undistorted markets, the market price indicates the marginal social value of an extra unit of the good. Often, there are no markets for environmental goods. While some natural products are sold in private markets, such as timber and fish, the analyst's concern will typically be with *non-market* inputs, which are, by definition, not traded in markets.[25] To overcome this lack of market data, economists have developed a number of methods to value environmental quality changes. Most of these methods can be broadly categorized as either revealed preference or stated preference methods.

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

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

As noted in Chapter 8, although there is an expanding body of work that uses computable general equilibrium (CGE) models to evaluate nonmarket goods (Smith et al. 2004; Carbone and Smith 2008), CGE models often lack a credible way to represent environmental externalities or the benefits that accrue to society from mitigating them. As such, a CGE model's economic welfare measure is typically incomplete and not a suitable means at present to capture the benefits of a regulation.

As a general matter, revealed preference methods have the advantage that they are based on actual tradeoffs and decisions made by individuals. Stated preference methods sometimes have the advantage that the choice question can be tailored to obtain values that more closely align with the needs of benefits analysis. Each of these revealed and stated preference methods is discussed in detail below, starting with an overview of the 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

25 There are examples in which environmental goods have been traded in markets. The Clean Air Act Amendments of 1990, for example, initiated a market in sulfur dioxide (SO_2). Similarly, studies in the computational economics literature have performed ex-ante calibrated analyses by constructing hypothetical markets for environmental goods. The non-market good valuation is determined through shadow prices in imposed regulation-induced scarcity in the environmental good. However, prices in such markets are determined by ex-ante policy driven quantity constraints, and not through empirically based statistical methods by considerations of marginal utilities or marginal products.

Guidelines were written. Analysts should work with the National Center for Environmental Economics (NCEE) to determine the usefulness of additional resources as they become available. In practice, analyses will often need to draw upon values from multiple methods to value benefits. Text Box 7.4 briefly describes original valuation studies using multiple methods conducted by the Agency to estimate benefits of improved water quality in the Chesapeake Bay using many of the methods discussed here.

7.3.1 Revealed Preference Methods and the Cost of Illness

- A variety of revealed preference methods for valuing environmental changes have been developed and are widely used by economists. While these methods all use observable data to estimate or infer value, they each have their own set of advantages and limitations. The following common types of revealed preference methods are discussed in this section:
- Production or cost functions;
- Travel cost models;
- Hedonic models; and
- Averting behavior models.

This section also discusses the cost of illness (COI) approach to valuation. It is worth noting that estimation approaches can span more than one method. For example, the random utility maximization framework in discrete choice models is commonly applied to travel cost models.[26]

7.3.1.1 Production and Cost Functions

Discrete changes in environmental circumstances generally cause both consumer and producer effects, and it is common practice to separate the welfare effects brought about by changes in environmental circumstances into consumer surplus and producer surplus.[27] Marginal changes can 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 product of the input by the marginal utility obtained from the consumption good, in which the input is employed in production. The marginal value of a consumption good is recorded in its price. While marginal products are rarely observed, the need to observe them is obviated when both inputs and outputs are sold in private markets because *prices* can be observed.[28] Environmental goods and services are typically not traded in private markets, and therefore the values of environmental inputs must be estimated indirectly.

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26 For an example of random utility maximization (RUM) and how it is applied to a travel cost approach, please see the travel cost applications later in this revealed preference section. RUM is also commonly applied in stated preference methods and its various estimation approaches.

27 See Appendix A for more detail.

28 The suitability of prices for welfare analysis depends on the structure of the market — this is discussed in the section titled, "Considerations In evaluating and understanding production and cost functions," within Section 7.3.1.1.

Text Box 7.4 - Benefits Analysis of the Chesapeake Bay TMDL

In 2010, the EPA established the Chesapeake Bay (CB) total maximum daily load (TMDL), a comprehensive “pollution diet” to restore clean water to the Bay and area streams, creeks and rivers. The EPA’s National Center for Environmental Economics (NCEE) was tasked with assessing the TMDL's benefits in a multi-faceted analysis of recreational and aesthetic amenities. NCEE began by conducting a scoping exercise to inform which detailed benefits analyses to pursue. Categories yielding small benefits in previous analyses were shelved (U.S. EPA 2002, 2009b). The EPA’s CB Program Office provided data on water clarity and pollutant loadings with and without the TMDL. NCEE engaged external experts on CB fisheries and water quality to obtain their best professional judgments of potential stock size changes relative to current water quality conditions, holding all other influences constant. NCEE used an extension of the U.S. Geological Service's SPARROW (SPAtially Referenced Regression on Watershed attributes) model to predict nutrient loadings and chlorophyll in lakes (Moore et al. 2011). Summaries appear below. Estimates are not additive across studies; overlaps may exist among homeowners, recreators and respondents.

Hedonic property value analysis: Walsh et al. (2017) used spatially explicit water quality data paired with economic, geographic and demographic variables to analyze the value of water clarity to home buyers using over 200,000 property sales in Maryland. Klemick et al. (2018) then used meta-analysis to synthesize the value of clarity improvements in Maryland and to transfer the results to properties in Delaware, Virginia and the District of Columbia. Together, they found that predicted water clarity improvements from the TMDL would result in a 0.7-1.3% increase in property value for waterfront homes. Properties farther from the water had smaller effects. Total near-waterfront property values could increase by about $458 to $802 million from water clarity improvements, which is equivalent to an annualized value of $14 to $56 million at discount rates of 3 and 7%.

Market analysis: Like many fresh goods, fish and shellfish are highly perishable; producers cannot easily adjust supply in the short run to respond to changes in demand. Moore and Griffiths (2018) developed a two-stage inverse demand model to describe how prices respond to supply changes in other commodity groups. The model allowed NCEE to estimate consumer welfare impacts of an increase in CB fish and shellfish harvests while allowing other areas’ harvests to act as substitutes. The estimated annual value of expected harvest improvements is $14.2 million.

Fishing model: NCEE estimated benefits to recreational anglers using a linked participation and site-choice recreation demand model. The model relied on historic catch rate data from the Marine Recreational Fisheries Statistics Survey, an intercept survey that uses weights based on historic visitation frequencies at each intercept site. The data were used to estimate a random utility site-choice model and trip counts from respondent zip codes were used to estimate a participation model conditional on the inclusive value of all sites as estimated by the site-choice model. The resulting estimates of recreational fishing benefits range between $5.7 and $67.6 million per year.

Other recreation demand: NCEE used a recreation demand model to estimate the benefits from other outdoor recreation activities using data on total visitor counts to national and state parks in Maryland, Virginia and Delaware, supplemented with survey data on the number of recreation trips taken to the CB area. The marginal effects of water quality on recreators’ site choices were estimated in a second-stage regression, using estimates of site-specific constants from the first-stage site-choice model as the dependent variable and measures of average water quality conditions and other fixed site attributes as explanatory variables. The estimated annual outdoor recreation benefits (exclusive of recreational fishing) range from $120 to $321 million.

Stated preference survey: Moore, et al. (2018) conducted an SP survey linking forecasted water quality changes to ecological endpoints to estimate use and nonuse values for aesthetic and ecological improvements in the CB and watershed lakes. The survey estimated WTP for improvements in water clarity; populations of three CB species (striped bass, blue crab, and oysters); and the condition of freshwater lakes in the CB Watershed. They found that benefits to watershed lakes and nonuse values account for a large proportion of total WTP and would significantly affect the benefit-cost ratio of pollution reduction programs. Estimated benefits from the projected environmental improvements after the TMDL range from $4.47 billion to $7.79 billion per year.

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.[29]

The value of a marginal change in some environmental condition can be represented as a marginal change in the value of production, as a marginal change in the cost of production, or as a marginal change in the profitability of production.[30] It should be noted, however, that problems of data availability and reliability often arise. Such problems may motivate the choice among these conceptually equivalent approaches, or favor of another approach.

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

In the statements above, note the emphasis that *marginal* effects are being estimated. Estimating the net benefits of larger, non-marginal changes represent a greater challenge to the analyst. In general, this requires consideration of changes in both producer and consumer surplus.

Links Between Production and Hedonic and Other Models

A fourth way to estimate environmental effects on production possibilities is through the profitability of enterprises engaged in production. The value of a fixed asset, such as a parcel of land, is related to the stream of earnings that can be achieved by employing it in its most profitable use. Its rental value is, therefore, equal to the profits that can be earned from it over the period of use. The purchase price of the land parcel is equal to the expected discounted present value of the

29 Varian (1992) describes the relationships among these functions.

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

stream of earnings that can be realized from its use over time. Therefore, the production, cost and profit function approaches described above are also equivalent to inferences drawn from the effects of environmental conditions on asset values. This fourth approach is known as "hedonic pricing," and will be discussed in detail in

Section 7.3.1.3. It is introduced now to show that production, cost, or profit function approaches are generally equivalent to hedonic approaches.

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

General Application of Production and Cost Functions

Empirical applications of production and cost function approaches are diverse. Among other topics, the empirical literature has addressed the effects of air quality changes on agriculture and commercial timber industries. It also has assessed the effects of water quality changes on water supply treatment costs and on the production costs of industry processors, irrigation operations and commercial fisheries.[31] Production, cost or profit functions have found interesting applications to the estimation of some ecological benefits. Probabilistic models of new product discovery from among diverse collections of natural organisms can also be regarded as a type of "production." Finally, work in ecology points to "productive" relationships among natural systems that may yield insights to economists as well.

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. Economists have typically preferred to estimate cost or profit functions. Data on prices are often more complete than are data on quantities, and prices are typically uncorrelated with unobserved conditions of production, whereas input quantities are not.

The model for estimation. Standard practice involves the estimation of more flexible functional forms (i.e., functions that can be regarded as second-order approximations to any production

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31 See, for example, Price and Heberling (2018) and the studies reviewed therein on source water quality.

technology). The translog and generalized Leontief specifications are examples.[32] Estimation often will be more efficient if a system of equations is estimated (e.g., simultaneous estimation of a cost function and its associated factor demand equations), although data limitations may impose constraints.

Market imperfections. Analysts should consider the impact of market imperfections and tax distortions. Most analyses assume perfectly competitive behavior on the part of producers and input suppliers and assume an absence of other distortions. When these assumptions do not hold, the interpretation of welfare results becomes more challenging. While there is an extensive literature on the regulation of externalities under imperfect competition, originating with Buchanan (1969), analysts should exercise caution and restraint in attempting to correct for departures from competitive behavior. The issues can become quite complex and there is typically no direct evidence of the magnitude of the departures. In many circumstances it might reasonably be argued that departures from perfect competition are not of much practical concern (Oates and Strassman 1984).

Perhaps a more pressing concern in many instances will be the wedge between private and social welfare consequences that arise with taxation. An increase in the value of production occasioned by environmental improvement typically will be split between private producers and the general public through tax collection. The issues here also can become quite complex (see Parry et al. 1997), with interactions among taxes leading to sometimes surprising implications. While it is difficult to give general advice, analysts may wish to alert policy makers to the possibility that the benefits of environmental improvements in production may accrue to different constituencies.

7.3.1.2 Travel Costs

Recreational values associated with an environmental improvement constitute a potentially large class of use benefits (see Table 7.1 for examples). However, measuring these values is complicated by the fact that the full benefits of recreation activities are rarely reflected in the price to access them. 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 the cost of visiting sites. A heuristic example is choosing between visiting a nearby recreation location with low environmental quality versus a more distant location with higher environmental quality. The outcome of the decision of whether to incur the additional travel cost to visit the location with higher environmental quality reveals information about the recreator's WTP for environmental quality. Travel cost models are often referred to as recreation demand models because they are most often used to value the availability or quality of recreational opportunities.

Economic Foundation of Travel Cost Models

Travel cost models of recreation demand focus on the choice of the number of trips to a given site or set of sites that a traveler makes for recreational purposes. Because there is generally no explicit market or price for recreation trips, travel cost models rely on the assumption that the "price" of a recreational trip is equal to the cost of visiting the site. These costs include both participants' monetary cost and opportunity cost of time. Monetary costs include all travel expenses. For example, when modeling day trips taken primarily in private automobiles, travel expenses would

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32 See Coelli, et al. (2005) for more details on the properties and estimation of a range of production functions.

include roundtrip travel distance in miles multiplied by an estimate of the average cost per mile of operating a vehicle, plus any tolls, parking and admission fees.

A participant's opportunity cost of time for a recreational day trip is the value of the participant's time spent traveling to and from the recreation site plus the time spent recreating since it is time that could have been devoted to other activities. In most cases, onsite recreational time is assumed to be constant across a recreator's choice alternatives and is, therefore, not included in the estimate of travel costs.[33] Although estimates of the opportunity cost of time ranging from zero to more than 100% of wage rates can be found in the literature, time spent traveling for recreational purposes is generally valued at some fraction of an individual's full wage rate.[34] The fraction of the wage rate used is important because it directly affects estimates of willingness to pay. As the fraction of the wage rate assumed to represent the opportunity cost of time rises, it causes total travel cost estimates to rise, which in turn cause estimates of willingness to pay to also rise.

Most commonly in the recreation demand literature, researchers have used one-third of a person's annual hourly wage as an estimate of participants' hourly opportunity cost of time, although estimates of two-thirds of the full wage rate can also be found (Parsons 2003b; English, Leggett, and McConnell 2015; Phaneuf and Requate 2017). Within that range, the U.S. Department of Transportation (DOT) guidance recommends valuing recreational travel at 50% of the hourly median household income for local travel and 70% for intercity travel (U.S. DOT 2016). A number of researchers have also developed methods for estimating recreators' opportunity cost of time endogenously, although no one method has yet been fully embraced in the literature.[35] Unless compelling reasons for deviating from the standard wage rate assumptions are present, analysts should generally rely on the standard one-third of the wage rate opportunity cost assumption when estimating recreation travel in original studies. Conducting analyses using one-half of the wage rate can also be justifiable in some cases when done in addition to the one third assumption as a way to check the sensitivity of estimates to opportunity cost assumptions.

Even among studies that use the same fraction of the wage rate to estimate the opportunity cost of time, care must still be taken in comparing estimates across studies. First, researchers in the literature vary in their use of personal or household income in calculating opportunity costs. Household income tends to be greater on average than personal income resulting in larger opportunity cost estimates. Second, when researchers do not have recreators' self-reported incomes they have often used population median or average income levels. Average income is generally higher than median income because higher incomes in the tail of the distribution tend to pull the average up. Lastly, in cases where household income is used, opportunity cost estimates will depend on whether costs are assumed to accrue to adults and children or only to adults. The literature is not clear on the preferred choices specification of opportunity costs, so the analyst must use best professional judgement to decide what is best on a case-by-case basis.

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33 If onsite time is assumed to be an additional choice variable, then estimation will require a model that accounts for the decision of how long to recreate at a site. Examples of models investigating onsite time include Bell and Leeworthy (1990); McConnell (1992); Larson (1993); Berman and Kim (1999); and Landry and McConnell (2007).

34 For an in-depth discussion of the valuation of time in different contexts, see U.S. EPA (2020).

35 For examples, see McConnell and Strand (1981); Smith, Desvousges, and McGivney (1983); Bockstael et al. (1987); McConnell (1992); McKean et al. (1995), Feather and Shaw (1999), Palmquist et al. (2010), Fezzi et al. (2014); and Larson and Lew (2014).

Hourly opportunity costs are multiplied by round trip travel time and time on-site to calculate a person's full opportunity cost of time. Total travel costs are the sum of monetary travel costs and full opportunity costs. Following the law of demand, as the cost of a trip increases the quantity of trips demanded generally falls, all else equal. This means that participants are more likely to visit a closer site than a site farther away.

While travel costs are the driving force of the model, they do not completely determine a participant's choice of sites to visit. Site characteristics, such as parking, restrooms or boat ramps; participant characteristics, such as age, income, experience and work status; and environmental quality also can affect demand for sites. Changes in the measures of environmental quality are generally the focus of economic analyses done in support of the regulatory decision-making process. The identification and specification of the appropriate site and participant characteristics are generally determined by a combination of data availability, statistical tests and the researcher's best judgment. Ultimately, every recreation demand study strikes a compromise in defining sites and choice sets, balancing data needs and availability, costs and time.[36]

General Application by Type of Travel Cost Model

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

Single-site models. Single-site travel cost models examine recreators' choices 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. The price of close substitute sites could also affect demand. Income and other participant characteristics act as demand curve shifters. For example, avid outdoor recreators (fishermen or birders for example) may be more likely to take more trips than non-avid recreators, all else equal. Most current single-site travel cost models are estimated using count data models because the dependent variable (number of trips taken to a site) is a non-negative integer. See Haab and McConnell (2003) and Parsons (2003a) for detailed discussions and examples of recreation demand count data models.

Single-site models are most commonly used to estimate the value of a change in access to a site, particularly site closures (e.g., the closure of a lake due to unhealthy water quality). The lost access value due to a site closure is given by the area under the demand curve between a participant's current price and the price at which trip demand falls to zero.[37] Estimating the value of a change in the cost of a site visit, for example the addition or increase of an admission fee, is another common application of the model. Although it is possible with alternative data and model structures, single site models are not generally used for valuing changes in site quality.

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

36 For a comprehensive treatment of the theoretical and econometric properties of recreation demand models, see Phaneuf and Smith (2005). Best practices are discussed in Lupi et al. 2020.

37 The price at which trip demand falls to zero is commonly called the choke price.

researchers have opted to refrain from using single-site models when examining situations with large numbers of substitute sites.[38]

Multiple-site models. Multiple-site models examine a recreator's choice of which site to visit from a set of available sites (known as the choice set) on a given choice occasion and, in some cases, can also examine how many trips to make to each specific site over a fixed period of time. Compared to the single-site model, the strength of multiple-site models lies in their ability to account for the availability and characteristics of substitute sites. By examining how recreators trade the differing levels of each site characteristic and travel costs when choosing among sites, it is possible to place a per trip (or choice occasion) dollar value on site attributes or site availability for single sites or multiple sites simultaneously.

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

Random utility maximization models.[39] In a RUM model each alternative in the recreator's choice set is assumed to provide the recreator with a given level of utility, and on any given choice occasion the recreator is assumed to choose the alternative that provides the highest level of utility on that choice occasion.[40] The attributes of each of the available alternatives, such as the amenities, environmental quality and the travel costs, are assumed to affect the utility of choosing each alternative. Because people generally do not choose to recreate at every opportunity, a non-participation option is often included as a potential alternative.[41] From the researcher's perspective, the observable components of utility enter the recreator's assumed utility function. The unobservable portions of utility are captured by an error term whose assumed distribution gives rise to different model structures. Assuming that error terms have a type 1 extreme value distribution leads to the closed-form logit probability expression and allows for maximum likelihood estimation of utility function parameters. Using these estimated parameters, it is then possible to estimate WTP for a given change in sites' quality or availability.

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38 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. See Bockstael, McConnell, and Strand (1991) and Shonkwiler (1999) for more discussion and other examples of extensions of the single-site model.

39 English et al. (2018) is a highly scrutinized RUM model conducted for the damage assessment following the Deepwater Horizon spill and demonstrates a number of sensitivity analyses. Additionally, the public archive for the case contains a wealth of information. See www.doi.gov/deepwaterhorizon/adminrecord under the heading "5.10 Lost Human Use;" see Section 5.10.4 for technical reports discussing issues surrounding RUM estimation.

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

41 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. In other words, 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.

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

The nested logit and mixed logit models are extensions of the basic logit model. The nested logit model groups similar alternatives into nests where alternatives within a nest are more similar to each other than they are to alternatives outside of the nest. In very general terms, recreators are first assumed to choose a nest and then, conditional on the choice of nest, they then choose an alternative within that nest. Nesting similar alternatives allows for more realistic substitution patterns among sites than is possible with a basic logit. The mixed logit is a random parameter logit model that allows for even more flexible substitution patterns by estimating the variation in preferences (or correlation in errors) across the sample. If preferences do not vary across the sample, then the mixed logit collapses to a basic logit.[42]

The Kuhn-Tucker (KT) model. The KT model is a seasonal demand model that estimates recreators' *choice of which sites to visit (like a multiple-site model) and how often to visit them over a season (like a single-site model).* The model is built on the theory that people maximize their seasonal utility subject to their budget constraint by purchasing the quantities of recreation and other goods that give them the greatest overall utility. Similar to the RUM model, the researcher begins by specifying the recreator's utility function. Taking the derivative of this utility function with respect to the number of trips taken, subject to a budget and non-negative trip constraint, yields the "Kuhn-Tucker" conditions. The KT conditions show that trips will be purchased up to the point that the marginal rate of substitution between trips and other spending is equal to the ratio of their prices. In cases where the price of a trip exceeds its marginal value none will be purchased. Given assumptions on the form of the utility function and the distribution of the error term, probability expressions can be derived, and parameter estimates may then be recovered. While recent applications have shown that the KT model can accommodate a large number of substitute sites (e.g., von Haefen et al. 2004), the model is computationally intensive compared to RUM models.[43]

Considerations in Evaluating and Understanding Recreation Demand Studies

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

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42 See Train (1998) and Train (2009) for detailed descriptions of the nested and mixed logit models.

43 For a basic application of the KT model see Phaneuf and Siderelis (2003). For more advanced treatments of the models see Phaneuf, Kling, and Herriges (2000), and von Haefen and Phaneuf (2005).

In addition to the definition of what constitutes a site, the number of sites included in a recreator's choice set can have a significant effect on estimated values. When defining choice sets, the most common practice in the literature has been to include all possible alternatives available to the recreator. In many cases availability has been defined by location within a given distance or travel time.[44] This strategy has been criticized on the grounds that people may not know about all possible sites, or even if they do know they exist they may not seriously consider them as alternatives. In response to this, a number of researchers have suggested methods that either restrict choice sets to include only those sites that the recreators seriously consider visiting (e.g., Peters et al. 1995, and Haab and Hicks 1997) or that weight seriously-considered alternatives more heavily than less-seriously-considered alternatives (e.g., Parsons, Massey, and Tomasi 2000).

Multiple-site or multipurpose trips. Recreation demand models assume that the particular recreation activity being studied is the sole purpose for a given trip. If a trip has more than one purpose, it likely violates the travel cost model's central assumption that the "price" of a visit is equal to the travel cost.[45] The common strategy for dealing with multipurpose trips is simply to exclude them from the data used in estimation.[46] See Mendelsohn et al. (1992) and Parsons (2003b) for further discussion.

Day trips versus multi-day trips. The recreation demand literature has focused almost exclusively on single-day trip recreation choices. Adding the option to stay longer than one day adds another choice variable in the estimation, thereby greatly increasing estimation difficulty. Also, as trip length increases multipurpose trips become increasingly more likely, again casting doubt on the assumption that a trip's travel costs represent the "price" of one single activity. 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), Shaw and Ozog (1999), and English et al. (2018) for examples. The few multi-day trip models in the literature find that the *per-day* value of multi-day trips is generally less than the value of a single-day trip, which suggests that estimating the value of multi-day trips by multiplying a value estimated for single-day trips value by the number of days of will overestimate the multi-day trip value.

7.3.1.3 Hedonic Models

Hedonic pricing models use statistical methods to measure the contribution of a good's characteristics to its price. These models are applicable to goods that can be thought of as "bundling" together many attributes that vary in quantity and quality. Houses differ in size, layout, location and exposure to environmental contaminants. Labor hours can be thought of as "goods" differing in attributes like safety risks and supervisory nature that should be reflected in wages. Hedonic pricing models use variation in prices of such goods to estimate the value of these attributes.

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

45 Parsons and Wilson (1997) suggest including a dummy variable to account for differences in multipurpose trips.

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

Economic Foundations of Hedonic Models

Hedonic pricing studies estimate economic value 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 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 the individual has to spend on food, clothing transportation, and all the other things desired or needed. So, if the difference in prices paid to live in a cleaner neighborhood is observable, then that price difference can be interpreted as the WTP for a better environment.

One key requirement in conducting a hedonic pricing study is that the available options differ in measurable ways. To see why, suppose that all locations in a city's housing market are equally polluted or that all jobs in a labor market expose workers to the same risks over a given period of time. The premium that homeowners place on environmental quality or that workers place on lower occupational risks could not be measured in this case. A hedonic pricing study requires a comparison to purchases of more expensive houses in less polluted neighborhoods, or to 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 are not very mobile between disparate sites. Comparing wages or home prices across decades or other long periods of time raises similar concerns if preferences change over time. For these reasons it is important to exercise care in defining the spatial and temporal market in which choices are made.

A related issue is that only environmental attributes or health risks that market participants are aware of and understand can be valued using hedonic pricing methods (and revealed preference methods more generally). If homeowners are unable to recognize differences in health outcomes, visibility and other consequences of differences in environmental 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. For example, groundwater contamination — such as from leaking underground storage tanks — can be difficult for homeowners to detect (Zabel and Guignet 2012). In contrast, stated preference surveys can directly ask respondents how they value changes in specific environmental commodities or health risks.

General Application by Type of Hedonic Pricing Study

Hedonic wage studies, also known as wage-risk or compensating wage studies, are based on the premise that individuals make trade-offs between wages and occupational risks of death or injury. Hedonic wage studies can assess the value to workers of changes in workplace morbidity and mortality risks, which may then be applied to environmental risks using benefit-transfer techniques. Viscusi (2013) provides an overview of the method. Most current hedonic wage studies begin with estimation of the risk, calculated as workplace fatalities per worker. The Bureau of Labor Statistics (BLS) Census of Fatal Occupational Injuries (CFOI) is the most common source for workplace risk information, a complete record of U.S. workplace fatalities since 1992. CFOI reports these fatalities by three-digit occupation and four-digit industry classifications, as well as the circumstances of the fatal events.[47] Typically, these data are used to construct the number of annual fatalities within categories such as a given industry, occupation or industry-occupation cell. This is the numerator for the annual risk rate for the hedonic wage study. Other data sources, most commonly from the Current Population Survey also conducted by BLS, are used to estimate the number of workers in these categories, providing the denominator for the annual risk rate, as well as characteristics of workers, including wage rate. The estimating equation then uses the wage rate as the dependent variable, usually in a linear or semi-log specification, and the coefficient on the risk measure provides the basis for the implicit wage-risk tradeoff for mortality risk valuation.

There are questions about the applicability of hedonic wage study results to environmental benefits assessment. For example, hedonic wage estimates are derived from populations that are working age and able to work, and they reflect preferences of those who have chosen relatively risky professions. These characteristics may differ from populations affected by environmental contaminants. There is also a difference in risk context between fatal workplace accidents and environment-related mortality from, for example, cancer. Still, hedonic wage studies have been widely used to estimate the value of fatal risk reductions, because they provide revealed-preference information on how people trade off risks for money.[48] Historically, the EPA has used a VSL estimate primarily derived from hedonic wage studies. For more information on the Agency's VSL estimate, see Appendix B.

Hedonic property value studies measure the contributions of various characteristics to property prices. These studies are typically conducted using residential housing data, but they have also been applied to commercial and industrial property, agricultural land and vacant land. Bartik (1988) and Palmquist (1988, 1991) provide detailed discussions of benefits assessment using hedonic methods. Bishop et al. (2020) review best practices for using hedonic property models to measure WTP for environmental amenities. Property value studies require large amounts of data. Market data on individual housing units' prices and other attributes are strongly preferred to aggregate data such as census tract median home values and characteristics.

Hedonic property value studies have examined the effects of air quality (e.g., Smith and Huang 1995, Bishop and Timmins 2018), water quality (e.g., Leggett and Bockstael 2000; Walsh et al. 2017; Guignet et al. 2022), natural amenities (e.g., Landry and Hindsley 2011; Guignet et al. 2017), and land contamination (e.g., Messer et al. 2006; Guignet 2013; Walsh and Mui 2017) on property values. As discussed in Section 7.1, the hedonic property approach can value changes in stressors, contaminant releases or media concentrations, or other intermediate endpoints linked to environmental benefits. The type of environmental amenity included in the analysis is often driven by data availability.

Other hedonic studies. Applicability of the hedonic pricing method is not limited to 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; Phaneuf et al. 2008; Kovacs 2012). Hedonic analysis of light-duty vehicle prices has been used to examine the value of fuel economy and other features (Espey and Nair 2005; Fan and Rubin 2010).

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

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

Considerations in Evaluating and Understanding Hedonic Pricing Studies

There are numerous statistical issues associated with applying hedonic pricing models to value changes in environmental quality and health risks. Below, we highlight the issues of identification and causality, spatial correlation, defining and measuring the environmental amenity, and interpretation of the estimates.

Identification and causality. A common challenge in hedonic pricing studies is establishing the direction of causality between the independent variable of interest (environmental quality or safety risks) and the dependent variable (e.g., home prices or wages). People choose among different houses not only because they can trade off differences in environmental amenities against price, but also because of other characteristics, like curb appeal, school quality and crime. If these other characteristics are not included in the hedonic regression and are correlated with environmental quality, then the analysis may not identify the causal impact of environmental quality on prices. In this situation, endogeneity or omitted variable bias would lead to incorrect estimates of the value of environmental quality to home buyers (Taylor, Phaneuf, Liu 2016). Similarly, if the risk of accidental death is correlated with the risk of serious, nonfatal injuries, the premium estimated in a hedonic wage equation would overstate WTP for reductions in mortality if these other risks were omitted from the regression.

Approaches to identify causal effects in the hedonic property value literature include repeat sales models, which can identify the effect of changes in environmental quality over time using the sample of homes that sold multiple times during the study period, and quasi-experimental approaches, which rely on "natural experiments" in which environmental quality varies for reasons that are exogenous to home prices. Quasi-experimental approaches include instrumental variables, regression discontinuity, matching and difference-in-difference models (e.g., Greenstone and Gayer 2009; Greenstone and Gallagher 2008; Gamper-Rabindran and Timmins 2013). Spatial fixed effects denoting discrete geographic units such as Census tracts or counties can also help control for difficult-to-measure local characteristics, but environmental quality must vary *within* this spatial unit for these models to yield useful valuation estimates (Abbott and Klaiber 2010). While fixed effects alone may not mitigate omitted variable bias if unobserved characteristics correlated with environmental quality also vary within the Census tract or other spatial unit, research has found that a combination of spatial fixed effects, quasi-experimental identification and temporal controls can greatly reduce bias (Kuminoff et al. 2010). There may also be spatial correlation in the dependent variable or the error term of the model if home prices are directly affected by the prices of nearby homes (for example, due to the home appraisal process). Spatial econometrics techniques allow analysts to account for some of these sources of dependence, reducing bias and improving the consistency or efficiency of parameter estimates (Anselin 2001). However, incorrect specification of the structure of the spatial correlation can also bias parameter estimates (Gibbons and Overman 2012). Spatial fixed effects and geographic clustering of standard errors are also useful approaches to address spatial correlation of property characteristics (Bishop et al. 2020).

Defining and measuring the environmental amenity. Another important issue is the way that the environmental amenity or health risk included in a hedonic model is defined and measured. The ideal measure is an indicator that market participants value and that can be linked to a change in environmental policy, but such measures are not always available. For instance, available water quality indicators may not fully reflect water quality or ecosystem health (Griffiths et al. 2012). Water clarity has been shown to positively affect property prices (Michael et al. 2000; Gibbs et al. 2002; Walsh et al. 2017; Guignet et al. 2022), but it is not always a good indicator of ecosystem health (Shaw, Mechenich, and Klessig 2004). Furthermore, data on water clarity may contain errors because clarity cannot be accurately measured under cloud cover (Olmanson, Bauer, and Brezonik 2008). If water clarity is measured with error or is not a good proxy for home buyers' perceptions of water quality, then measurement error could produce valuation estimates that are biased toward zero due to attenuation (Greene 2000), though empirical research has found that objective measures of water clarity have higher predictive power than individuals' subjective measure of water clarity (Poor et al. 2001).

Interpretation of the estimates. Understanding how to interpret hedonic model estimates is important. 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 WTP for a small change. Hedonic price functions typically reflect equilibria between consumer demands and producer supplies for fixed levels of the attributes being evaluated. Public policy, however, is sometimes geared to larger, discrete changes in attributes. When this is the case, using hedonic model estimates to calculate benefits is more complicated. Palmquist (1991) describes conditions under which exact welfare measures can be calculated for discrete changes. See Freeman et al. (2014) and Ekeland, Heckman, and Nesheim (2004) for treatments.

Studies that compare prices before and after a change in environmental quality using repeat sales and quasi-experimental approaches raise particular challenges for interpretation (Klaiber and Smith 2013). These approaches are sometimes called "capitalization" rather than hedonic studies because they estimate the extent to which changes in amenities are capitalized into prices over time. The capitalization effect only equals WTP if WTP remains stable over the study time horizon (Kuminoff and Pope 2010). If marginal WTP for environmental quality is increasing (decreasing) over time in the study area, then the capitalization estimate will tend to overestimate (underestimate) the benefits of cleanup.

In property value studies, if gentrification or re-sorting occurs such that people with a higher WTP move to neighborhoods with improving environmental quality, pre- and post-cleanup housing prices reflect the preferences of two distinct groups of people.[49] In addition, the capitalization estimate from repeat sales and quasi-experimental models represents the average rather than marginal change in property values that occurs in response to a change in an amenity (Parmeter and Pope 2013). If residents do not re-sort, their preferences and incomes are not changing over time, and WTP is linear in environmental quality, then a capitalization estimate can be interpreted as a measure of WTP. These conditions are less likely to hold in a study that examines a large change in environmental quality over a relatively long timespan. For example, Parmeter and Pope (2013) argue that Chay and Greenstone's (2005) quasi-experimental study of the housing price effects of improvements in air quality in nonattainment counties after passage of the 1970 Clean Air Act Amendments provides a capitalization rather than a WTP estimate because of the 10-year timespan of the study and the non-marginal reduction in air pollution. However, the assumption that a capitalization estimate provides a good approximation of WTP might be reasonable for studies covering relatively short periods of time and examining small changes in environmental quality.

49 Residential sorting models provide another alternative to hedonic and capitalization studies in the property value literature. These models derive estimates of WTP explicitly accounting for residential sorting behavior and resulting changes in a variety of neighborhood amenities (e.g., Klaiber and Phaneuf 2010; Kuminoff and Jarrah 2010).

7.3.1.4 Averting Behaviors

The averting behavior method infers values for environmental quality from observations of actions people take to avoid or mitigate the increased health risks or other undesirable consequences of reductions in environmental quality. Examples of such defensive actions can include the purchase and use of air filters, the activity of boiling water prior to drinking it and the purchase of preventive medical care or treatment. By analyzing the expenditures associated with these averting behaviors, economists can attempt to estimate the value individuals place on small changes in risk or environmental quality. Dickie (2017) provides a detailed overview of the approach.

Economic Foundations of Averting Behavior Methods

Averting behavior methods can be best understood from the perspective of a household production framework. Households can be thought of as producing health outcomes by combining an exogenous level of environmental quality with inputs such as purchases of goods that involve protection against health and safety risks (Freeman et al. 2014; Dickie 2017). 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 the individual's marginal WTP for these reductions. Thus, the value of a small change in health risks can be estimated from two primary pieces of information: (1) The cost of the averting behavior or good, and (2) its effectiveness, as perceived by the individual, in offsetting the loss in environmental quality.

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. As noted earlier in Section 7.2.1.2, 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 averting behavior method estimated the benefits of reduced soiling of materials from environmental quality changes (Harford 1984), recent research has primarily focused on health risk changes. Conceptually, the averting behavior method can provide WTP estimates for a variety of other environmental benefits such as damages to ecological systems and materials.

Some averting behavior studies focus on behaviors that prevent or mitigate the impact of specific symptoms (e.g., shortness of breath or headaches), while others have examined averting expenditures in response to specific episodes of contamination (e.g., groundwater contamination). Because many contaminants can produce similar symptoms, studies that estimate values for symptoms may be more amenable to benefit transfer than those are episode-specific and do not value specific symptoms or illnesses. 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. As in other revealed preference methods, analysts should remember that consumers base their actions on perceived benefits from their 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 regarding risk — for example, consumer risk-reduction expenditures increase as risk increases — there is also evidence that there are predictable differences between consumers' perceptions and actual risks. For example, individuals tend to overestimate risks that are very small or that are novel or unfamiliar (Renner et al. 2015). Thus, averting behavior studies can produce biased WTP estimates for a given change in objective risk. Surveys may be necessary to determine the benefits individuals perceive they are receiving when engaging in defensive activities. These perceived benefits can then be used as the object of the valuation estimates. For example, if surveys reveal that perceived risks are lower than expert risk estimates, then WTP for risk reduction can be estimated with the lower, perceived risk (Blomquist 2004).

Data requirements and implications. Data needed for averting behavior studies include information detailing the effect(s) being averted (e.g., specific illnesses, exposure to environmental contaminants); actions taken to avert or mitigate damages; the costs of those behaviors and activities; and other variables that affect health outcomes, like age, health status or chronic conditions. Another significant challenge in many averting behavior applications is that output level (e.g., health) is unobserved and may change when averting actions are taken, which complicates calculation of WTP.

Separability of joint effects. Analysts should exercise caution in interpreting the results of studies that focus on goods in which there may be significant joint benefits (or costs). Many defensive behaviors not only avert or mitigate environmental damages, but also provide other benefits. For example, air conditioners obviously provide cooling in addition to air filtering, and bottled water may not only reduce health risks, but also taste better. Conversely, it also is possible that the averting behavior may have negative effects on utility. For example, wearing helmets when riding bicycles or motorcycles may be uncomfortable. Failure to account for these "joint" benefits and costs associated with averting behaviors will result in biased estimates of WTP.

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

7.3.1.5 Cost of Illness

A frequent alternative to WTP estimates is the avoided COI, which estimates the resource costs associated with an adverse health effect (an "illness"). Though WTP is the preferred valuation measure for BCA, the COI method is straightforward to implement and explain to policy makers, and has been widely applied, particularly in health economics or studies of the burden of disease (e.g., Trasande et al. 2016). COI estimates for many illnesses are readily available from existing studies. COI estimates are usually less expensive to develop than WTP estimates using stated or revealed preference approaches, so it may be feasible to develop new COI estimates for a given benefits analysis even with typical time and resource constraints. Jo (2014) and Tarricone (2006) provide overviews of the method.

Economic Foundations of COI Studies

Relating cost of illness to WTP. The COI method does not estimate WTP but is based on estimating the market value of goods and services used to treat illness and the lost productivity due to that illness. It does not incorporate any disutility from illness, the costs of averting behaviors taken to avoid the illness, or risk preferences that would be inherent in estimates of WTP to reduce risks of illness. Because of these limitations COI is best viewed as a proxy for WTP when WTP estimates are not available, and is generally considered to be a lower bound on WTP, especially for more serious illnesses.[50] Available comparisons of COI and total WTP estimates suggest that the difference can be large but varies greatly across health effects; COI estimates cannot be simply "scaled up" to approximate WTP.

In some cases, COI may be additive to a WTP estimate that did not account for certain costs. COI estimates capture medical expenses passed on to third parties such as health insurance companies and hospitals, whereas WTP estimates generally do not. COI estimates can capture the value of lost productivity, something that may be overlooked in WTP estimates — especially when derived from consumers or employees covered by sick leave. In practice, because there is a risk of double-counting when adding COI and WTP, doing so requires a careful evaluation of the studies in question, what they each do and do not include, and how they can be appropriately added together.

Types of costs considered in COI studies. COI studies generally distinguish between direct costs (costs related to medical treatment) and indirect costs (costs related to lost productivity). Many COI studies estimate both direct and indirect costs, but some may focus solely on costs of treatment while other studies, broadly categorized as COI, only reflect lost productivity.

- **Direct Costs** are those related to treatment and care for the illness. These costs include the value of goods and services spent for items such as physicians' services, testing, hospitalization, medications and medical devices. But it also includes the value of household expenditures, transportation, accommodation and other resources spent on care for the illness. COI studies may not capture all of these costs. For example, studies relying solely on databases of medical expenditures might not capture the costs of household expenditures.
- **Indirect costs** refer to productivity losses associated with the illness, most often measured by the human capital approach where earnings reflect the value of productive time.[51] That is, assuming the wage equals the value of marginal product. Losses to productivity, therefore, are a social cost and can be measured by the wage rate.[52] Lost productivity may be focused on the short-term, e.g., for illnesses where the losses are associated with a loss of workdays or, for more serious illnesses a permanent loss of income.

  In principle, indirect costs should also consider the costs of lost home productivity and the value of leisure, but this is not always done in COI studies. Lost productivity for home health care, e.g., the time spent by members of the household in caring for family members or accompanying patients to medical appointments, should also be included in indirect costs.

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50 However, any particular COI estimate is not necessarily going to be lower than WTP for a given health condition. Depending on the design of the studies, WTP could reflect avoidance costs that are lower than the cost of illness once the illness has been contracted.

51 For an in-depth discussion of the valuation of time in different contexts, see U.S. EPA (2020).

52 The EPA has a similar approach for cost analysis that is also based on the opportunity cost of time; see U.S. EPA "Handbook on Valuing Changes in Time Use Induced by Regulatory Requirements and Other EPA Actions" (U.S. EPA 2020).

Note that the human capital approach applies not just to lost work time at a given wage in what would be considered a traditional COI study but for any impact on productivity associated with adverse health effects. For example, lowered IQ – an effect associated with exposures to many pollutants – has been related to labor participation and lower lifetime earnings, a loss of human capital (Salkever 1995; Lin et al. 2018). This relationship can be useful in economic analyses to value the benefits of avoiding IQ losses. Additionally, exposure to ozone has been linked to loss of productivity among agricultural workers (Crocker and Horst 1981; Graff Zivin and Neidell 2012).[53]

General Application by Type of COI Study

Prevalence-based estimates. Prevalence-based COI estimates are derived from the costs faced by all individuals who have a sickness in a specified time period. For example, an estimate of the total number of individuals who currently have asthma, as diagnosed by a physician, reflects the current prevalence of physician-diagnosed asthma. Prevalence-based COI estimates for asthma include all direct and indirect costs associated with asthma within a given time period, such as a year. Prevalence-based COI estimates are a measure of the financial burden of a disease but will still generally underestimate total WTP for avoiding the disease altogether. They are most applicable for valuation of policies that reduce or eliminate morbidity associated with existing cases of illness.

Incidence-based estimates. By contrast, incidence-based COI estimates reflect expected costs for *new* cases of an illness 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 should be discounted to the year the illness is diagnosed and expressed in present value terms. Incidence-based COI estimates are most applicable for valuation of policies that reduce the expected number of new cases of disease, which is often the case for environmental regulations.

Bottom-up, top-down and econometric approaches. There are three primary methods for estimating COI for a given health condition. The "bottom up" approach constructs a typical profile of treatment for the condition and then uses unit costs to estimate total treatment costs over time, usually based on databases of medical expenditures. The "top-down" approach, on the other hand, typically starts with aggregate expenditures across a number of illnesses and then attributes these expenditures across that set of illnesses. Finally, the econometric approach to COI typically uses data on total costs for a given sample over a given time period and then econometrically estimates the difference in costs between those with and without a given health condition. The difference provides an estimate of the cost of treatment for the illness. Bottom-up or econometric approaches are generally best-suited for benefits analysis.

Considerations in Evaluating and Understanding COI Studies

Technological change. Medical treatment technologies and methods are constantly changing, and this could push the true cost estimate for a given illness either higher or lower. When using

53 For examples of how productivity estimates have been used in economic analyses, see the primary benefits analysis for the 2011 Transport Rule as well as the supplemental benefits analysis for the 2015 Ozone NAAQS.

previous COI studies, the analyst should be sure to research whether and how the generally accepted treatment has changed from the time of the study.

Measuring health care costs. The COI literature uses a variety of methods to measure health care costs.[54] One important distinction is between medical expenditures and medical charges. Expenditures are the better indicator of social costs because they better represent actual resources used by healthcare providers rather than the "list price," which is often discounted. Studies that rely on medical charges may use them as-is or try to approximate expenditures using hospital-specific cost-to-charge ratios. For benefits analysis adjusted charges are better than unadjusted charges, but studies that use expenditure data are even more preferred.

Measuring the value of lost productivity. The value of lost productivity in many studies may only reflect persons in the work force, omitting the productivity costs 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. In cases where an affected individual requires a caregivers' assistance, e.g., when children, elderly or impaired individuals are affected, the caregiver may also incur time away from work and lost productivity. The value of lost leisure time to an individual and their family is not included in most COI studies.[55] A second set of considerations is the choice of wage rate in the study, which will reflect the study population and may not match the wage rate of the population in the policy case.

7.3.2 Stated Preference

The distinguishing feature of stated preference methods compared to revealed preference methods is that stated preference methods rely on people's responses to hypothetical questions while revealed preference methods rely on observations of actual choices. Stated preference methods use surveys that ask respondents to consider one or a series of hypothetical scenarios that describe a potential change in a non-market good. The advantages of stated preference methods include their ability to estimate non-use values and to incorporate hypothetical scenarios that closely correspond to a policy case. The main disadvantage of stated preference methods is that they may be subject to systematic biases that are difficult to test for and correct. For this reason, OMB Circular A-4 advises,

> "If both revealed-preference and stated-preference studies that are directly applicable to regulatory analysis are available, you should consider both kinds of evidence and compare or combine the findings when feasible. If the results diverge significantly, you should, when feasible, compare the overall quality of the two bodies of evidence. Other things equal, revealed preference data are preferable to stated preference data because revealed preference data are based on actual decisions." (p.37)

The *Report of the NOAA Panel on Contingent Valuation* is often cited as an early source of recommendations for best practices for stated preference studies. 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 for the National Oceanic and Atmospheric Administration (NOAA) (Arrow et al. 1993). The panel focused

54 See Onukwugha, et al. 2016 for a review of methods and their prevalence in the COI literature.

55 For an in-depth discussion of the valuation of time in different contexts, see U.S. EPA 2020.

on a rather narrow application of stated preference — the use of contingent valuation to estimate non-use values for litigation in the United States. In the years since, stated preference research has advanced significantly and its applications have expanded to more diverse contexts. In 2012, *The Journal of Economic Perspectives* published a symposium on contingent valuation including an assessment of the state of the science, prompting additional discussion in a 2013 *Applied Economic Perspectives and Policy* paper (Haab et al. 2013). More recently, Johnston et al. (2017) published an updated set of guidelines that reflects contemporary stated preference research, changes in survey methods and technology and the transfer of primary estimates to different policy scenarios.

7.3.2.1 Economic Foundation of Stated Preference Methods

The role of non-use value in BCA has been well established since the 1990s (see Kopp 1992, and Bishop and Welsh 1992 for early discussions of non-use value and welfare theory). Further, ignoring non-use value in environmental regulatory analysis can lead to large omissions in benefits estimation and a misallocation of resources. A regulatory analysis should carefully consider when non-use values might be substantial and, given stated preference is the only valuation approach that captures them, what studies are available to draw from and how to evaluate the validity of their results.

The responses elicited from stated preference surveys, if truthful, unbiased and well-informed, are either direct expressions of WTP or can be used to estimate WTP for the good in question. However, the caveats listed above are paramount. While many environmental economists believe that respondents can provide truthful answers to hypothetical questions and therefore view stated preference methods as useful and reliable if conducted properly, a non-trivial fraction of economists are more skeptical of the results elicited from stated preference surveys. Due to this skepticism, it is important to employ validity and reliability tests of stated preference results before applying them to policy decisions.

If the analyst decides to conduct a stated preference survey or use stated preference results in a benefit transfer exercise, then a number of survey design issues should be considered. Stated preference researchers have attempted to develop methods to make individuals' choices in stated preference studies as consistent as possible with market transactions or consequential referenda. Reasonable consistency with the framework of market transactions is a guiding criterion for ensuring the validity of stated preference value estimates. Three components of market transactions need to be constructed in stated preference surveys: the commodity, the payment and the scenario (Fischoff and Furby 1988).

Stated preference studies need to carefully define the commodity to be valued, including characteristics of the commodity such as the timing of provision, certainty of provision and availability of substitutes and complements. The definition of the commodity generally involves identifying and characterizing attributes of the commodity that are relevant to respondents. Commodity definition also includes defining or explaining baseline or current conditions, property rights in the baseline, the policy scenarios, as well as the source of the change in the environmental commodity.[56]

56 Depending on the scenario, the description of the commodity may produce strong reactions in respondents and could introduce bias. In these cases, the detail with which the commodity of the change is specified needs to

Respondents also must be informed about the transaction context, including the method, timing, and duration of payment. The transaction must not be coerced, and the individual should be aware of her budget constraint. The payment vehicle should be described as a credible and binding commitment should the respondent decide to purchase the good (Carson et al. 1997). The timing and duration of a payment involves individuals implicitly discounting payments and calculating expected utility for future events. The transaction context and the commodity definition should describe and account for these temporal issues.

The hypothetical scenario(s) should be described to minimize potential strategic behavior such as "free riding" or "yea-saying." In the case of free riding, respondents will underbid their true WTP for a good if they believe it will be provided regardless of their response. In the case of yea-saying, respondents pledge amounts greater than their true WTP with the expectation that they will not be made to pay for the good but believing that their response could influence whether the good will be provided. Incentive-compatible choice scenarios and attribute-based response formats have been shown to mitigate strategic responses. Both are discussed below.

It is recognized in both the experimental economics and the survey methodology literatures that different survey formats can elicit different responses. Changing the wording or order of questions can also influence responses. Therefore, the researcher should provide a justification for her choice of survey format and include a discussion of the ramifications of that choice.

7.3.2.2 General Application by Type of Stated Preference Study

Two main types of stated preference survey format are currently used: direct WTP questions and stated choice questions. Stated choice questions can be either dichotomous choice questions or multi-attribute choice experiments. 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 use focus groups, pretests and statistical validity tests to address known and suspected weaknesses in the selected approach.

Direct/Open-Ended WTP Questions

Direct/open-ended WTP questions ask respondents to indicate their maximum WTP for the specific quantity or quality changes of a good or service that has been described to them. An important advantage of open-ended stated preference questions is that the answers provide direct, individual-specific estimates of WTP which requires a smaller sample size and simpler estimation approach. While these advantages could lower the cost of the study, early stated preference studies found that some respondents had difficulty answering open-ended WTP questions and non-response rates to such questions were high. Such problems are more common when the respondent is not familiar with the good or with the idea of exchanging a direct dollar payment for the good. An example of a stated preference study using open-ended questions is Brown et al. (1996).

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be balanced against the ultimate goals of the survey. Regardless, the commodity needs to be specified with enough detail to make the scenario credible.

Various modifications of the direct/open-ended WTP question format have been developed 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; Whitehead 2002). A *payment card* is a list of dollar amounts from which respondents can choose, allowing respondents an opportunity to look over a range of dollar amounts while they consider their maximum WTP. Mitchell and Carson (1989) and Rowe et al. (1996) discuss concerns that the range and intervals of the dollar amounts used in payment card methods may influence respondents' WTP answers.

Stated Choice Questions

While direct/open-ended WTP questions are efficient in principle, researchers have generally turned to other stated preference techniques in recent years. This is largely due to 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 et al. 2014).

In contrast to direct/open-ended WTP questions, stated choice questions ask respondents to choose a single preferred option or to rank options from three or more choices. When analyzing the data, the dependent variable will be continuous for open-ended WTP formats and discrete for stated choice formats.[57] In principle, stated choice questions can be distinguished along three dimensions:

- *The number of alternatives each respondent can choose from in each choice scenario* — surveys may offer only two alternatives (e.g., yes/no, or "live in area A or area B"); two alternatives with an additional option to choose "don't know" or "don't care;" or multiple alternatives (e.g., "choose option A, B or C").
- *The number of attributes varied across alternatives in each choice question (other than price)* — alternatives may be distinguished by variation in a single attribute (e.g., mortality risk) or multiple attributes (e.g., mortality risk, length and severity of illness, source of risk, 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. 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

57 Some researchers use the term "contingent valuation" to refer to direct WTP and dichotomous choice/referendum formats and "stated preference" to refer to other stated choice formats. In these Guidelines, the term "stated preference" is used to refer to all valuation studies based on hypothetical choices (including open-ended WTP and stated choice formats), as distinguished from "revealed preference."

were $X?" However, these questions are not always posed as a vote decision and are now commonly called *dichotomous choice* questions.

Stated preference researchers have also adapted 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. Including cost as an attribute and varying it across options allows researchers to estimate marginal WTP for each attribute of the good. Holmes and Adamowicz (2003) refer to this as *attribute-based stated choice*.

Dichotomous choice WTP questions. Dichotomous choice questions present respondents with a specified environmental change costing a specific dollar amount and then ask whether 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 and less prone to manipulation than direct WTP questions, because the respondent is not required to determine their exact WTP, only whether it is above or below the stated amount. Sample mean and median WTP values can be derived from analysis of the frequencies of the yes/no responses to each dollar amount. Bishop and Heberlein (1979), Hanemann (1984), and Cameron and James (1987) describe the necessary statistical procedures for analyzing dichotomous choice responses using logit or Probit models. Dichotomous choice responses will reveal an interval containing WTP, and in the case of a "yes" response this interval will be unbounded from above. As a result, significantly larger sample sizes are needed for dichotomous choice questions to obtain the same degree of statistical efficiency in the sample means as direct/open-ended responses that reveal point-values for WTP (Cameron and James 1987).

To increase the estimation efficiency of dichotomous choice questions, some studies have 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.[58] Sometimes multiple follow-up questions are used to try to narrow the interval around WTP even further. These begin to resemble iterative bidding style questions if many follow-up questions are asked. Similar to starting point bias in iterative bidding questions, the analyses of double-bounded dichotomous choice question results suggest that the second responses may not be independent of the first responses (Cameron and Quiggin 1994, 1998; and Kanninen 1995).

Multi-attribute choice questions. In multi-attribute choice questions (also known as conjoint analysis), respondents are presented with alternative choices that are characterized by different combinations of attributes and prices. Multi-attribute choice questions ask respondents to choose the most preferred alternative (a partial ranking) from multiple alternative goods (i.e., a choice set), in which the alternatives within a choice set are differentiated by their attributes including price (Johnson et al. 1995 and Roe et al. 1996). The analysis takes advantage of the differences in the attribute levels across the choice options to determine how respondents value marginal changes in each of the attributes. To measure WTP, a cost (e.g. a tax or measure of travel costs) is included in multi-attribute choice questions as one of the attributes of each alternative. This price and the mechanism by which the price would be paid need to be explained clearly and plausibly, as with any payment mechanism in a stated preference study. Boyle and Özdemir (2009) examine the

58 Alberini (1995) illustrated an analysis approach for deriving WTP estimates from such responses and demonstrated 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.

impact of question design choices, such as the ordering of attributes and the number of alternatives in a single question, on the mean WTP estimate.

There are many desirable aspects of multi-attribute choice questions, including the nature of the choice being made. To choose the most preferred alternative from some set of alternatives is a common decision experience in posted-price markets, especially when one of the attributes of the alternatives is a price. One can argue that such a decision encourages respondents to concentrate on the trade-offs between attributes rather than taking a position for or against an initiative or policy. This type of repeated decision process may also diffuse the strong emotions often associated with environmental goods, thereby reducing the likelihood of yea-saying or of rejecting the premise of having to pay for an environmental improvement. Presenting repeated choices also gives the respondent some practice with the question format, which may improve the overall accuracy of her responses, and gives her repeated opportunities to express support for a program without always selecting the highest price option.[59] One challenge of attribute-based methods is representing the environmental changes in small number of separable attributes. Extensive focus group research is required to choose the most salient attributes and find the best way to convey those changes to respondents. To estimate marginal values, the attributes must be able to change independent of one another without respondent rejecting the scenario.

7.3.2.3 Considerations in Evaluating Stated Preference Results

Survey mode. The mode used to administer a survey is an important component of survey research design because it is the mechanism by which information is conveyed to respondents, and likewise determines the way in which individuals can provide responses for analysis. Common survey modes include telephone, in-person, mail and electronic surveys administered by computer or smart phone. 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. Electronic surveys are computerized and can be self-administered at a central facility, at home via the internet or on a smart phone wherever internet access is available. As technology and society have changed, so has the preference for one mode over the other. With the influx of market research, telemarketing, telephone scams and the abandonment of landlines, the telephone has become a less effective way to administer surveys. Similarly, response rates to mail surveys have declined substantially (Stedman et al. 2019). With increased prevalence of smartphone technology, internet access and email accessibility, computerized surveys have emerged as an expedient means of survey administration. Researchers may also choose to combine modes using one for recruiting and the other for survey administration. With every survey mode mentioned, there are inherent biases. These biases are generally classified as social desirability bias, sample frame bias, avidity bias, and non-response bias. See Maguire (2009), Loomis and King (1994), Mannesto and Loomis (1991), Lindberg et al. (1997), and Ethier et al. (2000) for a discussion of different biases in survey mode.

Framing effects. 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

59 Some applications of multi-attribute survey formats include Layton and Brown (2000), Boyle et al. (2001), Morey et al. (2002), and Moore et al. (2018). Studies that investigate the effects of multi-attribute choice question design parameters include Johnson et al. (2000) and Adamowicz et al. (1997).

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 compulsory and does not introduce strategic responses or bias. Voluntary payment vehicles such as donations can be subject to free riding behavior and cause respondents to overstate their willingness to pay.

Incentive Compatibility. A survey instrument is incentive compatible when respondents are motivated to answer truthfully and do not use their responses to try to influence a particular outcome. Incentive compatibility is driven primarily by the consequentiality of the survey and the question format used. Consequentiality requires that survey participants believe there Is a positive probability that the survey outcome will have actual consequences. Establishing a link between survey responses and actual outcomes described by the scenario mitigates several types of bias associated with stated preference valuation including hypothetical bias and yea-saying (Cummings and Taylor 1999; Carson and Groves 2007; Landry and List 2007; Vossler and Evans 2009; Herriges et al. 2010; Vossler and Watson 2013). An incentive compatible question format will reduce strategic behavior by respondents. Single binary choice formats meet this criterion with the least assumptions though other formats may be incentive compatible under strict conditions (Johnston et al. 2017).

Treatment of "don't know" or neutral responses. Based on recommendations from the NOAA Blue Ribbon panel (Arrow et al. 1993), many surveys have included "don't know" or "no preference" options. The *Contemporary Guidance for Stated Preference* (Johnston et al. 2017) recommends including a no-answer option for sensitive topics but not necessarily for all applications. There have been questions about how such responses should enter the empirical analysis. Examining referendum-style dichotomous choice questions, Carson et al. (1998) found that when those who chose not to vote were coded as "no" responses, the mean WTP values were the same as when the "would not vote" option was not offered. Offering the "would not vote" option did not change the percentage of respondents saying "yes". Thus, they recommend that if a "would not vote" option is included, it should be coded as a "no" vote, a practice that has become widespread. Stated preference studies should always be explicit about how they treat "don't know," "would not vote," or other neutral responses.

Reliability, in general terms, means consistency or repeatability. If a method is used numerous times to measure the same commodity, then the method is considered more reliable if the variability of the results is lower than an alternative.

- **Test-retest approach.** Possibly the most widely applied approach for assessing reliability in stated preference studies has been the test-retest approach. Test-retest assesses the variability of a measure between different time periods. Loomis (1989), Teisl et al. (1995), McConnell et al. (1998), and Hoban and Whitehead (1999) all provide examples of the test-retest method for reliability.
- **Meta-analysis of stated preference survey results** for the same good also may provide evidence of reliability. Meta-analysis evaluates multiple studies as though each was constructed to measure the same phenomenon. Meta-analysis attempts to sort out the effects of differences in the valuation approach used in different surveys, along with other factors influencing the elicited value. For example, Boyle et al. (1994) use meta-analysis to

evaluate eight studies conducted to measure values for groundwater protection (also see Section 7.4).

Validity tests seek to assess whether WTP estimates from stated preference methods behave as a theoretically correct WTP should. Three types of validity discussed below are: content validity, criterion validity and convergent validity.

- **Content validity.** Content validity refers to the extent to which the estimate captures the concept being evaluated. Content validity is largely a subjective evaluation of whether a study has been designed and executed in a way that incorporates the essential characteristics of the WTP concept.

  To evaluate a survey instrument, analysts look for features that researchers should have incorporated into the survey scenario. First, survey should clearly define the environmental change being valued. The description should include a careful exposition of the conditions in the baseline case and how these would be expected to change over time if no action were taken. Next, the action or policy change should be described, including an illustration of how and when it would affect aspects of the environment that people might care about. Boyd and Banzahf (2007), and Boyd and Krupnick (2013) put a finer point on this concept and advocate developing the valuation scenario based on "ecological endpoints" rather than intermediate goods that are less clearly associated with outcomes of interest. For example, if respondents ultimately care about the survival of a certain species, it is more sensible to structure questions to ask about WTP for the species' survival than to ask about degradation of habitat, as respondents are unlikely to know the relationship between habitat attributes and species survival. Respondent attitudes about the provider and the implied property rights of the survey scenario can be used to evaluate the appropriateness of features related to the payment mechanism (Fischhoff and Furby 1988). Survey questions that probe for respondent comprehension and acceptance of the commodity scenario can offer important indications about the validity of the results (Bishop et al. 1997).

- **Criterion validity.** Criterion validity assesses whether stated preference results relate to other measures that are considered to be closer to the concept being assessed (or WTP). Ideally, one would compare results from a stated preference study of use values (the measure) with those from actual market data (the criterion). Another approach would be to estimate a sample of individuals' WTP for a commodity using a stated preference survey and then later give the same sample of individuals or a different random sample of individuals drawn from the same population a real opportunity to buy the good (see Mitchell and Carson 1989; Carson et al. 1987; Kealy et al. 1990; Brown et al. 1996; and Champ et al. 1997 for examples.)

  When unable to conduct such comparisons, sensitivity to scope and income has been used to assess criterion validity. "Scope tests" are concerned with how WTP responds to changes in the amount of the referenced good provided in the valuation scenario (Smith and Osborne 1996; Rollins and Lyke 1998; Heberlein et al. 2005). If the referenced good is indeed a "normal good," utility theory implies that WTP should increase with the provision of the good. For the same reason one would expect WTP to exhibit positive income elasticity (McFadden 1994 and Schlapfer 2006). Neither test is necessary or sufficient to establish criterion validity (Heberlein et al. 2005), but either can serve as a useful proxy when an alternate measure of WTP for the same good is unavailable. Diamond (1996) suggests that stronger scope tests can be conducted by comparing departures from strict "adding up" of WTP for partial changes and relating them to the income elasticity of WTP. Other researchers, however, argue that the Diamond test may not be practicable and imposes a specific structure on the preference function which may not be appropriate (Carson et al. 2001).

Convergent validity. Convergent validity examines the relationship between different measures of a concept.[60] This differs from criterion validity in that one of the measures is not taken as a criterion upon which to judge the other measure. The measure of interest and the other measure are judged together to assess consistency with one another. If they differ in a systematic way (e.g., one is usually larger than another for the same good), it is not clear which one is more correct. If stated preference estimates are being compared with revealed preference estimates, care should be taken that the same values are being captured by both approaches. Stated preference estimates often include non-use values whereas revealed preference estimates do not capture that portion of total economic value.

Hypothetical bias occurs when the responses to hypothetical stated preference questions are systematically different than what individuals would pay if the transactions were to actually occur. Widely cited as one of the most common problems with the stated preference method (List and Gallet 2001 and Murphy and Allen 2005), researchers have made advances in techniques to minimize such bias. These techniques include the use of "cheap talk" methods to directly tell respondents about the potential for hypothetical bias (Cummings and Taylor 1999 and List 2001); calibrating hypothetical values (List and Shogren 1998 and Blomquist et al. 2009); and allowing respondents to express uncertainty in their responses and restricting the set of positive responses to those about which the respondent was most certain (Vossler et al. 2003). Several studies have shown that attribute-based choice experiments reduce hypothetical bias in the bid amounts and the marginal value of attributes relative to other elicitation methods (Carlsson and Martinsson 2001: Murphy and Allen 2005; List et al. 2006).

Tests for hypothetical bias often involve a comparison of actual payments and responses to hypothetical scenarios that use the same solicitation approach. The actual payments typically occur in one of three scenarios. Market transactions are the most common (Cummings et al. 1995 and List and Shogren 1998) but generally involve payments for private goods while most stated preference applications are concerned with public or quasi-public goods. Simulated markets can be used to solicit actual donations for public good provision (Champ et al. 1997). However, donation solicitations are subject to free riding, so while it may be possible to test for hypothetical bias using this approach, both the actual and hypothetical payment scenarios lack incentive compatibility and may not represent total WTP. In rare instances comparisons have been made between actual referenda for public good provision and hypothetical responses to the same scenario, but the conditions for a valid comparison of this sort are exceedingly difficult to satisfy (Johnston 2006). Some experiments have, however, simulated public good provision in various ways and with varying levels of success (Carson et al. 2001; Landry and List 2007; Vossler and Evans 2009; Vossler et al. 2012).

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.

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60 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.

- **Item non-response bias** occurs when respondents who agreed to take the survey do not answer all the choice questions in the survey. Information available about respondents from other questions they answered can support an assessment of potential item non-response bias for the WTP questions that were unanswered. The key issue is whether there were systematic differences in potential WTP-related characteristics of those who answered the WTP questions and those who did not. Characteristics of interest include income, gender, age, expressed attitudes and opinions about the good or service, and information reported on current use or familiarity with the good or service. Statistically significant differences may indicate the potential for item non-response bias, while finding no such differences suggests that the chance of significant non-response bias is lower. However, the results of this comparison are only suggestive because respondents and non-respondents may only differ in their preference for the good in question (McClelland et al. 1991).
- **Survey non-response bias** is created when those who refuse to take the survey have preferences that are systematically different from the preferences of those who do respond. Although it is generally thought that surveys with high response rates are less likely to suffer from survey non-response bias, it is not a guarantee.[61] 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 approach is to examine the relevant measurable characteristics of the respondent group, such as income, resource use, gender, age, etc., and to compare them to the characteristics of the study population. Similarity in mean characteristics across the two groups suggests that the respondents are representative of the study population and that non-response bias is expected to be minimal.

  A second way to evaluate potential survey non-response bias is to conduct a short follow-up survey with non-respondents. This can sometimes be accomplished through interviews conducted during the recruiting phase. Such follow-ups typically ask a few questions about attitudes and opinions on the topic of the study as well as collecting basic socioeconomic information. Questions need to match those in the full survey closely enough to compare non-respondents to respondents. The follow-up must be very brief, or response rates will be low (OMB 2006).

7.3.3 Combining Revealed and Stated Preference Data

Instead of looking at revealed preference and stated preference data as two separate methods for estimating environmental benefits, many researchers have used them in combination. The practice has been in use much longer in the marketing and transportation literature and many of the lessons learned by those researchers are now being employed in environmental economics. In theory, the strengths of each data type should help overcome some of the weaknesses of the other. As described by Whitehead et al. (2008) in an assessment of the state of the science, the advantages of combining revealed preference and stated preference data include:

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61 Note that OMB's Guidance on Agency Survey and Statistical Collections (OMB 2006) has fairly strict requirements for response rates and their calculation for Agency-sponsored surveys, recommending that "ICRs for surveys with expected response rates of 80 percent or higher need complete descriptions of the basis of the estimated response rate...ICRs for surveys with expected response rates lower than 80 percent need complete descriptions of how the expected response rate was determined, a detailed description of steps that will be taken to maximize the response rate...and a description of plans to evaluate non-response bias" (pp. 60-70).

- Helping to ground the hypothetical stated preference data with real world behavior, potentially decreasing any hypothetical bias;
- Providing the ability to test the validity of both data sources;[62]
- Increasing the range of historical stated preference data to include conditions not observed in the past and thereby reducing the need to make predictions outside of the sample;
- Increasing the sample size;
- Extending the size of the market or population to include larger segments than captured by either method alone; and
- Exploiting the flexibility of stated preference experimental design to overcome revealed preference data's potential multicollinearity and endogeneity problems (von Haefen and Phaneuf 2008).

The different strategies for combining revealed preference and stated preference data can be grouped into three categories. The first two methods rely on joint estimation. If the revealed preference and stated preference data have similar dependent and independent variables and the same assumed error structures, then they can simply be pooled together and treated as additional observations (Adamowicz et al. 1994; Boxall, Englin, and Adamowicz 2003; and Morgan, Massey, and Huth 2009). If the revealed preference and stated preference data sources cannot be pooled, it is sometimes possible to use them in a jointly estimated mixed model that relies on a utility theoretic specification of the underlying WTP function (Huang, et al. 1997; Kling 1997; Eom and Larson 2006, Jeon and Herriges 2016, Whitehead and Lew 2020; Hindsley et al. 2022). If the data cannot be combined in estimation, it can still be useful to estimate results separately and then use them to test for convergent validity between the two data sources (Carson et al. 1996 and Schlapfer et al. 2004).

7.4 Benefit Transfer

As noted at the outset of this chapter, benefit transfer is the approach most often used by the Agency for monetizing benefits in economic analysis. Benefit transfer refers to the use of estimated values of environmental quality changes from primary studies to the evaluation of similar changes that are of interest to the analyst (Freeman et al. 2014). 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 preference or revealed preference study cases that sufficiently relate to the policy context and transfers their results to the policy case.

Benefit transfer is necessary when it is infeasible to conduct an original study focused directly on the policy case and is the most common approach for completing a BCA at the EPA. Given the time and analytical resource constraints under which most regulatory analysis activities are conducted, conducting new revealed or stated preference studies that are tailor-made to examine all of the (sometimes numerous) endpoints changed by the policy or regulation in question is near impossible (Newbold et al. 2018a). Because 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. Benefit transfer might also be useful as a scoping exercise to predict the

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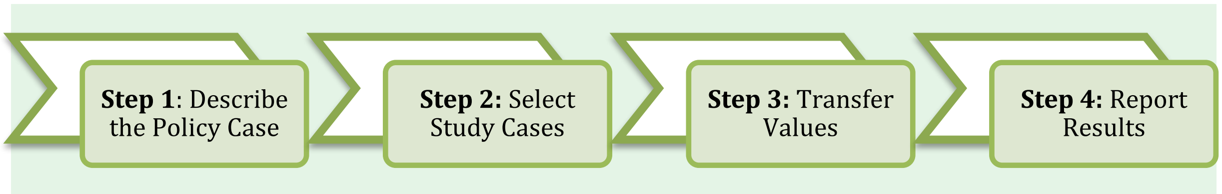
62 Herriges, Kling, and Phaneuf (2004) point out that revealed preference may not always be valid for estimating WTP for quality changes when weak complementarity cannot be assured.

approximate magnitude of benefits that might then be more precisely estimated with an original study.

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.

Figure 7.3 - Steps for Conducting Benefit Transfer



Step 1: Describe the Policy Case

The first step in a benefit-transfer study is to clearly describe the policy case with respect to the baseline so that its characteristics and consequences are well-understood. Are human health risks reduced by the policy intervention? Are ecological benefits expected (e.g., increases in populations of species of concern)? It is also important to identify to the extent possible the population affected by the proposed policy and to describe their demographic and socioeconomic characteristics (e.g., users of a particular set of recreation sites, children living in urban areas or older adults across the United States). Information on the affected population is generally required to translate per person (or per household) values to an aggregate benefits estimate.

Step 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 conducting a comprehensive literature search. Online searchable databases summarizing valuation research may be especially helpful at this stage.[63] Because peer-reviewed academic journals may be more likely to publish work based on methodological contributions, analysts should be aware of the potential for publication bias. Some studies of interest may be found in government reports, working papers, dissertations, unpublished research and other "gray literature" (Rosenberger and Stanley 2006; Johnston and Rosenberger 2010; Johnston et al. 2015).[64] While including studies from the gray literature may help mitigate publication bias, use of valuation estimates that are not published in peer-reviewed journals may necessitate subsequent peer review in some form (i.e., formal peer review or a less formal peer input). See U.S. EPA (2015a) for more guidance (in particular, Sections 3.5.7 and 3.5.8).

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63 For example, the Environmental Valuation Reference Inventory (EVRI) is maintained by Environment Canada and managed by a cross-county working group. EVRI contains summaries of over 4,000 studies that can be referenced according to keyword, study type, region and environmental asset. EVRI also provides a bibliography on benefit transfer. See www.evri.ca for more information.

64 Newer, unpublished research may also be on the cutting edge of methods.

Next, the analyst should develop an explicit set of selection criteria to evaluate each of the potentially relevant studies for quality and applicability to the policy case. The quality of the value estimates in the study cases will in large part determine the quality of the benefit transfer. As a first step, the analyst should review studies according to the criteria listed for each methodology in the previous sections in this chapter. Results from study cases must be valid as well as relevant. Concerns about the quality of the studies, as opposed to their relevance, will generally hinge on the methods used. Valuation approaches commonly used in the past may now be regarded as unacceptable for use in benefits analysis. Studies based on inappropriate methods or reporting obsolete results should be removed from consideration.

It is unlikely that any single study will match perfectly with the policy case; however, study cases potentially suitable for use in benefit transfer should be similar to the policy case in their: (1) definition of the environmental commodity being valued (include scale and presence of substitutes); (2) baseline and extent of environmental changes; and (3) characteristics of affected populations. [65] Analysts should avoid using benefit transfer in cases where the policy or study case have large differences in context--especially concerning goods with unique attributes (such as a national park), where the valuation estimate is *ex ante* and the policy case is *ex post*--especially if the policy introduces a significant change in the attributes of the good, or where the magnitude of the change or improvement across the two cases differs substantially (OMB 2023).[66] It is crucial to remember that economic value is determined on the margin and depends upon how scarce something is relative to the demand for it at the time and place it is provided (Simpson 2017).

The analyst should determine whether adjustments should and can be made for important differences between each study and policy case. For example, some case studies will be based on stated preference methods while others may be based on revealed preference methods. The ability of the analyst to make these adjustments will depend, in part, on both the number of value estimates for suitably similar study sites and the method used to combine these estimates. These methods are now discussed in turn.

Step 3: Transfer Values

There are several approaches for transferring values from study cases to the policy case. These include unit value transfers and function transfers, and they may use techniques from meta-analysis if multiple studies are available. Transfers may also be structural or non-structural (referring to a utility-theoretic structure). 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 estimate total benefits. In general, when reporting transfer results, researchers should provide information on the background of the problem, the strategy for selecting studies, analytic methods used, results, discussion and conclusions.

- ***Unit value transfers*** are the simplest of the benefit-transfer approaches. They take a point estimate of WTP for a unit change in the environmental resource from a study case or cases and apply it directly (or simply adjusted) to the policy case. The point estimate may be a

65 In some cases, the transfer method itself may inform the choice of study cases to include. Meta-analysis approaches (discussed below) can facilitate some forms of statistical validity testing. For example, Moeltner (2015, 2019) uses Bayesian methods in a meta-analysis to identify optimal pooling of studies.

66 OMB Circular A-4 provides other guidance for benefit transfer. Analysts can also consider whether a function transfer that includes the adding-up condition (Text Box 7.6) can account for differing magnitudes across policy cases and study cases.

single estimated value from a single case study, but it can also be the average of a small number of estimates from a few case studies. For example, a study may have found a WTP of $20 per household for a one-unit increase on a water quality scale. A simple unit value transfer would estimate total benefits for the policy case by multiplying $20 by the number of units by which the policy is expected to increase water quality and by the number of households who will benefit from the change. This approach can be useful for developing preliminary, order-of-magnitude estimates of benefits, but it should be possible to base final benefit estimates on more information than a single point estimate from a single study. If multiple studies are available, the mean or median WTP value may provide a useful point estimate, though analysts should consider weighting estimates by inverse variance or sample size to give more weight to more precise estimates when calculating the mean (Nelson and Kennedy 2009; Nelson et al. 2013). Point estimates reported in study cases are typically functions of several variables, and simply transferring a summary estimate without controlling for differences among these variables can yield inaccurate results. Therefore, unit values are often adjusted to account for these differences, e.g., changes in income over time (Boardman et al. 2018). It is important to recognize that unit value transfer assumes that the original good, as well as the characteristics and tastes of the population of beneficiaries, are the same as the policy good.

- ***Function transfers*** use information on other factors that influence WTP to adjust the unit value for quantifiable differences between the study case and the policy case. This is accomplished by transferring the estimated function upon which the value estimate in the study case is based to the policy case. This approach implicitly assumes that the population of beneficiaries to which the values are being transferred has potentially different characteristics but similar tastes as the original one and allows the analyst to adjust for these different characteristics. Generally, benefit function transfers may be preferable to unit value transfers as they incorporate information relevant to the policy scenario (OMB 2023; Johnston and Rosenberger 2010).

  To implement a function transfer, 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 will result. Feather and Hellerstein (1997) provide an example of a function transfer that attempts to correct for such bias. Although function transfers can adjust and compensate for small differences between the case and policy study populations, they are subject to the same basic usage rules governing unit value transfers. Function transfers should only be used if the case and policy studies are evaluating sufficiently similar environmental goods, change in environmental levels, and affected populations.

- ***Meta-analysis*** uses results from multiple valuation estimates in a new unit or function transfer.[67] Meta-analysis is an umbrella term for a suite of techniques that synthesize the results of empirical research. This could include a simple ranking of results, a meta-analytic average or other central tendency estimate, or a multivariate regression. The advantage of these meta-analytic methods is that they incorporate and can potentially control for more information than transfers based on a single estimate. This approach has been widely used in environmental economics (see Rolfe et al. 2015, Johnston et al. 2018, Guignet et al. 2022, and Newbold et al. 2018a).

  There are several guidelines for meta-analyses that outline protocols that should be followed in conducting or evaluating a study. See Bergstrom and Taylor (2006); Nelson and Kennedy (2009); Nelson et al. (2013); Nelson (2015): and Boyle et al. (2013, 2015) for more information. Some may choose to follow a systematic review protocol as described in Text Box 7.5. The EPA's Peer Review Handbook (U.S. EPA 2015b) recommends that meta-analyses used in regulatory analysis should generally be peer-reviewed. Boyle and Wooldridge (2018) emphasize that the purpose of a meta-analysis for benefit transfer is prediction and the purpose of a traditional meta-analysis is to summarize a literature. This latter paper provides a number of technical suggestions to "provide the best econometric prediction of value for a benefit-transfer application."

- ***Structural benefit transfer*** involves deriving a benefit transfer function from an assumed form of the direct or indirect utility function and calibrating or estimating the form of the transfer function using insights from economic theory.[68] The advantages of structural transfer functions are that they can accommodate different types of economic value measures (e.g., WTP, WTA or consumer surplus) and can be constructed to satisfy certain theoretical consistency conditions (e.g., WTP bounded by income). Using a structural benefit transfer or preference calibration approach is one way to ensure that that the adding-up condition holds (see Text Box 7.6). However, Johnston et al. (2018) discuss the tradeoff between theory and accuracy of the transfer in structural benefit transfer. They conclude that core concepts such as diminishing marginal utility are necessary but there can be a trade-off between empirical accuracy of transfers and imposing a specific functional form to satisfy stronger theoretical restrictions; and there is not a consensus in the empirical literature on the appropriate balance.

67 A typical meta-analysis combines estimates from many studies, but meta-analyses that combine multiple estimates from one study or more than one application of the same protocol are also common. This latter type is often referred to as an internal meta-analysis. See Text Box 7.5 for an example of a study that applied an internal meta-analysis.

68 See Smith and Pattanayak (2002) and Smith, Pattanayak, and van Houtven (2006) for descriptions on the method. See Newbold et al. (2018b), discussed in Text Box 7.6, for an example of a functional form of a meta-analysis being based on theory.

Text Box 7.5 - Systematic Review Approaches and Benefit Transfer

EPA risk assessments have increasingly adopted systematic review approaches as recommended by the National Research Council (NRC) (NRC 2014), and these approaches may provide a useful model for identifying and evaluating literature for a benefit transfer or meta-analysis. The Institute of Medicine (IOM) has defined systematic review as “a scientific investigation that focuses on a specific question and uses explicit, prespecified scientific methods to identify, select, assess, and summarize the findings of similar but separate studies” (IOM 2011). A key element in conducting a systematic review is preparation of a protocol which details in advance the methods that will be used in conducting the review. Major advantages of systematic review include improved documentation and transparency, as well as minimization of potential bias in how the review is conducted (NRC 2014).

The steps in conducting a systematic review, as outlined by the NRC (2014), are:

- Problem Formulation: define the study question (roughly equivalent to “describe the policy case”);
- Develop a protocol for conducting the systematic review: the protocol defines the methods to be used (e.g., search strategy, inclusion/exclusion criteria study evaluation criteria);
- Evidence Identification: conduct the literature search and screen the literature search results (apply the search strategy and the inclusion/exclusion criteria from the protocol to identify relevant studies);
- Evidence Evaluation: evaluate quality of studies by applying the criteria specified in the protocol to each included study; and
- Evidence Integration: develop conclusions from the included studies to answer the study question.

- ***Which benefit transfer method to choose*** is not always obvious. Boyle et al. (2013) note there is no consensus on which method works best. There have been numerous studies comparing the (convergent) validity and reliability of transfers (see Rosenberger 2015 and Kaul et al. 2013 for summaries). Some general lessons are "that function transfers tend to be more accurate than value transfers; transfers of values for environmental quantity changes tend to be more accurate than those for quality changes; geographic similarity between sites improves the accuracy of transfers, especially for value transfers; combining information from multiple studies improves the accuracy of transfers; and that transfers based on stated preference valuation formats with more options per question, such as choice experiments, have larger transfer errors than methods with fewer choices per question, such as contingent valuation surveys" (Newbold et al. 2018a). Few studies test the validity and reliability of meta-analytic transfers, however. Johnston et al. (2018) describe how increasingly complex methods may not always be worthwhile, noting that more flexible transfer functions tend to outperform unit value transfers but value transfers may outperform other types of transfers when the sites are very similar. Function transfers that adjust estimates for a few key variables (e.g., income elasticity) may have lower transfer error than complex function transfers that control for numerous characteristics. The benefit transfer literature is large and diverse. The EPA will continue to monitor it and update these recommendations as necessary.

Text Box 7.6 - The Adding-Up Condition in Benefit Transfer

When a benefit transfer function is estimated in a meta-analysis or in some other fashion, it is important to carefully consider the form of the estimating equation used to relate study characteristics to willingness-to-pay. For consistency in regulatory accounting, a willingness-to-pay function must satisfy a basic adding-up condition (Kling and Phaneuf 2018). In other words, WTP for good X, plus WTP for good Y given good X, must equal WTP for X and Y together.

A benefit transfer function that violates the adding-up condition can lead to inconsistent policy evaluations. For example, an omnibus policy that appears to have lower net benefits than a set of component policies that, when combined, yield the same water quality outcomes and have the same total cost as the omnibus policy does not satisfy the adding up condition. In this case, the policy change appears to have larger benefits if it is broken up into several smaller policy changes. A benefit transfer function that violates the adding-up condition also could lead to inconsistent policy rankings, since independently evaluating the provision of goods X and Y could pass a benefit-cost test while evaluating the provision of both goods X and Y together could fail a benefit-cost test.

Analysts who use meta-analysis to estimate a benefit transfer function or apply a benefit transfer function developed in a previous study should ensure that the resulting willingness-to-pay function satisfies the adding-up condition. If the function fails to satisfy the adding-up condition, the analyst should consider re-estimating the benefit transfer function using a different functional form that does satisfy the adding-up condition. One way to ensure that a benefit transfer function complies with the adding-up condition is to use a "structural benefit transfer" or "preference calibration" approach, as described in the main text.

Newbold et al. (2018b) examine existing valuation studies and document violations of the adding-up condition and impacts on benefit-cost results because of these violations. They further describe a structural meta-analytic model that meets the adding-up condition and compare it to a non-structural model that does not. They find that the nonstructural model produces much larger benefits estimates than the structural model and that the violations of the adding-up condition are severe in the non-structural model.

Step 4: Report the Results

In addition to reporting the final benefit estimates from the transfer exercise, the analyst should clearly describe all key judgments and assumptions, including the criteria used to select study cases and the choice of the transfer approach. The uncertainty in the final benefit estimate should be quantified and reported when possible. Any limitations should also be discussed[69] (see Chapter 11 on Presentation of Analysis and Results).

7.5 Accommodating Benefits that Cannot Be Quantified and/or Monetized

It often will not be possible to quantify and value every significant benefit or endpoint for all policy options. For example, it often is not possible to quantify the various ecosystem changes that may

69 See Stanley et al. (2013) for additional recommendations for reporting on meta-analyses.

result from an environmental policy. While Chapter 11 discusses how to present these benefits to provide a fuller accounting of all effects, this section discusses what analysts can do to incorporate these endpoints more fully into the analysis.

7.5.1 Qualitative Discussions

When there are potentially important effects that cannot be quantified, the analyst should include a qualitative discussion of the benefit endpoints. The discussion should explain why a quantitative analysis was not possible and the reasons for believing that these non-quantified effects may be important for decision making. Chapter 11 discusses how to describe benefit categories that are quantified in physical terms but not monetized.

7.5.2 Alternative Analytical Approaches

Alternative analyses exist that can support benefits valuation when robust value estimates and/or risk estimates are lacking. These analyses, including cost-effectiveness, break-even and bounding analysis, can provide decision makers with some useful information. However, analysts should remember that because these alternatives do not estimate the net benefits of a policy or regulation, they fall short of BCA in their ability to identify an economically efficient policy. This shortcoming and any others should be discussed when presenting results from these analyses to decision makers.

7.5.2.1 Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) is most useful when outcome measures from a policy are not in dollar terms, for example, the number of expected premature mortalities avoided, or "lives saved." Cost-effectiveness is calculated by dividing the annualized cost of the option by the annual non-monetary outcome measure, resulting in a ratio of cost per unit (e.g., dollar per life saved). Because the outcome is a ratio, it is more sensitive to how benefits and costs are characterized. Whereas a net benefits outcome (benefits minus costs) is robust to whether negative benefits are counted as costs, the same is not true for a ratio. This is one reason net benefits are generally preferred to benefit-cost ratios.

The economically preferred option from CEA is clear when all options achieve the same result (e.g., the same number of tons reduced): the option with the lowest cost per unit is the most cost-effective option. More typically, however, options vary not only in costs but also in the outcomes they produce, and there is no generally accepted criterion defining the economically preferred option in this case. Each cost-effectiveness ratio represents a different trade-off between the outcome measure and costs, but there is no information on which, if any, of the options is efficient. Still, cost-effectiveness on outcomes can inform decision-making in the absence of monetized benefits.

Because cost-effectiveness is defined by cost-per-unit, CEA requires a single outcome measure. It is not possible to perform a CEA where there are two separate outcomes, which is often the case for environmental regulations. For example, if a program reduces both hydrocarbon and nitrogen oxide emissions, it is probably not possible to develop a cost per ton of hydrocarbons reduced and a separate cost per ton of nitrogen oxides, because the same costs produce both outcomes. For health and safety regulations, however, there are a number of measures that integrate disparate health outcomes into a single metric for cost-effectiveness calculations. These metrics were largely developed for comparing public health or medical interventions. The most common metric is a quality-adjusted life year (QALY), which combines health-related quality of life with longevity. Cost-effectiveness using QALYs is sometimes referred to as "cost-utility analysis" (CUA) because the health-related quality of life component is based on stated preferences about the impact of different health conditions.

The application of QALYs to regulatory analysis has been evaluated in detail by the IOM (IOM 2006). It is important that cost-effectiveness analysis using QALYs be distinct from BCA. Converting QALYs to a monetary value using a "cost per QALY" is not fully consistent with utility theory underlying BCA (IOM 2006; Hammitt 2002), but is an approach suggested for consideration in OMB Circular A-4 (2023). When there is a BCA, cost-effectiveness analysis should be considered a complement that provides a different perspective on the trade-offs of a regulatory action.

7.5.2.2 Break-Even and Bounding Analysis

Chapter 5 describes several approaches for analyzing and characterizing uncertainty. Two methods that can be particularly useful for benefits analysis with missing information are break-even and bounding analysis. For example, analysts who have per unit estimates of economic value, but lack risk estimates can use break-even analysis to 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.[70] This estimate then can be assessed for plausibility either quantitatively or qualitatively. The same sort of analysis can be performed when analysts lack valuation estimates, producing a break-even value that should again be assessed for credibility and plausibility. Policy makers will need to determine if any break-even value is acceptable or reasonable. Bounding analysis can help when analysts lack value estimates for a particular endpoint. 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.

70 Circular A-4 (OMB 2023) refers to these values as "switch points" in its discussion of sensitivity analysis. Section 5.4.4 on uncertainty analysis also contains related discussions on switch points.

Chapter 7 References

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Chapter 8 - Analyzing Costs

As discussed in Chapter 1, the distinction between what is labeled a benefit or a cost in regulatory analysis is, to some extent, arbitrary so long as the analysis is internally consistent. The compliance cost of a regulatory action might be considered a benefit of a deregulatory action, and vice versa. Likewise, reduced fuel expenditures that accrue to a firm or consumer due to regulation have been described as both negative costs and positive private benefits. On which side of the ledger specific categories fall is somewhat immaterial. What is important is that analysts make every effort to account for all costs and benefits to illuminate key differences across the options under consideration with regards to *net* benefits. As noted above, these *Guidelines* are framed from the perspective of a policy that improves environmental quality, but at a cost. This chapter discusses methods and modeling approaches for estimating a regulation's costs, which are typically reflected in market outcomes, for use in benefit-cost analysis (BCA). For a discussion of methods and modeling approaches for estimating a regulation's benefits, see Chapter 7.

Estimating the costs of regulation involves a series of decisions. The analyst must determine the scope of the analysis to appropriately capture the range of anticipated effects from the policy.[1] Analysts must determine the types of costs that are likely to occur within a specific regulatory context and choose the most defensible way to measure them based on the best available data and methods. Both the scope of the analysis and how costs are measured will affect the choice of economic model.[2] Models vary in their ability to capture certain costs; whether they are static or dynamic; their level of geographic and sectoral detail; and their scope; among others. After selecting one or more economic models, analysts face a series of implementation decisions, such as how to best parameterize the model and how to account for uncertainty.

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1 Several executive and legislative mandates require that different aspects of costs be considered in a regulatory analysis. For instance, Executive Order (EO) 12866 specifies that an assessment of the costs of a regulation should include "any adverse effects on the efficient functioning of the economy and private sector (including productivity, employment, and competitiveness)." The Unfunded Mandates Reform Act (UMRA) of 1995 requires that cost estimates account for indirect and implicit costs on state and local governments. Many of these "costs" are categorized as economic impacts and therefore discussed in Chapter 9.

2 A model is a "simplification of reality that is constructed to gain insights into select attributes of a particular physical, biologic, economic, or social system" (National Research Council (NRC) 2009) and the simplifications necessary to tractably model complex systems will introduce uncertainty.

8.1 The Economics of Social Cost

While estimating the costs of regulation is often portrayed as relatively straightforward — particularly compared to estimating benefits — it must be guided by economic theory. As such, the appropriate measure of cost to use in a BCA is social cost. **Social cost** represents the total burden that a regulation will impose on society, defined as the sum of all opportunity costs incurred as a result of the regulation. These opportunity costs consist of the value to society of the goods and services no longer produced and consumed as resources are reallocated away from other activities towards activities such as pollution abatement. To be complete, an estimate of social cost should include the opportunity costs of current and future consumption and leisure that will be forgone as a result of the regulation (e.g., effects in the future could occur because of effects on capital investment).[3] For example, errors can easily occur if the analyst confuses transfers with costs or ignores pre-existing regulation or taxes in the affected market.

The social cost of a regulation is generally not the same as its effects on gross domestic product (GDP) or other broad measures of economic activity.[4] See Section 8.2.2.1 for more discussion. Likewise, social cost is distinct from but includes the cost of compliance borne by the regulated entity. The **compliance cost** is the private cost that a regulated entity incurs to reduce or prevent pollution to comply with an environmental regulation — for instance, through the installation and operation of pollution abatement equipment.

To estimate social cost, analysts may use one or some combination of compliance cost, partial equilibrium and/or general equilibrium approaches. A compliance cost approach assesses the costs of abatement and other actions taken to comply (e.g., monitoring, testing, reporting and recordkeeping requirements) for the directly regulated sector(s).[5] A partial equilibrium approach models the supply and demand responses of the regulated sector(s) to these compliance costs and may be extended to consider a small number of related sectors (e.g., markets that supply intermediate goods to the regulated sector(s), markets for substitute or complementary products, or markets that supply abatement equipment or other services to comply with requirements). When broader economy-wide impacts are expected due to a regulation, a compliance cost or partial equilibrium approach will miss these impacts. In this case, a general equilibrium approach is needed to more fully estimate social cost. Regardless of the approach taken, it is expected that analysts will need to estimate compliance costs associated with abatement requirements, as they are a necessary input for generating estimates that rely on a partial or general equilibrium approach. Models that utilize each of these three approaches to estimate social cost are discussed in Section 8.3.

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3 For a more detailed treatment of the material in this section, see Pizer and Kopp (2005).

4 GDP is defined as the sum of the value (price times quantity) of all market goods and services produced in the economy and is equal to either Consumption (C) + Investment (I) + Government (G) + (Exports (X) – Imports (M)), or Labor (L) + Capital (K) + Taxes (T).

5 The term direct cost is sometimes used to refer to the costs incurred by regulated entities to comply with the regulation. The term indirect cost is sometimes used to refer to costs incurred in related markets or experienced by consumers or government not subject to the regulation, often transmitted through changes in the prices of goods or services produced by the regulated sector.

8.1.1 Compliance Cost Approach

A compliance cost approach estimates the direct compliance expenditures incurred by regulated entities (e.g., individual emitting units or facilities) when installing and operating abatement technologies or processes to comply with a regulation, conditional on a given level of output. It does not attempt to estimate welfare impacts associated with a change in the amount of production or use of inputs but generally assumes that regulated sources are cost-minimizing in their compliance behavior. Its primary advantage is the ability to generate highly detailed and, when data are available, relatively specific information on compliance options and their associated costs that reflect the heterogeneity of regulated entities. This detailed information can be very useful, as many stakeholders are keenly interested in understanding the anticipated cost of meeting regulatory requirements. Furthermore, reporting detailed assessments of how regulated entities are expected to respond to a regulation can generate useful public comments that further the U.S. Environmental Protection Agency's (EPA's) understanding of available compliance options and their costs.

A compliance cost approach typically does not account for other producer or consumer behavioral changes that may result from a new regulation.[6] However, it can still provide a reasonable estimate of social cost when changes in a regulated sector's outputs and input mix (aside from direct compliance activities) are expected to be minimal. However, when significant changes on the producer or consumer side are expected to occur or profit maximizing firm behavior differs from cost minimizing behavior due to market imperfections, a compliance cost approach may substantially misestimate the social cost of a regulation.[7] Likewise, a compliance cost approach does not capture supply side responses, such as changes in the composition of goods produced by the industry or changes in product quality, and the associated changes in consumer and producer welfare that result. A key question for analysts is whether it is worth expending additional resources to expand beyond a compliance cost approach to capture other potentially substantial costs within the sector itself, related sectors or in the overall economy.

8.1.2 Partial Equilibrium Approach

In contrast to a compliance cost approach, a partial equilibrium approach to cost estimation accounts for market changes in the regulated sector. Market responses to the regulation may include reduced industry output or higher prices as firms pass on some costs directly to consumers. The goal of a partial equilibrium approach is to measure the net change in consumer and producer surplus relative to the pre-regulatory equilibrium.[8]

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6 A compliance cost approach does not imply that the costs are ultimately borne by producers. Rather, these costs may be passed through to consumers (see Chapter 9). The key assumption with a compliance cost approach is that there are no significant changes in markets except for the compliance activity.

7 The degree to which a demand response influences social cost depends on a variety of factors, such as the magnitude of the price change, the price elasticity of demand for output of the regulated sector and the degree of competition in the market. An elasticity is a measure of how responsive a firm or consumer is to a change in price. In the case of demand, it is the percentage change in the quantity of the product that is demanded by consumers divided by the percentage change in the product's price. See Appendix A for more discussion of elasticities.

8 Consumer surplus is the sum of consumers' net benefits — i.e., what they are willing to spend on a good or service over and above market price. Thus, it is the area under the market demand (marginal benefit) curve but

In theory, in the absence of market distortions (e.g., pre-existing taxes), the social cost of a regulation can be assessed with a partial equilibrium approach of the regulated market (Just et al. 2005; Harberger 1964).[9] While a policy may have effects in many other markets, market clearing conditions effectively cancel out these effects with regard to aggregate welfare (Farrow and Rose 2018).[10] Thus, a partial equilibrium approach is sufficient for estimating social cost when the analyst expects that a regulation will result in appreciable changes in market activities, but the effects will be confined primarily to a single market or a small number of markets. The use of a partial equilibrium approach assumes that the effects of the regulation in all other markets, outside of those being modeled, will be minimal. See Appendix A.4.3 for more detailed discussion.

Figures 8.1 and 8.2 illustrate how social cost can be defined in partial equilibrium. Figure 8.1 shows a competitive market before the imposition of an environmental regulation. 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.[11] The sum of these two areas defines the total welfare generated in this market (i.e., the net benefits to society from producing and consuming the good or service represented in this market). For simplicity, total welfare as depicted ignores the negative pollution externality arising in this market, which the regulation is designed to correct.[12]

In this market, the imposition of a new environmental regulation raises firms' production costs. Each unit of output is now more costly to produce because of expenditures incurred to comply. 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 from P_0 to P_1 and to decrease the equilibrium output from Q_0 to Q_1, holding all else constant. As seen by comparing

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above market price. Producer surplus is producers' revenues minus the variable cost of production. Thus, it is the area above the market supply (marginal cost) curve but below market price. See Appendix A.

9 As defined in Chapter 5, market distortions are factors such as pre-existing taxes, externalities, trade barriers, federal, state or local regulations or imperfectly competitive markets that move consumers or firms away from the economically efficient outcome. These factors should be accounted for in the baseline and analyzed when they interact with the policy under consideration.

10 In theory, impacts in undistorted related markets are "pecuniary" and do not need to be included if the social costs have been correctly measured in the primary market, but pecuniary effects are important to consider in inefficient related markets (Boardman et al. 2011). It is also likely that most regulations will result in winners and losers. Economic impact analysis evaluates how different groups are impacted by a regulation. See Chapter 9.

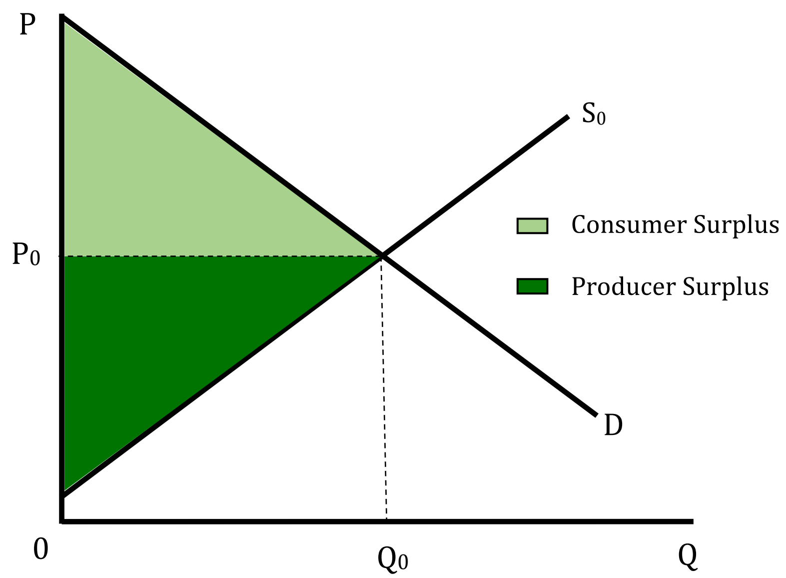
11 Producer surplus may be interpreted as the profits plus the fixed cost of producers. Profits equal total revenues, price multiplied by quantity of total output, minus the total costs that vary with the level of production (i.e., variable costs). These costs equal the area under the supply curve and exclude costs that do not vary with production (i.e., fixed costs). Over time, the share of costs attributable to investments that are fixed declines and the supply curve becomes more elastic. See Section 8.2.3.2.

12 Appendix A presents a graphical representation of how to account for this externality. Reduction of the negative externality would be quantified in the benefits portion of an analysis. The supply curve in Figure 8.1 corresponds to the marginal private cost (MPC) curve described in Figure A.5.

Figures 8.1 and 8.2, the overall effect on welfare is a decline in both producer and consumer surplus.[13]

In the long run (i.e., when all costs are variable), compliance costs in this market equal the area between the old and new supply curves, bounded by the new equilibrium output, Q_1.[14] Useful insights about the total costs of the regulation can be derived from Figures 8.1 and 8.2. First, when consumers are price sensitive — as reflected in the downward sloping demand curve — a higher price causes them to reduce consumption of the good. If costs are estimated ex-ante and this price sensitive behavior is not taken into account (i.e., the cost estimate is based on the original level of output, Q_0), realized compliance costs will be misestimated. Extending the vertical dotted line in Figure 8.2 from the original equilibrium quantity to the new supply curve (S_1) and estimating costs assuming this quantity illustrates this point. A second insight is that realized compliance costs are only part of the total social costs of a regulation. The black triangle shown in Figure 8.2 is an additional, real cost arising from the regulation. It reflects the forgone net benefit (or opportunity cost) from the reduction in output.

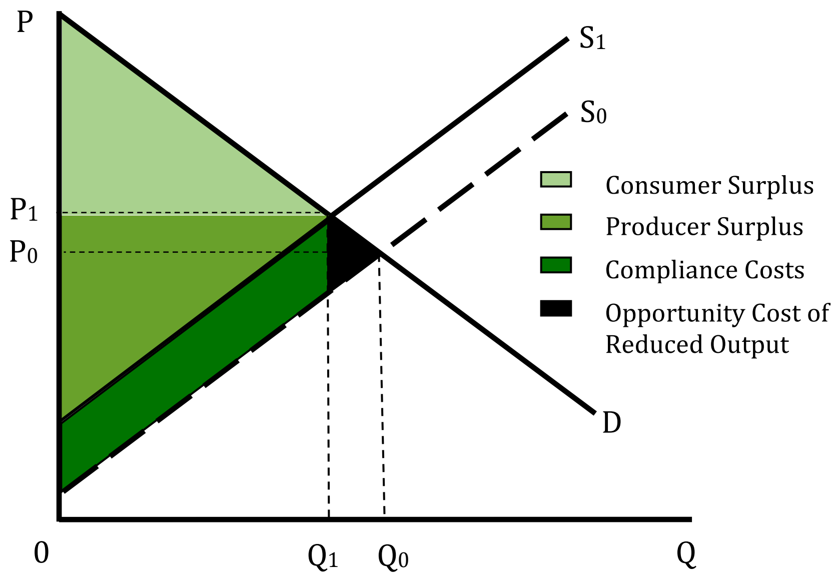
Figure 8.1 - Competitive Market Before Regulation



13 The figure depicts an equal distribution of welfare between consumers and producers in both the old and new equilibria. Depending on the elasticities of supply and demand, this may not be the case. The elasticities determine the magnitude of the price and quantity changes induced by the cost increase as well as the distribution of costs.

14 In the long run, costs are variable and are fully represented in the movement of the supply curve. In the short or medium run, fixed costs may not affect the supply curve although they could contribute to compliance costs. See Tietenberg (2002).

Figure 8.2 - Competitive Market After Regulation



Under the assumption that impacts outside this market are not significant, the social cost of the regulation is equal to the sum of the compliance cost and the opportunity cost of reducing output shown in Figure 8.2. It is exactly equal to the reduction in producer and consumer surplus from the pre-regulation equilibrium shown in Figure 8.1.

When the effects of a regulation are expected to impact a limited number of markets beyond the regulated sector, it still may be sufficient to use a partial equilibrium approach to estimate social cost. A multi-market approach extends a single-market, partial equilibrium representation of the directly regulated sector to include closely related markets. These may include the upstream suppliers of major inputs to the regulated sector (including pollution abatement equipment or services), downstream producers who use the regulated sector's output as an input and producers of substitute or complimentary products. Vertically or horizontally related markets will be affected by changes in the equilibrium price and quantity in the regulated sector. As a consequence, they will experience equilibrium adjustments of their own that can be analyzed in a similar fashion.[15]

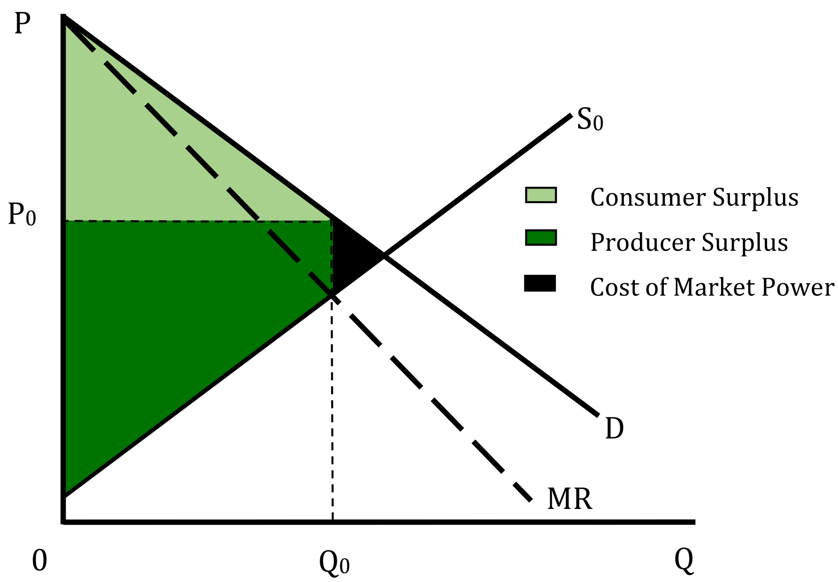
The preceding discussion describes the use of a partial equilibrium approach when the regulated market is perfectly competitive and both producers and consumers are price takers (i.e., their behavior does not meaningfully affect prices). In many cases, however, some form of imperfect competition (e.g., market power) may better characterize the regulated market or closely related markets. Firms in imperfectly competitive markets will adjust differently to the imposition of a new

15 Just et al. (2005) detail methods for evaluating partial equilibrium welfare changes across multiple related markets (see also Bullock 1993). Estimating welfare is only possible when the relevant relationships among the sectors (e.g., cross-price elasticities) are correctly specified. Pizer and Kopp (2005) and Kokoski and Smith (1987) provide additional discussion of when these methods are suitable for estimating social cost.

regulation, which can alter the estimate of social cost.[16] If the regulated markets or closely related markets are imperfectly competitive, this may significantly influence compliance behavior and costs, in which case the market structure should be reflected in the analysis.[17]

Figures 8.3 and 8.4 demonstrate how imperfect competition can result in additional social costs. Figure 8.3 begins with the case of a monopolistic firm. Unlike the case of perfect competition, the price of the good is not set equal to its marginal cost.[18] Firms with market power can instead set price equal to their marginal revenue to maximize profit.[19] However, this results in less of the good being produced (shown as Q_0) than is socially optimal (i.e., what would occur under perfect competition), Q_0^*. The welfare loss from the lower-than-optimal level of production is the black triangle.

Figure 8.3 - Monopoly Market Before Regulation



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16 For further discussion of the welfare effects of environmental regulation in the context of imperfectly competitive markets, see Chapter 6 of Baumol and Oates (1988), Requate (2006), and Chapter 6 of Phaneuf and Requate (2017). See Ryan (2012), Ferris et al. (2014), and Wolverton, et al. (2019) for examples where accounting for the way market structure affected firm decision-making would have potentially led to a different estimate of a regulation's costs. Kellogg and Reguant (2021) provide additional examples in the energy sector.

17 Section 8.2.3.6 describes how environmental regulations may create conditions that lead to imperfectly competitive markets.

18 For expositional purposes we continue to label the marginal cost curve as the supply curve. However, in the case of a monopoly, this curve no longer identifies the amount that will be produced at a given market price (Mankiw 1998). However, as in the case of perfect competition, it still identifies the amount that will be produced given the marginal revenue received.

19 To maximize its profit, the monopolist produces Q0 units such that the additional revenue it receives from selling one more unit of the good (its marginal revenue) equals the additional cost of producing that unit of the good (its marginal cost). MR0 represents the additional revenue that the firm receives for each additional unit of production.

Figure 8.4 - Monopoly Market After Regulation

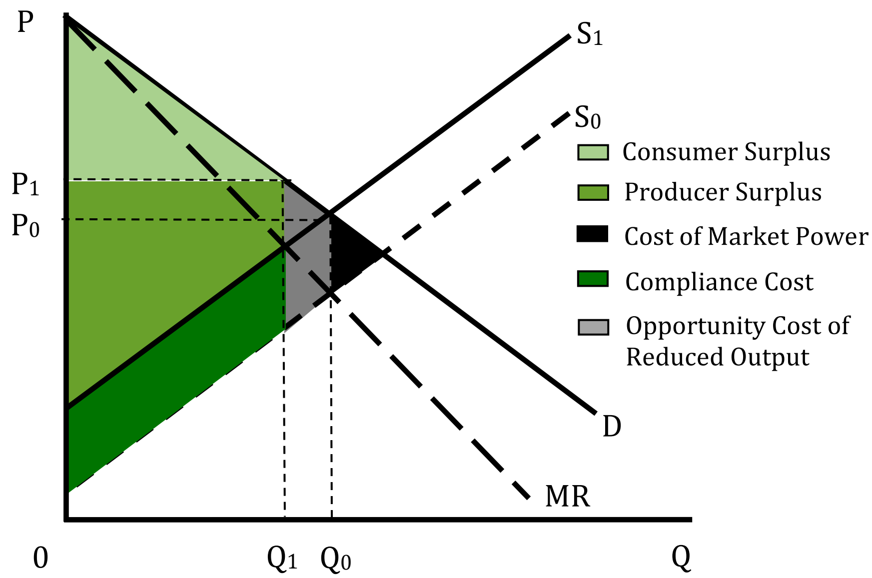

Figure 8.4 shows what occurs when the monopolist is subject to an environmental regulation. As in the case of perfect competition, the regulation causes the supply curve to shift upward from S_0 to S_1. The equilibrium price rises from P_0 to P_1 and output falls from Q_0 to Q_1. By further restricting output, the exercise of market power exacerbates the welfare loss from the regulation. The opportunity cost of this reduced production has now increased. The additional opportunity cost of the reduced output as a result of the regulation is greater than in a perfectly competitive market. Imperfectly competitive input markets may also exacerbate the social cost of a regulation (e.g., Busse and Keohane 2007).

Existing regulations may distort the behavior of regulated sources and other market participants in response to an environmental regulation and, in turn, result in additional social costs. A well-known example is in the context of electric utilities where some states regulate investment and retail prices to assure that producers do not exercise market power. However, this results in two potential distortions. First, retail prices in these states are lower than would occur in a competitive market, which means more electricity is consumed than is economically efficient.[20] Second, the ability to only pass along some types of costs and not others to consumers results in more capital-intensive production (including pollution abatement) than is economically efficient (e.g., Parry 2005; Burtraw and Palmer 2008; Fowlie 2010). Thus, it is important for analysts to be mindful of the potential for interactions with existing market regulations and, when feasible, account for them when modeling compliance behavior of market participants.

20 In some regulated electricity markets, prices are set equal to the average cost of production, which is lower than the additional (marginal) cost of production. In these circumstances, the production and consumption of electricity is greater than is economically efficient because the additional benefit of production exceeds the cost to produce it.

8.1.3 General Equilibrium Approach

A general equilibrium approach to cost estimation concurrently considers the effect of a regulation across all sectors in the economy. It is structured 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 approach 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, 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, will be determined in general equilibrium. 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.[21]

When the imposition of an environmental regulation is expected to have appreciable effects in markets beyond those that are directly subject to the regulation, a partial equilibrium approach may be insufficient to adequately estimate social cost. A general equilibrium approach, which captures linkages between markets across the entire economy, is most likely to add value when both cross-price effects and pre-existing distortions (e.g., taxes, regulations, market power in other markets) are expected to be significant (U.S. EPA 2017).[22,23]

Consider as an example a regulation that imposes emission limits on the electric utility sector. In the long run, we expect at least some, if not all, compliance costs are passed through to consumers as increases in the electricity price. Because electricity is used as an input in the production of many goods, the prices of these products may also increase to reflect the increase in their marginal cost of production. Increases in prices may cause households to alter their choices. For example, their consumption of energy-intensive goods and services may decrease relative to other goods. Furthermore, the number of hours they are willing to work may change in part because when goods become more expensive, households can afford less with the same income; thus, their real wage has declined.[24] On the margin, they respond by changing the number of hours worked. When an environmental regulation affects the real wage such that individuals opt to work fewer hours, it can exacerbate pre-existing tax distortions in the labor market (Goulder et al. 1997). The impacts of a regulation also may interact with pre-existing distortions in other markets, which may cause

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21 Computable general equilibrium (CGE) models are discussed in Section 8.3.3. Hazilla and Kopp (1990), Jorgenson and Wilcoxen (1990), U.S. EPA (1997), U.S. EPA (2011), and Marten et al. (2019) use CGE models to estimate the social cost of environmental regulations.

22 Cross-price effects are measured by elasticities. For example, the cross-price elasticity of demand is defined as the percentage change in the quantity of product X demanded by consumers in response to a change in the price of product Y. If two markets are unrelated, then the cross-price effect is expected to be near or equal to zero. If they are substitutes, then the cross-price elasticity is positive. If they are complements, then it is negative.

23 The previous section shows how the social cost of a regulation can be estimated in a single market using partial equilibrium analysis. The example demonstrates how a regulation may cause an opportunity cost in that market. Distortions with similar opportunity costs 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, which may cause additional impacts on welfare.

24 In general equilibrium analysis, all prices and wages are real, i.e., measured relative to a numéraire, a specific single price or weighted average of prices such as the gross domestic product (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.

additional impacts on welfare.[25] In cases such as these, a general equilibrium approach is capable of identifying the nature and magnitude of the costs of complying with a regulation as they flow through the economy, including changes in substitution among factors of production, trade patterns, endogenous demands and even intertemporal consumption. These effects are partially or wholly missed by compliance cost and partial equilibrium approaches.

Figure 8.5 illustrates how a regulation can interact with pre-existing tax distortions in the labor market. A pre-existing tax equal to a share of the gross (pre-tax) wage (W_g^0) causes the net (after-tax) wage (W_n^0) to be lower than the gross (pre-tax) wage by the amount of the tax. With this tax distortion, the quantity of labor supplied is L_0 and there is an opportunity cost of reduced labor supplied. When a new regulation is imposed in another market, raising production costs will increase the price level and may lower labor supply. This is shown in Figure 8.6 as a decrease in the net wage to W_n^1 and a decrease in the amount of labor supplied to L_1. The opportunity cost of the labor tax along with the increased distortion as the difference between the gross and net wage has increased.[26] The interaction between the effect of a regulation and the distortion from a tax is especially pronounced in the labor market.[27] Similar interactions are likely to occur in other markets with significant pre-existing distortions (e.g., capital markets). In cases where they are likely to have a significant impact, analysts should incorporate these distortions into models used to estimate social cost.[28]

25 See Text Box 8.3 for a discussion of interactions that could also affect benefits estimation.

26 Recall in this example that the tax is a share of the gross wage, so as the gross wage goes up, the distortion from the tax also increases. Alternatively, if the amount of tax does not depend on the gross wage, it may need to be increased to maintain government revenues, which is a common assumption in general equilibrium analysis. However, it is not necessary for the tax level to change for the new regulation to exacerbate the pre-existing tax distortion in the labor market.

27 The labor tax distortion affects individual labor supply decisions at the margin. While full-time workers may not change (or be able to change) hours worked in response to a fall in the real wage, part-time workers — those in households with more than one full-time worker — or potential retirees may be more likely to adjust the number of hours they work. Parry (2003) discusses the theoretical and empirical basis for this depiction of the labor market.

28 Economists have long recognized the "tax interaction effect" (Ballard and Fullerton 1992), and a rich body of work has focused on them in the context of environmental regulation (Goulder, 2000; Parry and Bento 2000; Murray et al. 2005; and Bento and Jacobsen 2007). If an environmental regulation raises revenue through a tax on pollution or another 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, and in some cases, the overall efficiency of the tax system may actually improve. 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. One offsetting factor is that society also incurs a welfare loss from raising revenues through taxes due to the difference between the value of an additional dollar raised by the government and the value of that dollar to a private individual (i.e., the marginal cost of public funds).

Figure 8.5 - Labor Market with Pre-Existing Distortion Before Regulation

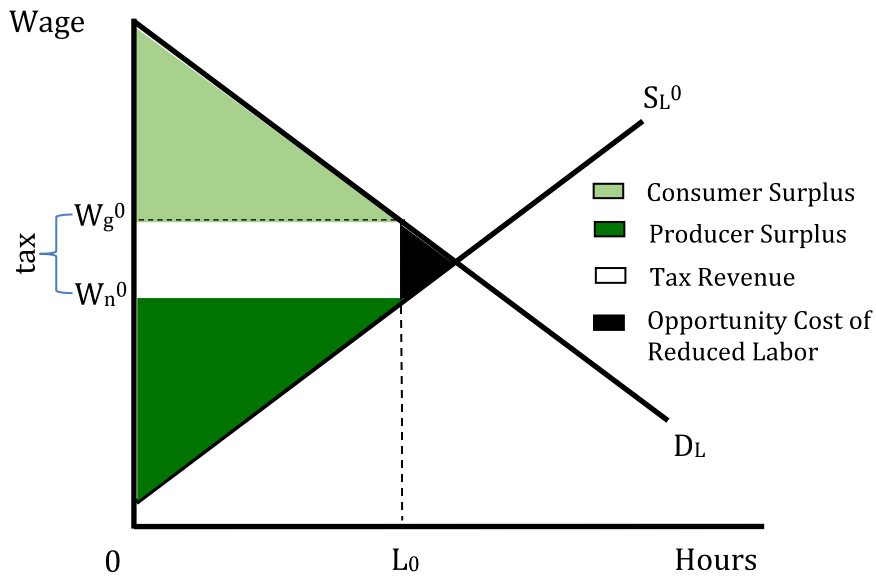
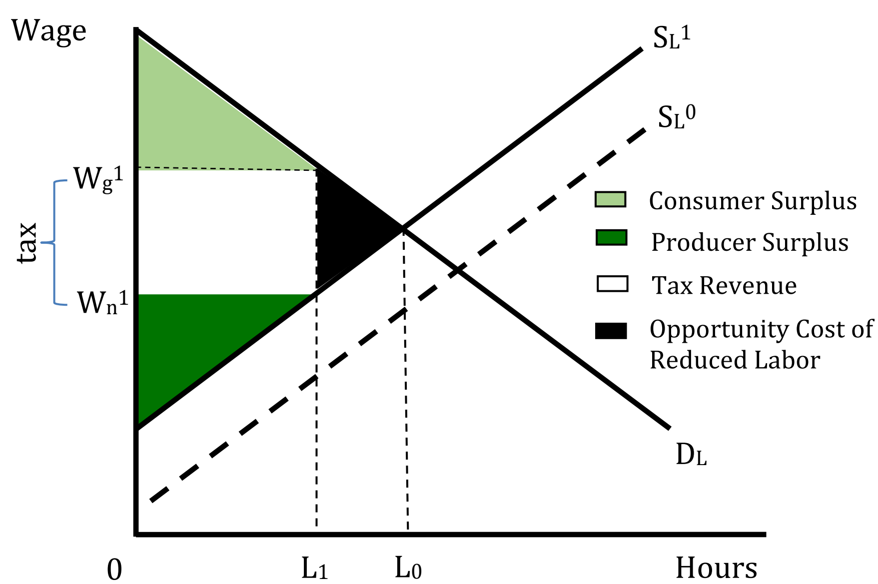


Figure 8.6 - Labor Market with Pre-Existing Distortion After Regulation



8.2 Estimating Social Cost

When estimating social cost, the objective is to measure the incremental cost for each regulatory option under consideration. Incremental cost is defined as the additional cost associated with a new requirement relative to a baseline.[29] Often when specifying a baseline from which to measure costs, the analyst needs to first identify what abatement activities are already in place or anticipated as a result of existing regulations. The costs associated with previously installed abatement controls are not counted toward the cost of the rule, as these occurred prior to the regulation under consideration and are therefore in the baseline.[30] Similarly, costs that have not yet been realized but will be incurred to comply with existing regulations should not be counted towards the cost of the regulation under consideration. Identifying which abatement controls are already in use also aids the analyst in identifying what additional abatement control options may be available to further reduce emissions.[31]

It is important that analysts derive the most defensible central estimates of the compliance costs associated with identified abatement strategies, as they are the building block for developing social cost estimates. Social cost estimates should continue to rely on central assumptions and inputs that are well-supported by standard engineering practice and the published scientific literature. In addition, analysts should ensure that: (1) the information supporting cost estimation is appropriate for its intended use; (2) the scientific and technical procedures, measures, methods and/or models employed to generate the information are reasonable for, and consistent with, the intended application; and (3) the data, assumptions, methods, quality assurance, sponsoring organizations and analyses employed to generate the information are well-documented.[32,33] As previously discussed, analysts are advised to focus quantification on categories of costs that are expected to have a large influence on the net benefits and relative ranking of the options under consideration. Analysts also should describe the process for quantifying the different types of costs that underlie the aggregate cost estimate and present them in a disaggregated and informative manner.

8.2.1 Compliance Cost Estimation

Recall that compliance costs are the additional costs that regulated entities incur to reduce or prevent pollution to comply with the regulation. There are a variety of different types of compliance costs including but not limited to the following.

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29 While this chapter focuses on the anticipated social costs of regulation, the same approach also applies in a retrospective setting (see Chapter 5).

30 See Chapter 5 for a detailed discussion of establishing the baseline. Other issues relevant to specifying a defensible baseline for cost estimation include, for example, ensuring consistency in key assumptions across costs and benefits; and treatment of anticipatory actions to meet regulatory requirements.

31 Note that the expected abatement strategies that underpin estimates of social costs also affect the expected change in the level and exposure to the environmental contaminant and therefore the benefits of the regulation.

32 At times, EPA uses externally derived (e.g., contractor, industry association or advocacy group) cost estimates for its regulatory analyses. Any cost estimate produced by an external source and used by the EPA in its analyses should meet these criteria.

33 Some statutes require the EPA to choose a regulatory option that is demonstrably affordable. In this case, analysts should continue to rely on the most defensible central estimate of costs. Estimating an upper bound (instead of a central estimate) of the compliance cost associated with the chosen option to demonstrate affordability will bias the net benefits of the regulation downward and/or could result in artificially low levels of stringency.

- **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 on-site or off-site processing of waste for an alternative use.
- **Disposal:** The cost of the final placement, destruction, or disposition of waste after pollution treatment/capture and/or recycling has occurred.
- **Prevention:** The cost of preventing pollution from being generated or contamination from occurring during the production process.

Entities that directly incur compliance costs to meet regulatory requirements may include firms, households, and government agencies. For example, firms normally incur costs to purchase and operate pollution control equipment; households may incur the costs of periodic inspections of pollution control equipment on vehicles; and government agencies may implement, administer, monitor, and enforce a regulation. In the case of product standards, compliance costs include the incremental cost of designing and manufacturing the compliant product relative to the already existing noncompliant product.

It is relatively straightforward to infer a value for compliance costs when an explicit monetary payment (e.g., purchasing pollution abatement equipment) is made. Compliance costs for which monetary values are not readily available are often more difficult to quantify. For example, the value and length of time households spend on vehicle inspections may be uncertain. Guidance on how to value time spent on such activities is discussed in Section 8.2.4. Compliance costs are also more difficult to quantify when, instead of installing abatement equipment, firms modify production processes to prevent emissions. Regardless of the ease with which compliance costs can be estimated or what terminology is used to characterize them, if the compliance activities require resources that are redirected from other activities relative to the baseline, the value of those resources should be accounted for in compliance costs.

A compliance cost estimate reasonably approximates the social cost of a regulation when the value of the resources used for compliance generally reflect their social opportunity cost and prices or other producer and consumer behavior are not expected to change significantly as a result of a regulation. Determining whether compliance activities change prices or behavior requires: (1) estimates of supply and demand conditions, (2) an assessment of how compliance costs affect production costs, and (3) evidence of whether producers in the sector significantly change their level of production relative to one another. When compliance costs are used to estimate social costs, the analysis should provide evidence that justifies their choice.

It is common to refer to different categories of compliance costs, such as fixed or variable costs, as a way to systematically identify the costs that may result from a regulation. In practice, these categories of compliance costs may not be entirely distinct. Table 8.1 summarizes the main cost categories discussed in this and subsequent sections.

8.2.1.1 Fixed Costs

Fixed costs do not change with the level of production or abatement over a specific time period, often referred to as the short run. They are typically one-time costs, or costs that only occur once over the time horizon of the analysis, such as the installation of pollution control equipment. However, fixed costs may also refer to recurring costs that are independent of the level of production or abatement over a given time period (note that in the long run, virtually all fixed costs are variable.) Two common categories of fixed costs are described below.

Capital costs are costs related to the installation or retrofit of structures or equipment. These expenditures include materials and labor used for equipment installation and startup. Once equipment is installed, capital costs generally do not change with the level of abatement or production. Capital costs may also include changes to the production process.

Table 8.1 - Types of Costs Associated with Environmental Policies: Categories, Examples and Commonly Used Approaches to Quantification

Compliance Costs	Examples	Common Approach to Quantification[34]
Fixed costs	Capital costs; Research and Development investments	Compliance cost
Variables costs	Operating costs; Monitoring, reporting and recordkeeping costs; Transaction costs	Compliance cost

Other Opportunity Costs	Examples	Common Approach to Quantification
Reduced output in regulated markets	Higher product prices cause reduced quantity demanded	Partial equilibrium
Changes in product quality	Trade-offs with reliability or longevity	Partial equilibrium
Changes in behavior in the regulated or final goods markets	Changes in investment (e.g., delayed adoption); Rebound effects	Partial equilibrium
Transition costs	Search costs for new jobs; Costs of initially scarce new equipment	Partial equilibrium

Economy-Wide Costs	Examples	Common Approach to Quantification
Interactions with pre-existing distortions in related markets	Tax interaction effect; Trade barriers	General equilibrium
Macroeconomic feedbacks	Capital-induced growth effects	General equilibrium

Research and development (R&D) costs are incurred to develop new products, processes, or techniques. These costs are in addition to capital costs and should be accounted for when estimating compliance costs. Similarly, if a supplier to the regulated entity is expected to incur R&D expenditures in response to the regulation those costs should also be included in the estimate of the social costs.

In the case where a supplier incurs the R&D expense, care should be taken to avoid double-counting; in the long run, the supplier will reflect these R&D costs in the price charged to the

34 While opportunity and economy-wide costs may be quantified without explicit partial or general equilibrium modeling, such modeling may be necessary to avoid the double-counting that would occur if these costs were added directly to cost estimates from a compliance cost approach.

regulated entity. If social costs are estimated using the prices regulated entities pay, accounting for these additional R&D investments by the supplier, or any other resources reflected in prices charged to the regulated entity, do not need to be added to the estimate of social costs.

R&D costs incurred in the past by a regulated entity or supplier should not be counted as a cost of the regulation, as these costs are sunk.[35, 36] If past R&D costs are reflected in the market prices of inputs sold by the supplier (for instance, due to market power) then the cost associated with these past R&D expenditures should also be excluded from the social cost estimate when possible.

8.2.1.2 Variable Costs

Variable costs change with the level of production or abatement.[37] They are the sum of the marginal cost for each unit that is produced. Common categories of variables costs are described below.

Operating costs are recurring expenditures associated with the operation and maintenance of equipment, including salaries and wages, energy inputs, materials, and supplies, purchased services and maintenance or repair of equipment associated with pollution abatement or waste management. In general, operating costs increase with the level of abatement or the amount of production or use.

Monitoring, reporting, and recordkeeping costs are incurred to demonstrate or assure compliance with a regulation. They may be incurred by regulated entities or regulators and generally reflect the use of resources that should be accounted for when estimating the cost of a regulation. While these types of costs are identified here as variable costs, some may also be a fixed cost, such as the installation of pollution monitoring equipment.

Transactions costs are the costs incurred when buying or selling a good or service. They may include the costs of searching out a buyer or seller, bargaining and enforcing contracts. Transaction costs reflect the use of real resources (e.g., time, equipment) and should be included in an estimate of social costs.

8.2.2 Social Cost Estimation

Social costs most often differ from compliance costs because the imposition of the regulation causes changes in behavior beyond just the compliance activity required by the regulation. The most straightforward example of an opportunity cost not accounted for by a compliance cost approach is the value to producers and consumers of **reduced output in the regulated market** as demand

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35 "Sunk costs" are costs that have already been incurred and cannot be reversed (e.g., existing investments in pollution control equipment or previous use of labor). For a deregulatory action, those costs that are sunk should not be accounted for in the benefits of that action while avoided future compliance costs should be accounted for. Care should be taken when identifying whether certain costs are sunk as certain investments and other fixed costs may actually be, in part, reversible, in which case there is an opportunity cost of continuing to use those resources. For example, there may be scrappage value of pollution abatement equipment.

36 Note, however, that anticipatory actions — such as planning and designing for future R&D activities — that are initiated in expectation of promulgation (for instance, in response to the proposal) may still be attributable to the regulation.

37 Use of some resources, especially energy, also can cause negative environmental or other externalities. Techniques for non-market valuation can be applied even when impacts are counted on the cost side of the ledger in a benefit-cost analysis (BCA) (see Chapter 7).

responds to higher product prices. However, social costs can differ from compliance costs even without additional behavioral responses. For example, differences may occur when environmental regulations apply to the performance of a final good — for instance, a vehicle or cleaning product — and the installation of an abatement technology can sometimes also result in **changes in other product attributes** valued by consumers. While these changes may be positive or negative, examples in the literature mainly focus on trade-offs between emissions and attributes such as decreased performance or reliability or reduced safety due to material substitution (e.g., Klemick et al. 2015; Klier and Linn 2016).

Other behavioral changes in the regulated or final good markets are also not captured by a compliance cost approach. For example, the design of the regulation itself may result in changes in investment behavior to avoid the costs of new requirements. In particular, vintage-differentiated regulations that impose more stringent abatement requirements on new emission sources can result in delays in the adoption of new, cleaner equipment and potentially increase investment in older, dirtier equipment (e.g., Gruenspecht 1982; Nelson et al. 1993; Jacobsen and Bentham 2015). While this behavior lowers the cost of complying with the regulation, it also weakens its overall stringency. In other cases, the design of the regulation can lead to changes in the utilization of the regulated product. For example, the literature has estimated a rebound effect from energy efficiency and vehicle fuel economy standards; because these regulations make it cheaper to consume energy or fuel on a per-unit basis, demand for these services and therefore emissions from them increase relative to the case without the rebound (Gillingham and Rapson 2016).[38]

As already discussed, economy-wide costs also may arise when the regulation interacts with **pre-existing distortions** such as taxes, other regulations, trade barriers or market power to move private behavior further away from the economically efficient outcome.

When compliance costs do not fully represent all the opportunity costs of a regulation, partial or general equilibrium analytic approaches can be used to estimate social cost. In some cases, the analyst can construct or use available partial or general equilibrium modeling tools to credibly estimate expected changes in the social cost of regulation. When a model is not available, it may still be possible to estimate the social cost associated with anticipated behavioral change by applying findings from the peer-reviewed literature. Analysts should justify their choice of estimates and explain their applicability to the specific context of the rule. If a range of credible estimates are available, analysts should reflect that range in the analysis and discuss key factors or sources of uncertainty that influence the estimates. Regardless of the origin of the estimate — be it modeled, empirically estimated, or taken from published studies — it is also important that the underlying behavioral assumptions are consistent with the rest of the BCA.

8.2.2.1 Measuring Social Cost

It is possible to estimate the social cost of a regulation by adding up the net change in consumer and producer surplus in all affected markets. Consumer's equivalent variation (EV) and compensating

38 As Section 5.2 mentions, behavioral economics can have implications for benefits and costs of a regulation. For example, if consumers mis-optimize or are loss averse, they may not adopt energy-saving technologies for which private benefits of adoption appear to exceed private costs. This raises the possibility that a regulation could yield positive private net benefits to consumers or firms. See Section 7.2 for more discussion.

variation (CV) are other measures that have been utilized.[39] As households are the ultimate beneficiaries of government and investment expenditures, the EV and CV measures focus on changes in consumer welfare rather than on changes in demand.[40]

The social cost of a regulation is generally not the same as a change in GDP or aggregate consumption (U.S. EPA 2017). As measures of social cost, changes in GDP and aggregate consumption both miss potentially important regulatory effects, such as impacts on leisure demand or the demand for nonmarket goods.[41] GDP is also comprised of more than just changes in consumption as it is a measure of total economic output.[42] For instance, a regulation that requires firms to install new capital in a given year will see an increase in investment. However, capital also affects the availability of goods and services that can be consumed over a much longer time period. As a result, GDP effectively double counts the new capital installed since investment and consumption are both components of GDP.[43]

8.2.2.2 Transfers

Environmental regulations may also affect **transfers**. Transfers are shifts in money or resources from one part of the economy (e.g., a group of individuals, firms, or institutions) to another in a way that does not affect the total resources that are available to society. In other words, the loss to one part of the economy is exactly offset by the gain to another. Since social cost represents the total burden that a regulation imposes on the economy, it nets out transfers.[44, 45] Examples of transfers

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39 Both EV and CV are monetary measures of the change in household utility brought about by changes in prices and incomes resulting from the imposition of a regulation. Appendix A describes the relationship between consumer surplus, equivalent variation and compensating variation. EV and CV are particularly well-suited for partial and general equilibrium analysis because both modeling frameworks require the explicit characterization of consumer preferences. Calculating EV and CV requires only pre- and post-policy price and utility levels.

40 EV and CV can also provide a complete welfare metric (incorporating both benefits and costs) if non-market goods are explicitly accounted for in consumer utility functions. However, these metrics are often only used to assess the social cost of a regulation because traditional economic models do not yet incorporate non-separable benefits, or explicit linkages between environmental quality and economic costs (see Text Box 8.3).

41 See Paltsev et al. (2009), U.S. EPA (2011), and Paltsev and Capros (2013) for examples of how these measures differ in specific policy contexts.

42 It is also the case that transfer payments, which are excluded from BCA, are subsumed within the government spending category of GDP. In addition, while changes to trade patterns due to a regulation may be reflected in both GDP and welfare, they are not necessarily equivalent measures (Paltsev et al., 2009; Paltsev and Capros 2013).

43 Several reasons exist for why GDP is not the preferred measure of social cost and overall welfare in general. For instance, GDP does not include non-market environmental costs or benefits. An example of where it is a potentially misleading metric is when improvements in environmental quality from a regulation also lead to reductions in hospital visits that reduce GDP. GDP is also a flow measure of expenditures and does not account for changes in the capital stock. An example of when this is potentially important is if pollution damages buildings, then the expenditures on maintenance and repair would increase GDP at the expense of returning the stock of capital to its original state.

44 Transfers are important for understanding how a regulation affects the private cost of a regulation for different groups. Thus, they are included in an economic impact analysis. See Chapter 9.

45 An exception is when one group has economic standing in the analysis, and the other does not. See Chapter 5.

include payments for most taxes and subsidies received, as well as higher revenues for producers in imperfectly competitive markets due to higher prices.[46] However, it is important to note that not all taxes or subsidies should necessarily be excluded from estimates of social costs under the assumption that they are transfers. For example, the opportunity cost of a firm employing labor in a compliance activity is inclusive of payroll taxes since they are a form of compensation (e.g., insurance against old age) and not a transfer.

While transfers should be excluded from an estimate of social cost, the conditions leading to the transfer may create additional costs that should be accounted for in a partial or general equilibrium framework. For example, when existing taxes are already distorting behavior in a socially inefficient manner — for instance, by changing the decision of how much to work — the change in behavior induced by the regulation can cause the welfare loss associated with these distortions to also change. These additional changes in welfare due to interactions between the environmental policy and pre-existing tax distortion should be included in an estimate of social costs.

8.2.3 Evaluating Costs Over Time

After the imposition of a new environmental regulation, the economy moves to a new long-run equilibrium set of prices and quantities that allow all markets to clear. Since compliance costs represent permanent additions to the cost of production for a firm, effects in closely related sectors are incurred in the new equilibrium.

However, in some contexts it is possible that firms and/or consumers may incur additional short-term costs during the period when the economy is adjusting to the new equilibrium. These are known as **transition costs**. Examples include costs to train workers to use new equipment, search costs as some workers seek employment in other sectors, and additional costs associated with initially limited availability of new monitoring or abatement equipment. It is also possible that at least some factors of production are fixed initially, limiting the ability of firms to respond quickly to new regulatory requirements. For instance, contractual or technological constraints may prevent firms from fully adjusting their input mix or output decisions until those contracts expire or technology is ready to be replaced. Similarly, the number of firms may change over time depending on the cost of new firms to enter. If these types of adjustment costs are substantial, a sole focus on long run costs may underestimate the total social cost of regulation.

Thus, it is important to consider both short- and long-run effects when measuring costs over time. In addition, analysts must make choices about the time horizon of the analysis, the use of a static versus a dynamic framework, discounting and technical change, employment effects and effects on market structure.

8.2.3.1 Time Horizon

The time horizon for calculating producer and consumer adjustments to a new regulation should be considered carefully. The analyst should strive to estimate the present value of all future costs of a regulation (see Chapter 6). If the analyst is only able to estimate a regulation's costs for one or a few representative future years, the analyst must take care to ensure that the year(s) selected are truly

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46 For example, taxes are generally thought of as transfers between households or firms and government such that an offsetting change in government revenue and household income due changes in behavior induced by the regulation are not social costs. Regulations may also create scarce compliance assets, such as allowances in cap-and-trade systems. Generally, the gratis receipt of or any payments for allowances are also a transfer (see, for example, Burtraw and Evans 2009).

representative, that no important transitional costs are effectively dismissed by assumption and that no one-time costs are assumed to be ongoing.

In the short run, at least some factors of production and consumer demand are fixed. If costs are evaluated over a short period of time, then contractual or technological constraints can prevent firms from responding quickly to increased compliance costs by adjusting their input mix or output decisions. In the long run, by contrast, all factors of production are variable. Firms can adjust any of their factors of production in response to a new regulation and can even change their production processes. Similarly, consumers, including producers in other sectors, may not be able to adjust demand for the output of the regulated sector in the short run, but have more flexibility in the long run. The time horizon for the analysis should be long enough to capture any flexibility the regulation provides firms in their compliance approach.

However, if transition costs seem likely, analysts should also consider presenting evidence that sheds light on the length of the transition period and the magnitude of these costs. In some cases, regulatory requirements are phased in gradually over time, either explicitly through graduated compliance dates or requirements or implicitly through characteristics like vintage differentiation (i.e., varying regulatory requirements based on the age of the plant). For example, consider a regulation that enacts more stringent requirements on new sources of a pollutant. Selecting a time period of analysis that is early in the program when only a few new sources of production are affected may not accurately capture a future year in which most sources of production are new for the purposes of the regulation. A regulation also may influence the rate at which old sources are replaced by new ones. Chapter 5 contains additional guidance on how to determine the most appropriate time horizon for analysis.

8.2.3.2 Dynamics

One key decision for the analyst is whether to assume that economic conditions are invariant over time (i.e., static) or attempt to account for expected future changes in prices and economic activity (i.e., dynamic). Costs that are estimated at a given point in time or for a selection of distinct points in time and compared to the baseline are static. They provide snapshots of costs faced by firms, government and households but do not allow behavioral changes from one time period to affect responses in another time period. A dynamic framework, one that explicitly captures trade-offs across time periods, allows for this possibility.[47]

In most cases, a regulation will continue to have economic impacts after its initial implementation. If these intertemporal impacts are likely to be significant, they should be included in the estimation of social cost. Pizer and Kopp (2005) note that static productivity losses from environmental regulations are amplified over time due to their effect on capital accumulation (a lower capital stock over time reduces economic output and therefore welfare). A static model would miss this effect. In some cases, the potential effect of a regulation on long-term growth may be significantly larger than its effect on the regulated sector alone.[48] In addition to these **capital-induced growth effects**, the

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47 Note that a comparative static framework compares snapshots of key economic outcomes before and after a change in an exogenous factor such as a regulatory requirement.

48 Pizer and Kopp (2005) estimated that the "additional cost of this accumulation effect on welfare can be as much as 40 percent above the static cost that ignores changes in capital stock." Hazilla and Kopp (1990) and Jorgenson and Wilcoxen (1990) also showed that this effect is potentially significant. It is important to note, however, that this conclusion is based on studies of large-scale changes in environmental regulation (i.e., the welfare effects of the 1972 Clean Water Act and 1977 Clean Air Act Amendment).

evaluation of costs in a dynamic framework may be important when a proposed regulation is expected to affect product quality, productivity, innovation and/or changes in markets indirectly affected by the environmental policy. Dynamic effects also impact net levels of measured consumer and producer surplus over time. See Section 8.4 for how a regulation's potential dynamic impacts affect model choice.

Conceptually, a dynamic framework allows the analyst to specify the process by which the economy moves between equilibria in response to a regulation across time. In practice, however, economists have more experience characterizing long-run equilibria than the pathways between them. While shorter run equilibria can be approximated by treating some factors of production as fixed (e.g., labor or capital), very near-term transitional costs are typically ignored in the modeling approaches discussed in Section 8.3. [49]

8.2.3.3 Discounting

Costs that occur over time must be properly and consistently discounted to allow for legitimate comparisons with benefits.[50] Procedures for social discounting in economic analyses are reviewed in considerable detail in Chapter 6.

There are two applications of discounting that are closely related to the modeling of social costs. First, when modeling firms' behavior, the analyst should use a discount rate that reflects the industry's cost of capital, just as a firm would. The social cost of the regulation, on the other hand, is calculated using the social discount rate, the same discount rate used for estimating the benefits of the regulation. Section 6.4 provides additional details on the choice of discount rate when modeling behavior, such as firms' compliance decisions. Second, when a dynamic general equilibrium model is used to estimate social costs, any displacement of investment due to the regulation has already been accounted for and the social cost estimates should only be compared to present value estimates of benefits discounted at the consumption discount rate.

8.2.3.4 Technical Change and Learning

Estimating the social cost of an environmental regulation over a relatively long time horizon requires assumptions about future technological change. Jaffe et al. (2002) lay out a conceptual framework for understanding how technological change in response to environmental regulation may affect the relationship between inputs and output, ultimately reducing the unit costs of production. It is possible for an environmental regulation to change overall productivity in a sector over time in one of two ways: 1) the sector is more productive than before, but inputs are used in the same proportion as before to produce output (i.e., unbiased technical change) or 2) the regulation affects the growth rate of one or more inputs over time in a way that changes the relative productivity of the inputs to production (i.e., biased technical change).

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49 Dynamic stochastic general equilibrium (DSGE) models represent business cycles within an economywide framework via random autocorrelated productivity shocks. While built on rely on highly aggregate representations of the economy, they may offer general insights into the role of modeling uncertainty and how regulations interact with short-run dynamics (Annicchiarico et al. 2021, U.S. EPA 2017).

50 It is equally important to properly discount cost estimates of different regulatory approaches to facilitate valid comparisons.

Compliance with an environmental regulation may result in the adoption of existing technology, improvement, or application of existing technology to a new use and/or development of entirely new technologies or processes (Sue Wing 2006). Whether it is more appropriate to capture these compliance responses as affecting overall productivity or the relative productivity of one or more inputs is an empirical matter.

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 how compliance is achieved and the relative role of technological innovation (e.g., Fischer et al. 2003).[51] As a result, the same environmental end may be achieved at significantly different costs, depending on the pace and direction of technical change.

The empirical economics literature also has observed that variable costs of production or environmental abatement often decline over time with cumulative experience. Ferioli et al. (2009) note that just the act of deploying a new technology can result in substantial process improvements that translate into cost reductions. Building on this empirical observation, log-linear or S-shaped learning curves that related the scale of production to per unit costs of production have often been used to represent how costs decline over time with experience. However, explanations for why this occurs vary (e.g., workers learn from mistakes and determine shortcuts; ad hoc processes become standardized), and substantial uncertainty remains regarding how learning occurs over time for a specific technology (Yeh and Rubin 2012). For instance, what learning rate is appropriate for a new versus a mature technology? To what extent does the learning rate change over time or remain relatively constant? Do costs always decline or increase in some cases?[52]

The EPA's Advisory Council on Clean Air Compliance Analysis stressed the importance of relying on sector-specific empirical data to inform assumptions regarding learning effects whenever possible.[53] When no data are available or the evidence is outdated, it recommended the use of a single default learning rate for transparency reasons. Sensitivity analysis is also recommended to better understand the influence that learning curves can have on costs (U.S. EPA 2007).[54,55] Given uncertainty regarding how and when learning curves should be applied, analysts also should

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51 For instance, the realized costs of Title IV of the 1990 Clean Air Act Amendment's Sulfur Dioxide (SO2) Allowance Trading program are considerably lower than initial predictions, in part due to incentives to innovate in response to the policy (e.g., Bellas and Lange 2011; Frey 2013; Chan et al. 2018). See Chapter 4 for a discussion of how different regulatory approaches may affect innovation.

52 Yeh and Rubin (2012) point to examples where technologies that were tested on a smaller scale or a controlled setting actually experienced increased costs upon deployment due to unexpected performance or reliability challenges.

53 OMB's Circular A-4 recommends that a cost analysis incorporate credible changes in technology over time, noting evidence from the literature that variable costs of deploying new technologies or existing technologies in new applications decrease over time (OMB 2023).

54 A useful description of the calculations used to identify a learning curve are found in van der Zwaan and Rabl (2004). The U.S EPA (2016b) reviews learning rates in the published literature for manufacturing and electric utilities with a specific focus on the production of transportation-related goods (e.g., cars, ships, trucks). Grubb et al. (2021) report estimated learning rates for energy and related technologies. Note that the empirical estimates in the literature represent a biased sample, since they only represent technology that has been successfully deployed (Sagar and van der Zwaan 2006).

55 Note that cost decreases associated with technological change and learning may have additional costs associated with them such as training costs.

discuss and justify their assumptions. See Section 5.5.3 for additional discussion of technical change and learning.

8.2.3.5 Social Cost and Employment Effects

Recall that compliance costs include the value of labor for activities such as the installation and maintenance of abatement technologies as well as monitoring, recordkeeping, and reporting. The social cost of a regulation also includes the value of lost output associated with the reallocation of resources (including labor) away from production of output and toward pollution abatement, the value of induced changes in consumption and the deadweight loss from changes in the use of time (i.e., due to pre-existing tax distortions in the labor market). Employment effects more generally, such as those driven by labor-leisure choice, are not part of social costs under two commonly held assumptions: if the economy is at full-employment (i.e., every worker who wants a job at the prevailing wage has one) and with *de minimus* transition costs (Ferris and McGartland 2014). Typically, a regulation reallocates workers among economic activities — increasing employment in some industries and decreasing it in others — rather than affect the general employment level (Arrow et. al 1996).[56] For these reasons, employment effects should be characterized in the economic impact analysis, as explained in Chapter 9.

8.2.3.6 Effects on Competitiveness

As discussed in Section 8.1.2, imperfect competition in baseline market conditions may influence the social cost of a regulation. Introducing an environmental regulation may also create conditions that affect the size and market structure of industry, which may then allow firms to exercise market power.[57] Analysts should assess any expected changes to market structure as a result of the regulation and, in particular, whether it will lead to imperfect competition and impact social cost.

Environmental regulations can potentially affect the number of producers and the market structure of the regulated sector by raising production costs, modifying economies of scale, or affecting barriers to entry. For example, spatial heterogeneity in the stringency of environmental regulations or compliance costs, and in turn their effect on production costs, can lead to market consolidation at existing firms (e.g., Gray and Shadbegian 2002). Market structure can also be affected by the impact of compliance activities and abatement technologies on the minimum efficient scale for firms in the industry.[58] Positive economies of scale for abatement technologies can lead to reduced entry and greater exit (Millimet et al. 2009). Similarly, larger firms in the industry may have a competitive advantage in the presence of economies of scale (Dean et al. 2000). Differences in product offerings by firms may also affect market structure. If some firms subject to new product standards already have compliant products, they will have a distinct advantage over others. Regional differences in

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56 This does not mean, of course, that specific individual workers are not harmed by a policy (e.g., if they lose their jobs).

57 The focus of this section is on how these changes may affect market structure, including reducing competition (i.e., increase market power), and consequently affect the social cost of a regulation. These consequences of a regulation may also affect the composition and distribution of costs within a sector and closely related markets. See Chapter 9 for discussion of these market impacts.

58 Note, however, that it is theoretically ambiguous as to whether a reduction in output will be accompanied by a net reduction in the number of firms in a regulated market for common regulatory designs such as performance standards (e.g., Requate 1997; Lahiri and Ono 2007).

regulatory requirements may also lead to product differentiation, which can then create or increase market concentration (e.g. Brown et al. 2008; Chakravorty et al. 2008). Regulations can also create barriers to entry either due to vintage-differentiated standards, whereby new entrants have stricter standards, or through the control of patents on abatement technologies held by incumbents who innovated as a result of the regulation. By decreasing opportunities for entry or the number of firms in a sector, it is possible that incumbent or otherwise advantaged firms may now be able to charge higher prices, which in turn further reduces competition and, all else equal, increases the social cost of the regulation (see Figure 8.3 which shows the additional cost associated with reduced output due to market power).

The effects of imperfect competition on the social cost of regulation may also increase over time as markets adjust. For example, Fowlie et al. (2016) evaluate the social cost of a cap-and-trade program in the cement sector, where firms have significant market power in local markets. They show that, in the short run before production methods and the number of firms adjust, the additional social cost of the regulation due to imperfectly competitive markets is relatively small. However, in the longer run as firms change their production processes and firms exit the market, the effect of imperfect competition on the social cost of the regulation is much higher. It is therefore important that analysts evaluate potential differences between short and long run effects of a regulation on market power.

8.2.4 Valuing Time

Compliance with environmental regulations changes the use of productive resources, including people's time. Often, these changes occur at the workplace where labor is required to undertake pollution control activities. Less often, time outside of the workplace is also affected; for example, product bans might cause consumers to switch to substitutes that occupy more time. Changes in time use can affect the social costs of a regulation. The EPA has produced a separate document on how to value work time and nonwork time in regulatory analyses. We summarize the recommendations below, but analysts should consult U.S. EPA (2020) for a detailed discussion.

The opportunity cost of worktime is determined by the value of the marginal product that would have occurred absent the regulation. As a proxy for this opportunity cost, analysts should use the employer's cost of employing a worker, consisting of the wage, fringe benefits and any overhead costs.[59] The value of work time will vary based on the industries and occupations affected by the regulation. If overhead data are not available, U.S. EPA (2020) recommends that analysts use a default multiplier applied to wages plus fringe benefits for the value of worktime.[60] U.S. EPA (2020) provides links to data sources for wages, benefits and overhead rates that normally are included when valuing worktime.

Non-work time includes time spent on leisure, household production or other unpaid activities. Its opportunity cost may vary by the types of activities forgone; the utility derived from the activity

59 Overhead costs are employer costs associated with labor, but not paid directly to workers, such as the value of personnel services and training activities. For more information on how wages, fringe benefits and overhead costs are defined, see Section 2.1 of U.S. EPA (2020).

60 The default multiplier in U.S. EPA (2020) reflects multiplier values used in prior analyses based on industry Source: U.S. EPA (2020). The Bureau of Labor Statistics (BLS) Occupational Employment Statistics (OES) are available at https://www.bls.gov/oes/tables.htm, while the Employer Costs for Employee Compensation and occupation-specific benefit and overhead rates affected by EPA regulations.

that occupies time; whether workers have a continuous choice over their hours of paid work; the socioeconomic characteristics of affected individuals; and more. As a proxy for the opportunity cost of non-work time, analysts should add the value of voluntary fringe benefits to the wage net of any taxes paid by workers to federal, state and local governments on earned income.[61] Table 8.2 summarizes the recommended approach and data sources for estimating work time and non-work time.

In unusual circumstances, analysts may have access to information that allows an alternative approach to estimating the value of work time or nonwork time. If utilized, analysts should explain why the alternative is preferred to the approach recommended here and in U.S. EPA (2020).

Table 8.2 Estimating the Value of Work and Non-Work Time

Type of time affected	Displaced activity	Estimation approach	Data sources
Work time: Tasks completed while working for pay	Other market work in the same industry and occupation as workers asked to complete the required tasks	Employer costs of labor = Wages + Fringe benefits + Overhead costs	• BLS OES or ECEC data on wages and fringe benefits • For overhead costs, use industry specific data as available • If overhead is not available, use the recommended multiplier to obtain a fully loaded wage
Non-work time: Tasks completed outside of paid work time	Other nonmarket activities such as leisure and nonmarket work	Individual valuation of time = (Wages – Taxes on earned income) + Voluntary fringe benefits	• BLS OES or ECEC data on wages and voluntary benefits • Adjust wage estimates using Census CPS data on median household income before and after taxes to estimate average income tax rate

Source: U.S. EPA (2020). The Bureau of Labor Statistics (BLS) Occupational Employment Statistics (OES) are available at https://www.bls.gov/oes/tables.htm, while the Employer Costs for Employee Compensation.

8.2.5 Compliance Assumptions

In most cases, analysts should develop baseline and policy scenarios that assume full compliance with existing and newly enacted (but not yet implemented) regulations. Assuming full compliance focuses the analysis on the incremental effects of the new regulatory action without double-counting benefits and costs already accounted for in previous regulatory analyses. That said, it is important to determine whether specific policy options are more likely to result in compliance issues or may be more difficult to enforce. In such cases, it is important to evaluate these effects (e.g., options that require monitoring and reporting may have higher costs, but compliance is easier

61 Voluntary fringe benefits are the categories of employer-paid benefits that are not legally required and include paid leave, supplemental pay (e.g., for overtime), insurance and retirement and savings plans.

to verify) and explore whether alternative options would result in improved compliance and/or easier enforcement.

Assumptions about compliance behavior in the baseline and policy scenarios should be clearly explained in the analysis. When compliance rates are uncertain or expected to vary across policy options, analysts should explore the sensitivity of the results to these assumptions. See Section 5.4.2 in Chapter 5 for a more in-depth discussion.

8.3 Models Used in Estimating the Costs of Environmental Regulation

Several types of models have been used to estimate the social 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 U.S. economy. In this section, we focus on three main model types: compliance cost models, partial equilibrium (PE) models, and computable general equilibrium (CGE) models. Input-output and input-output econometric models should not be used to estimate social cost; however, these approaches and their limitations are also described. Analysts are encouraged to consult with the National Center for Environmental Economics (NCEE) early in the rulemaking process for help in identifying the most appropriate approaches for estimating the costs of a specific regulation.

In practice, some models are simple enough to be implemented in a spreadsheet. Others consist of systems of hundreds or even thousands of equations that require specialized software. Many models are data intensive.[62,63] Given model complexity, a simple model that captures key economic features may be useful to identify which aspects of the regulatory options under consideration likely matter from a cost perspective and therefore warrant further investigation in a more complex model. Likewise, the use of a simple analytic general equilibrium approach is a less resource-intensive way to build intuition within an internally consistent framework before utilizing a CGE model (U.S. EPA 2017). Analysts should rely on a model that is "no more complicated than necessary to inform the regulatory decision" (U.S. EPA 2009).

In some cases, use of more than one type of model may be warranted. Specifically, a more aggregate CGE analysis may complement the cost estimates of a detailed compliance cost or PE model (U.S. EPA 2017). For example, direct cost estimates from a compliance cost model can be used as an input into a PE or CGE model. In some cases, models also can be linked to combine the sectoral detail of a PE approach with the economy-wide features of a CGE model. Text Box 8.1 discusses linking models. Table 8.3 summarizes key attributes by model type.

When selecting a model, it is important to evaluate whether it is the most appropriate for the question at hand (i.e., fit for purpose) and does a reasonable job of approximating the market(s)

62 Data requirements for these models vary, though advances in computing power, data availability and more user-friendly software packages continually reduce the barriers to sophisticated model-based analysis. Refer to Chapter 9 for a discussion of the public and private data sources that can be used for cost estimation.

63 Analysts should take great care in ensuring the quality of a model's data and specifications. See Section 8.4 for a discussion of approaches to parameter selection, and ways to address parameter and model uncertainty.

and behavioral responses of interest. Most model types involve tradeoffs between different strengths and weaknesses. Below are several factors that may be helpful in choosing a model.[64]

- **Types of impacts being investigated.** Models differ in their abilities to estimate different types of costs.
- **Geographic scope of expected impacts.** Some models are well-suited for examining regional or local impacts but may not capture the full range of costs at the national level and vice versa.
- **Sectoral scope of expected impacts.** Some models are highly aggregate and lack the detail necessary to capture important aspects of compliance behavior within a single sector. Likewise, highly detailed sector models often do not capture effects on other sectors and may not adequately capture demand response.
- **Expected magnitude of impacts.** A model well-suited for estimating the cost of a regulation with large effects may have difficulty estimating the cost of a regulation with relatively smaller expected effects, and vice versa.
- **Expected importance of interactions and feedbacks with other sectors.** When regulations are expected to have substantial effects on the broader economy, it is important to choose a model that can capture those effects.

Other criteria for ensuring a specific model are appropriate and of sufficient quality to analyze the effects of a regulation are discussed in Text Box 5.2. For example, chosen models should be subject to credible and objective peer review to ensure consistency with scientific and economic theory before being used in regulatory analysis. Comprehensive documentation of model components, and as possible, underlying data sources should be publicly available.

Usually, some combination of the above factors will determine the most appropriate type of model for a specific application. Analysts should present a reasoned discussion of the factors that inform their model choice. Analysts should describe the main upstream and downstream sectors affected, whether close substitutes to the regulated good are available, the extent to which the goods affected are substitutes or complements to leisure, and the existence of pre-existing distortions in affected sectors (e.g., subsidies, imperfect competition, other regulations, or externalities). Evidence from the literature such as supply and demand elasticities that indicate market responsiveness (e.g., of consumers, input markets, substitutes, and complements) will aid the analyst in justifying model choice. Ultimately, models need to be supported by the data: for example, a single-market PE analysis requires demand and supply elasticities, while a multi-market or CGE analysis requires cross-price elasticities.

64 This list of factors is informed by Industrial Economics, Inc. (IEc 2005).

Text Box 8.1 - Linking Models

CGE models are aggregate representations of the economy that allow an analyst to capture the interactions of producers and consumers as changes in prices and quantities in the regulated sector percolate through the rest of the economy. These economy-wide interactions are captured through exogenously specified elasticities of substitution that approximate detailed demand and supply responses from the policy. There may, however, be instances when CGE models do not have sufficient detail to quantify how regulated entities may respond to a regulation, such as the types of compliance methods that are available.

Partial equilibrium and compliance cost approaches typically do not suffer from a lack of detail. They often have technology-rich representations that reflect the range of salient characteristics for regulated sources as well as installation and operation costs for each individual compliance technology. However, often demand and supply are specified in a very simple way and interactions with other potentially affected markets are not considered.

Much could be gained by linking these two modeling approaches in a coherent and sensible way to take advantage of the technological detail of compliance cost or PE models and the theoretically consistent economic structure of CGE models (Böhringer and Rutherford 2008). There are a number of studies, many in the energy context, that have leveraged such linkages (e.g., Cai and Arora 2015; Rausch and Karplus 2014; Kiuila and Rutherford 2013; Lanz and Rausch 2011; Sue Wing 2006; Schafer and Jacoby 2005; McFarland et al. 2004). The Science Advisory Board (SAB) (U.S. EPA 2017) recommended that EPA make linking more aggregate CGE models to more detailed models of households, industries or sectors a research priority. It signaled a clear preference for two-way linkages between models: the CGE model simulates prices and investment for use as inputs to the compliance cost or PE model, while the compliance cost or PE model computes technology capacities and output supplies that are used as inputs by the CGE model. The two models are run in an alternating fashion until convergence.

It is important to note that, in practice, any linking exercise is dependent on the information available from the sector model and the representation of relevant sectors and markets in the CGE model. As the information and available models may differ significantly across regulatory analyses, any application of linking also may present unique challenges and considerations.

To link a compliance cost model with a CGE model, the accounting of outputs and inputs between the two models needs to be sufficiently aligned. To do this, it is important to disaggregate compliance costs into the factors (e.g., labor, capital, energy, materials) that correspond to the inputs to the sector's production function as specified in the CGE model. However, this is often not a straightforward exercise. For instance, the fixed cost of a compliance method may include both the capital used for a compliance technology and the labor to install it. Likewise, variable costs may include materials as well as labor for maintenance. However, in both cases the shares of the compliance cost from the specific inputs are rarely available. It is also a challenge to aggregate compliance cost information up to the sector level for the purpose of linking to the CGE model. A compliance cost model often provides information on the expected compliance behavior and cost for each affected entity. The CGE model usually represents a sector with a single representative firm.

Many of the challenges of linking compliance cost models to CGE models also apply to linking CGE models with PE models. However, because PE models may have their own sets of assumptions on baseline forecasts, elasticities of demand and supply, functional forms, and/or technical change that may differ from the underlying assumptions in the CGE model, there may be additional complexities that must be grappled with and reconciled in some way to ensure that the linkage remains feasible and produces sensible results.

Table 8.3 Summary of Key Attributes by Model Type

Attributes	Sector-Specific Compliance Cost	Sector-Specific Partial Equilibrium	Economy-Wide CGE
Significant industry detail; rich set of technologies	✓	Sometimes	None
Account for facility or market constraints	Sometimes	✓	✓
Model changes in regulated producer behavior (e.g., input and process changes)	Sometimes	✓	✓
Represent interactions and feedbacks between sectors	None	Limited or none	✓
Model demand side response	None	Limited	✓
Can directly estimate welfare effects	None	Sometimes	✓

8.3.1 Compliance Cost Models

Compliance cost models are used to estimate the direct costs of compliance with a regulation. Estimates by engineers and other experts are used to produce algorithms that characterize the changes in costs resulting from the adoption of various compliance options and are usually determined for individual facilities or for categories of model facilities with varying baseline characteristics. To estimate the control costs of a regulation for an entire sector, disaggregated data that adequately reflect the industry's heterogeneity are used as an input into the model. The disaggregated cost estimates are then aggregated to the industry sector level. These models are most informative when the data are available to capture heterogeneity across facilities, both in terms of individual characteristics (e.g., facility age and production technology, input costs) and compliance options.

The structure of compliance cost models can vary depending on the scope of an analysis. For instance, compliance cost models may include many of the categories of costs previously described in Section 8.2.1 (e.g., capital costs, operating and maintenance expenditures, monitoring, measurement, and reporting costs). Moreover, some compliance cost models are designed to allow the integrated estimation of control costs for multiple pollutants and multiple regulations. Some models account for cost changes over time, including technical change and learning. While most compliance cost models are for facilities within a specific industry, they may also be models of households.

While precise estimates of compliance costs are an important component of any analysis, recall that, in cases where the regulation is not expected to significantly affect market supply and demand in the regulated market, compliance costs can be considered a reasonable approximation of social cost. Compliance cost models usually focus on the supply side because regulations are typically imposed on producers. In circumstances where producer and consumer behavior are appreciably affected, these models are not able to provide estimates of changes in industry prices and output resulting from the imposition of a regulation.

Advantages

- Compliance cost models often contain significant industry detail and can provide relatively precise estimates of the costs incurred by regulated sources (or categories of regulated sources) when complying with a regulation.
- Once constructed, compliance cost models often require fewer resources to implement and are relatively straightforward to use and easy to interpret.

Limitations

- As they usually focus on the supply side and do not capture changes in production among affected sources, compliance cost models can only provide estimates of social cost in certain cases.
- Compliance cost models are often limited to estimating the costs of complying with regulatory requirements for a single industry.

Linear Programming Models

Often linear programming models are used in the analysis of EPA regulations to estimate compliance costs. Linear programming models minimize (or maximize) a linear objective function by choosing a set of decision variables, subject to a set of linear constraints. In the EPA's regulatory context, the objective function is usually to minimize compliance costs incurred by the regulated sources. The decision variables represent the production and compliance 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 even economy-wide effects, they are 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.

In addition to compliance costs, the outputs from the model may include other related variables, such as projected input use, emissions, and demand for new capacity in the regulated industry. In some cases, linear programming models may also include supply and demand representations (e.g., elasticities) of multiple markets and therefore more closely resemble the partial equilibrium models described in Section 8.3.2.

While the estimated change in expenditures incurred by the regulated sector may be of policy interest, it is not equal to social cost when input or output prices change. If the linear programming model captures changes in market prices in response to the policy, then it is possible to use the model outputs to estimate a partial equilibrium estimate of social cost (e.g., changes in producer and consumer surplus).

8.3.2 Partial Equilibrium Models

In cases where the effects of a regulation are confined to a single or a few markets, partial equilibrium single or multi-market models that incorporate anticipated demand and supply responses can be used to estimate social cost.

Inputs into a partial equilibrium model may include regulatory costs estimated using a compliance cost model and the supply and demand elasticities for the affected market (as well as cross-price elasticities when there are multiple affected markets). The model then can be used to estimate the change in market price and output. Changes in producer and consumer surplus reflect the social cost of the regulation.

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 in the region at which expected changes are to occur. The shapes of these curves reflect the underlying elasticities of supply and demand. These elasticities either can be estimated from industry and consumer data or taken from previous studies. While in practice these models often assume perfect competition, it is also possible to construct a partial equilibrium model that accounts for the role of market power in production decisions.

If the elasticities used in an analysis are drawn from previous studies, they should reflect:

- A similar market structure and level of aggregation;
- The appropriate spatial resolution (i.e., local, regional, or national);[65]
- Current economic conditions; and
- 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 to these additional markets as well.

Advantages

- Because they usually simulate only a single or small number of markets, partial equilibrium models generally have fewer data requirements relative to a CGE approach and are more straightforward 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 broader effects in the overall economy.
- 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.

8.3.3 Computable General Equilibrium Models

The most appropriate type of model to estimate the social cost of a regulation in a general equilibrium framework is a computable general equilibrium model. This type of model is comprehensive and internally consistent, accounting for budgetary and resource constraints operating throughout the economy. A key advantage over the other types of models discussed in this section is its ability to capture interactions between economic actors (often delineated with multiple sectors and regions) and with pre-existing distortions (e.g., taxes, other regulations, or externalities) across the entire economy. Relative to PE and compliance cost approaches, however,

65 For instance, Bernstein and Griffin (2006) estimated short-run price elasticities of demand for electricity in the United States that varied from -0.04 to -0.31 by region, and long-run price elasticities of demand for electricity that varied from about -0.05 to almost -0.6 by region.

CGE models are highly aggregate and often use simplified representations of production decisions (e.g., perfect competition, characterization of abatement opportunities) within a sector. They may also be more difficult to parameterize.

CGE models assume that an economy can be characterized by a set of conditions in which supply equals demand in all markets. When the imposition of a regulation alters conditions in one market, a general equilibrium model determines a new set of relative prices for all markets that return the economy to a long-run equilibrium. These prices in turn determine changes in sector outputs and household consumption of goods, services and leisure in the new equilibrium. In addition, the model determines a new set of relative prices and demand for factors of production (e.g., labor, capital and land) — the returns to which compose business and household income. The social cost of the regulation is estimated in CGE models as the change in economic welfare in the post-regulation, simulated equilibrium compared to the pre-regulation, "baseline" equilibrium.[66]

CGE models are built using structural micro-theoretic foundations to capture behavioral responses.[67] In canonical CGE models,[68] firms are generally assumed to be profit-maximizers with constant returns to scale in production; households maximize utility from the consumption of goods and services using a specific functional form; and markets are perfectly competitive. Multiple household types can be included in the model (for instance, differentiated based on geography or income) to calculate distributional impacts of policy changes. A common feature in many models is an underlying model of international trade following Armington (1969) where preferences for goods are differentiated by country of origin to allow for two-way trade for otherwise identical goods. Labor and capital are typically fully mobile between sectors with labor fully employed and no involuntary unemployment.

CGE models are generally more appropriate for analyzing medium- or long-term effects of regulation, when most inputs are free to adjust and consumers can modify purchasing and labor-leisure decisions in response to new prices. A longer time horizon also affords greater opportunities for firms to change production processes (i.e., innovate). The time required to move from one equilibrium to another after a new policy is introduced is not defined in a meaningful way (and is usually assumed to be instantaneous). As such, CGE models are generally not well-suited for

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66 Regulatory compliance creates a need for additional inputs to produce goods in the regulated sector along with pollution abatement. While the total cost of these additional inputs can be derived from detailed compliance cost estimates, it is not always clear how to allocate the total cost among the inputs specified in the CGE model because CGE models are by their nature an aggregated, parsimonious representation of the economy.

67 Structural models explicitly specify underlying preferences, production and resource allocation in ways that are consistent with economic theory. The calibration of structural or behavioral model parameters with actual data ensures that the model represents important economic features while remaining in agreement with the underlying theory (Woodford 2009).

68 Here, the term "canonical" is indicative of off-the-shelf models, or models with features that are most common in the literature. In reality, a CGE model may contain several hundred sectors or only a few and may include a single "representative" consumer or multiple household types. It may focus on a single economy with a simple representation of foreign trade, or contain multiple countries and regions linked through an elaborate specification of global trade and investment. The behavioral equations that govern the model allow producers to substitute among inputs and consumers to substitute among final goods as the prices of commodities and factors shift. The behavioral parameters can be econometrically estimated, calibrated or drawn from the literature. In some models, agents may make intertemporal trade-offs in consumption and investment.

analyzing transition costs as the economy moves to the new equilibrium unless a transition path can be appropriately specified.[69]

The case for using CGE models to evaluate a regulation's effects is strongest when the regulated sector has strong linkages to the rest of the economy and the regulation is expected to affect most firms in a broadly defined sector. Narrowly targeted regulations are more difficult to capture without explicitly linking a CGE model to a detailed PE sector model (U.S. EPA 2017). Linking models is discussed in Text Box 8.1. The extent to which CGE models will add value to the analysis also depends on data availability (see Text Box 8.2 on input-output data efforts).[70] When developing a plan for analysis, analysts should consult with NCEE if they anticipate using a CGE model to evaluate the effects of a regulation.

Note that absent a credible way to represent environmental externalities in a CGE model — or the benefits that accrue to society from mitigating them — a CGE model's economic welfare measure is incomplete.[71] However, the inability to account for interactions between costs and benefits in a CGE model does not invalidate their use to estimate costs or make it impossible to design consistent approaches to cost and benefit estimation (U.S. EPA 2017). The possibility of incorporating benefits into a CGE framework is discussed in Text Box 8.3.

Advantages

- CGE models are best suited for estimating the cost of policies that will have a broad set of economy-wide impacts, especially when indirect and feedback effects are expected to be significant.
- CGE models are most appropriate for analyzing medium- or long-term effects of policies or regulations.

Limitations

- Because of their equilibrium assumptions, CGE models are generally not appropriate for analyzing short-run transition costs.
- CGE models are highly aggregate and do not provide detailed cost estimates for narrowly defined sectors or small geographic areas.
- CGE models may be more difficult to parameterize and use highly simplified representations of sector production and abatement decisions.

8.3.4 Other Input-Output Based Models

Several other economy-wide approaches are referenced in the literature, including input-output (I-O) models and I-O econometric models. These methods should not be used to estimate the social cost of environmental regulation (U.S. EPA 2017).

69 For instance, Williams and Hafstead (2018) embed short run transitional unemployment costs in a general equilibrium model.

70 Data limitations are a significant obstacle for all of the modeling approaches discussed in Section 8.3, both in terms of achieving the granularity needed to adequately represent a regulation and to estimate its effects.

71 An expanding body of work has begun to include non-market goods in CGE models (Smith et al. 2004; Carbone and Smith 2008).

Text Box 8.2 - Input-Output Data and Open-Source Initiatives

Input-output (I-O) data are a basic input into any CGE model. An I-O table assembles data in a tabular format that describes the interrelated flows of market goods and factors of production over the course of a year. It may consist of hundreds of sectors or just a few sectors. In the United States, the Bureau of Economic Analysis (BEA) provides a time series of national level I-O accounts with multiple levels of sectoral aggregation (between 15 and 402 sectors) based on North American Industry Classification System codes. For more information on constructing I-O tables, see Miller and Blair (2009), Horowitz and Planting (2009), and https://www.bea.gov/industry/input-output-accounts-data.

Below is an aggregated I-O table for the U.S. for 2022 based on BEA data. The columns for the individual sectors denote how much of each commodity is used to produce that sector's output (cost of annual production). A sector's cost schedule (upstream sectoral linkages) is composed of intermediate inputs, factors of production (labor and capital) and tax payments. Payments to factors (wages and profits) and tax payments comprise sectoral value added. Take the agricultural sector -- intermediate input costs consisted of $195 billion of agricultural inputs, $129 billion of manufactured inputs, $338 billion of other intermediate inputs and $286 billion of value added, for a total of $948 billion in input costs. The row for each commodity shows how that commodity is consumed (also known as downstream linkages). For the agricultural sector, $711 billion is consumed as intermediate inputs for sectoral production ($195 + $6 + $443 + $66), while $237 billion is consumed as final demand (i.e., C + G + I + (X-M), or $246 + $0 - $25 + $15). In this framework, the total output receipts must equal total input costs.

I-O Table for the United States (2022)

Account	Agriculture	Mining	Utilities	Construction	Manufacturing	Transportation	Services	C	G	I	X-M	Other Taxes	Total Outputs
Agriculture	195	0	1	5	443	0	66	246	0	-25	15	0	948
Mining	2	89	47	33	583	1	84	0	0	99	-21	-32	886
Utilities	10	21	52	15	89	19	319	359	0	0	0	-36	847
Construction	2	5	13	2	23	10	315	0	382	1497	0	-3	2243
Manufacturing	129	108	65	808	2650	226	2391	5097	187	1545	-1315	-596	11296
Transportation	94	106	17	3	413	249	561	320	0	0	51	-4	1809
Services	230	175	141	228	4543	384	9600	11489	3878	1641	298	-285	32322
Labor	67	70	185	726	1225	529	10653						
Capital	206	282	289	410	1258	370	7793						
Prodn. Taxes	13	30	39	14	70	21	540						
Total Inputs	948	886	847	2243	11296	1809	32322						

Source: Numbers are based on tables from the BEA. All values are in billions of 2022 dollars. Note: zeros capture small numbers that round to zero; missing entries reflect actual zeros in the data. C is household consumption (excluding leisure), G is government expenditures, I is investment and X-M is exports minus imports. The sum across these demand accounts equals U.S. GDP in 2022.

In a CGE framework, columns for the individual sectors determine input shares used to calibrate production functions. Columns for final demand determine expenditure shares used to calibrate household expenditure functions. These data, along with "transactions and transfers between institutions related to the distribution of income in the economy," form the basis of the social accounting matrix (Miller and Blair 2009). Constructing a social accounting matrix requires reorganizing the data shown above to link sources of household income to expenditures. I-O accounts with sub-national or international detail are not provided by the BEA but have been established by others. For instance, the Global Trade and Analysis Project (GTAP) compiles and reconciles data from many sources to have consistent sectoral and agent aggregations across countries (https://www.gtap.agecon.purdue.edu/). Further, the Wisconsin National Data Consortium (WiNDC), develops consistent subnational I-O tables based on publicly available data (http://windc.wisc.edu/).

8.3.4.1 Input-Output Models

I-O models are highly disaggregated empirical descriptions of the interrelated flows of good and factors of production.[72] They are generally static and assume a fixed, strictly proportional relationship between inputs and outputs via multipliers.[73] 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, most input-output models are demand driven and not constrained by limits on supply, which would normally be transmitted through increases in prices. While some of the rigidities in the models may be reasonable assumptions in the very short run or for regional analysis with limited ties to the broader national economy, they limit the applicability of I-O models for evaluating medium- to long-run effects or national issues. For instance, the lack of resource constraints and substitution effects that occur over the longer run means that I-O models tend to overestimate the effects of a policy.[74] Importantly, the I-O approach does not necessarily account for shifts in economic activity toward the pollution abatement sector (e.g., when the directly regulated sector purchases pollution abatement equipment or services to comply with the regulation). 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.[75]

72 Miller and Blair (2009) is a standard reference on input-output analysis.

73 The assumption that output changes translate directly to proportional changes in inputs is not empirically founded and therefore should not be used, even in the short run, because it ignores the potential for factor substitution. Such shifts may change the labor-, capital-, energy- or materials-intensity of production.

74 Studies that rely on I-O models often calculate some combination of direct, indirect and induced effects. Direct effects are the changes in output that result from an increase in the cost of inputs (e.g., fuel) in the directly regulated sectors, using the fixed, proportional relationship mentioned above. Indirect effects of a regulation are calculated by using the I-O relationship between outputs in the directly affected sectors and required inputs in related sectors (e.g., suppliers). Induced effects are general re-spending effects that result from changes in household income.

75 See U.S. Chamber of Commerce and NERA Consulting 2013; OECD 2004, and Dwyer et al. 2006.

Text Box 8.3 - Separability between Benefits and Costs

When estimating the benefits and costs of environmental regulation, it is almost always assumed that the two are separable, such that the beneficial impacts of the polices do not meaningfully affect the factors that determine the cost of the policy, and vice versa. This is due, in part, to a lack of empirical evidence regarding the sign or importance of the relationship between environmental quality, which is typically not priced in the marketplace, and market goods (Carbone and Smith 2008).

Benefits and costs are non-separable when either the compliance costs borne by firms or households interact with and alter the valuation of environmental benefits (other than through changes in environmental contaminants), and/or the beneficial impacts of the regulation alter the costs. Non-separability may occur for several reasons. The costs of an environmental policy may alter the budget constraint for households — for instance, when compliance costs are passed on to consumers as higher prices for goods such as electricity, and this, in turn, affects their willingness to pay for the beneficial impacts of the policy. It may also be the case that changes in environmental quality and health status lead to changes in household behavior, which for large policies could affect relative prices in equilibrium and the cost of complying with the policies. For example, greenhouse gas mitigation polices in the electricity sector may reduce future demand for space cooling and therefore electricity, in turn reducing the costs of complying with the policy.

Ongoing work also suggests that reductions in mortality risk may affect how households smooth consumption over time (i.e., through savings), which may interact with pre-existing capital taxes or affect the price of investment in pollution abatement capital (Marten and Newbold, 2017). The fact that changes in environmental quality and health status can affect behavior in markets underpins the revealed preference approaches for estimating willingness to pay discussed in Chapter 7.

As noted by the SAB (U.S. EPA 2017), when either the costs or benefits of a regulation are estimated while holding the other constant, any potential non-separability between costs and benefits is missed, which complicates comparing them and calculating social net benefits. The specific magnitude and ultimate impact of non-separability on the net benefits of environmental regulations is an empirical question that requires additional study and is the subject of an emerging literature (Sue Wing 2011). The SAB noted that potential non-separability for large policies does not invalidate estimates of costs and benefits using existing methods. However, caution should be applied when obvious interactions exist.

8.3.4.2 Input-Output Econometric Models

I-O based econometric models integrate the high level of detail from an input-output model with the forecasting properties of a macro-econometric forecasting model. Unlike standard I-O models, this approach accounts for supply-demand conditions in the economy, including resource constraints, through a series of accounting (e.g., savings equal investment) and econometrically estimated relationships (Hahn and Hird 1991). Feedbacks between supply and demand occur via econometric equations (CGE models accomplish this via a price mechanism and market clearing assumptions (West 1995)). The predictions generated by this type of model "are integrated and simultaneously determined … price increases in one sector are translated into cost and price increases in other sectors" (Portney 1981). This is a key advantage over standard I-O models that assume away these effects.

While CGE models assume full market clearing, I-O econometric models assume imperfect knowledge of product and factor markets, with an emphasis on tracking short run disequilibrium (West 1995). This makes them particularly attractive for analyzing transition costs. However, a major drawback of I-O econometric models is that because they are reduced-form models predicated on historical relationships, they cannot take into account the possibility that a firm or consumer may modify their long-run behavioral response to changes in policy (referred to as the Lucas critique). The inability to account for this dependence invalidates these models for purposes of policy evaluation outside of short-term forecasting (Schmidt and Wieland 2013; Fischer and Heutel 2013; U.S. EPA 2017).

8.4 Modeling Decisions and Challenges

Even when the analyst has determined what types of models are most appropriate for the estimation of social cost, several important modeling decisions remain, including deciding on the level of sectoral and regional aggregation, whether to use a static or dynamic framework, and how to parameterize the model. In addition, analysts should evaluate key uncertainties and take care not to double count, particularly when using outputs from one type of model as an input into another.

8.4.1 Aggregation

The level of sectoral and regional aggregation assumed in a model will determine what aspects of the sector or economy can and cannot be captured explicitly in a regulatory analysis. Matching the level of aggregation in a model to the level needed to evaluate a policy's main effects is important to ensure that the analysis does not miss important contributors to the cost. For example, consider the effects of a new regulation on refrigerant gases in the frozen bakery products sector. In a CGE model, the frozen bakery products sector is not typically separated out as its own sector. Instead, it is captured in a more aggregate category — food products — along with many other related industries such as soft drinks, cereal and chewing gum. As a result, the frozen bakery products that are affected by the policy "may be too small a part of the model's food products sector to give meaningful results due to 'aggregation bias.'[76] Put another way, there are too many products in the model's sector to accurately isolate the frozen bakery products industry" (Rivera 2003).

The level of aggregation can affect sectoral and economy-wide results. For instance, sectoral disaggregation allows for a differentiated representation of production technologies, behavioral parameters (e.g., elasticities) and emission intensities that may matter for estimating costs and other impacts (Alexeeva-Talebi et al. 2012).[77] However, models that are highly specialized for

76 Caron (2012) defines "aggregation bias" for a specific variable as the difference in its value from an aggregated model relative to a disaggregated model that has been re-aggregated after the fact to a comparable level.

77 Alexeeva-Talebi et al. (2012) and Caron (2012) found that the range and standard deviation of sectoral impacts increased with disaggregation. In some cases, even the direction of the estimated impacts was reversed relative to more aggregate results. However, while a highly aggregated model may not be a reliable predictor of sub-sectoral impacts, for many applications, they found that these models produce satisfactory estimates of the overall impacts on the economy.

capturing impacts in a specific sector will usually miss impacts on a broader set of sectors. It is also important to consider how costs are allocated spatially (and temporally) to avoid a mismatch between affected facilities' locations and the scale of the model. While proficient at capturing major impacts and interactions between sectors, CGE models generally are not well-suited for focusing on a single or small number of specialized sectors because of their level of aggregation.

8.4.2 Choosing Between a Static and Dynamic Framework

It is possible to construct static or dynamic versions of all three types of economic models discussed in this chapter (i.e., compliance cost, PE and CGE). In a compliance cost framework, the analyst may assume that economic conditions are static or dynamic. If future economic conditions are expected to change meaningfully, a dynamic framework should be used because compliance decisions may be influenced by future economic conditions even when the regulation is not expected to meaningfully influence production or prices. For example, if an affected source anticipates operating for a long time it may choose a more capital-intensive compliance option over a less-capital intensive option because there is a longer period over which it can recover the cost of that investment. Similarly, if the number of affected sources is anticipated to change over time, then the cost of complying may change over time. A dynamic compliance cost framework may also be preferred if, for example, regulated sources may make anticipatory investments prior to a regulation's compliance dates or to account for the potential for technological change (see Section 8.2.3.4).

When the analyst expects intertemporal effects of a regulation to be confined to the regulated sector or a few related sectors, it may be appropriate to simply apply partial equilibrium analysis to multiple periods. As with compliance cost models, relevant conditions, like expected changes in market demand and supply over time, should be taken into account in the analysis. The costs in individual years can then be discounted back to the initial year for consistency.

If the intertemporal effects of a regulation on non-regulated sectors are expected to be significant, analysts also can estimate social cost using a dynamic CGE model. Dynamic CGE models can capture the effects of a regulation on affected sectors throughout the economy. They can also address the long-term impacts of changes in labor supply, savings, factor accumulation and factor productivity on the process of economic growth. 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.

Analysts should keep in mind that the evolution of variables in a dynamic model sometimes depends on exogenously imposed assumptions that are not always easy to validate. For instance, modelers sometimes need to constrain the pace at which some variables in the model change (e.g., how quickly technology changes) based on an external assessment of what is technically feasible. Key exogenous assumptions should be clearly documented and explained. In some cases, it also may be useful to explore the robustness of cost estimates to alternative assumptions.

8.4.2.1 Expectations

Dynamic models must specify the ways in which households and firms formulate and update expectations about future prices, returns, growth or other key economic variables. There are a variety of ways to formulate expectations about the future, but they generally fall into two general categories: backward-looking and forward-looking. With advances in computer power, forward-looking expectations are the more common assumption in CGE models.

The two main backward-looking formulations are myopic and adaptive expectations. Myopic households and firms do not anticipate future changes to the economy or regulatory setting, and do not make investments or change consumption and savings behavior until the period when the change takes effect (Paltsev and Capros 2013). Households and firms with adaptive expectations base their expectations about the future primarily on past experiences and are, therefore, relatively slow to modify behavior in response to new information.

In forward-looking models, households and firms have either perfect foresight or rational expectations. A household or firm with perfect foresight knows what the future values of key economic variables will be with certainty and incorporates this information immediately into current decisions (Paltsev and Capros 2013). Modeling rational expectations allow for households and firms to account for uncertainty in future conditions; in this case, they incorporate all relevant information, both past and future, into decision-making and are assumed to get future values correct on average. In other words, they do not systematically make forecasting errors.

There are several analytic implications tied to the degree of model foresight assumed in a dynamic model. For instance, a backward-looking model may lead to higher estimates of compliance costs and welfare impacts compared to a forward-looking model since it restricts the response flexibility of households and firms relative to reality. However, in cases where they are assumed to have perfect foresight but the future path of key variables is uncertain in reality, a deterministic forward-looking model may underestimate the compliance costs and welfare impacts of regulation. Note that many EPA regulations phase in standards or allow for intertemporal smoothing of compliance (e.g., banking of emissions allowances) that could at least partially alleviate this concern. Section 5.5.1 provides additional information on the role of uncertainty on household and firm behavior when estimating the impact of regulation.

Another consideration is the large number of variables and constraints that must be simultaneously determined in a forward-looking model. This, in turn, restricts the level of detail that can be included in the model, which may be critical to adequately assessing the social cost of a regulation. As such, a more aggregate forward-looking CGE model should be viewed as a complement to analysis supported by detailed compliance cost or PE sector model. All else equal, these considerations are even more restrictive for forward-looking models of decision making under uncertainty that also necessitate integrating over all temporal sources of uncertainty to form household and firm expectations.

8.4.2.2 Time Steps

Static models provide cost estimates for one period, typically a year. They either assume that conditions are invariant over time or that the cost estimate is indicative of a typical or representative period. Static models exist for all three frameworks discussed in this chapter (i.e., compliance cost, PE and CGE). As discussed above, if economic conditions are expected to change over time, or if changes in behavior to come into compliance and/or to new market equilibria take time, static models may provide incomplete estimates of costs (and benefits).

Most dynamic models operate using discrete time steps. Time steps between periods are chosen to provide enough detail regarding the adjustment to policy over time, while using a manageable number of time periods for computational reasons. For instance, because dynamic CGE models are often solved over periods of 50 years or more, it is not always practical to solve the model for each individual year. However, when using a dynamic CGE model, the year in which a regulation comes into effect may not be explicitly modeled. Due to the expense and time required to adjust the model and baseline, adding a new solution year may not be an option. In this instance, analysts may use the model year closest to the year in which the regulation will come into effect as a proxy.

Regulations that are introduced gradually or vary timing of compliance by region or state pose additional challenges for model representation.

In addition, if the end-year chosen for a dynamic model stops short of capturing important regulatory effects, the social cost estimate may be biased downward. When compliance costs cannot be estimated for all future years, a forward-looking model may smooth them over time, which can also lead to biased social cost estimates, though the direction of the bias will depend on what is assumed about future compliance costs.

8.4.3 Model Parameterization

Regardless of the chosen modeling framework, there is a distinction between values determined within the model (those that are endogenous) and values determined outside of the model (those that are exogenous). Model parameterization is concerned with the latter.[78] The values that are imposed exogenously will depend on the type of model used to capture economic behavior.

In general, model parameterization takes place in two steps. First, the analyst attempts to accurately represent the current structure of the sector(s) or markets of interest. For compliance cost models, this step typically relates to specifying relevant compliance options and constraints (e.g., production capacities). In the case of CGE models, this step consists of characterizing a baseline and calibrating model functions to a reference equilibrium. Second, parameter values are chosen that best characterize economic relationships (i.e., the curvature of different functions) in the model. In the case of compliance cost models, the analyst may need to specify constraints on economic behavior (e.g., production levels, other regulatory requirements) and cost functions (e.g. slopes, nonlinearities). For partial and general equilibrium models, this second step is more difficult and requires the analyst to choose appropriate behavioral parameters (e.g., elasticities).

Parameter values can be estimated or based on existing values from the literature. While basing parameter values on the existing literature is the more common approach, inconsistencies between the underlying structure of the model and the empirical analyses from which values are drawn can lead to modeled responses that are not supported by the underlying data of the empirical analyses.[79] To alleviate some of these concerns, some researchers have econometrically estimated the model parameters in a framework that is consistent with the underlying model (e.g., Jorgenson et al. 2013). If parameters are estimated by the analyst, preference should be given to using publicly available data where possible. When borrowing estimates from the literature to parameterize a model, analysts should use estimates that reflect the most recent scientific methods and data as possible, discuss the reasons for choosing one value over another, and discuss any limitations. For instance, available parameter values in the literature may not be regularly updated and produced using data that are significantly older than the modeled year(s) of interest. In cases where there is

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78 Specifically, parameters are "terms in the model that are fixed during a model run or simulation but can be changed in different runs, either to conduct sensitivity analysis or to perform an uncertainty analysis when probabilistic distributions are selected" (U.S. EPA 2009).

79 This point also applies to instances where analysts estimate their own parameters for a model. Identifying assumptions in the empirical framework need to be consistent with assumptions in the model of interest. For CGE analysis illustrating this point, see Shoven and Whalley (1984) and Canova (1995).

no clear consensus in the literature on the most defensible estimates to use, sensitivity analysis to understand the robustness of cost estimates to the parameters chosen is recommended.[80]

Often, a regulation covers many highly heterogeneous facilities where both the compliance options available and abatement costs vary widely. When highly disaggregated source-level data are unavailable, analysts may pursue a model plant approach to estimate compliance costs, where a subset of individual facilities sharing certain characteristics (e.g., plant age, type of production process, industrial sector) are represented by a single model plant. Analysts also may use a model plant approach to reduce the computational requirements of a compliance cost model.

The model plant is intended to represent the typical conditions of a group of facilities. While this provides a way to overcome data limitations and simplify the model, parameterization can still prove challenging. This is particularly true when conditions vary significantly across seemingly similar facilities. For example, if an abatement technology exhibits positive economies of scale, the compliance cost of an average-sized facility will not equal the average cost of all the facilities represented by that model plant. This is because, with economies of scale, the higher cost to smaller facilities will outweigh the lower cost to larger facilities relative to the average-sized facility. In this case, the compliance costs of the facilities represented by the model plant will be under-estimated. It is therefore important for the analyst to carefully consider the number of model plants needed to capture the heterogeneity among constituent facilities that could affect compliance cost estimates.

Assumptions on variables and parameters determined outside of the model can be important drivers in applied CGE analysis. For instance, estimates of elasticities that help define production processes and agent preferences are of particular interest because model results are often sensitive to these parameters. CGE-derived social cost estimates are particularly sensitive to parameters that affect behavior in labor markets due to their pre-existing distortions, such as the assumed elasticities governing the labor-leisure choice of consumers and production elasticities between factors of production (Marten et al. 2019).[81] Model results tend to be more sensitive to behavioral assumptions, for instance, the values chosen for elasticities relative to other data inputs such as the benchmark input-output data (see Text Box 8.2) (Elliot et al. 2012). Additional model closures often used in CGE modeling, like a fiscal budget closure, can also impact social costs and/or incidence depending on the modeling context. It may be important to conduct additional sensitivity analysis around these assumptions to understand the robustness of results if there is uncertainty in which closure mechanisms should be chosen.[82]

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80 Moreover, while many models are parameterized by a point estimate, Hertel et al. (2007) suggest that the confidence in modeled results may depend on the precision of the parameter estimate. The authors note that standard errors derived from the estimation framework for the parameter can also be used to guide sensitivity analysis. Also, see Section 8.4.4.

81 Previous research has also illustrated these sensitivities in other contexts. For instance, Shoven and Whalley (1984) observe that results from CGE analyses of the U.S. tax system are sensitive to labor supply, saving and commodity-demand elasticity assumptions. Fox and Fullerton (1991) find that estimates of welfare changes associated with tax reform are more sensitive to assumptions about the elasticity of substitution between labor and capital than the actual level of detail about the U.S. tax system in the model.

82 Closure rules are exogenous assumptions made in the CGE model that characterize aspects of the economy that are not explicitly modeled. For instance, many CGE models hold government consumption fixed by assigning a fiscal closure rule that assigns how budget surpluses or deficits induced by a policy are recycled back to households (e.g., lump sum, through the tax system). Goulder (2013) summarizes the economics literature on the implications of alternative revenue recycling closures for climate policy analysis.

8.4.4 Uncertainty

Clear communication of uncertainties is critical for transparency of the analysis. Uncertainty in social cost estimates can arise from uncertainty regarding the baseline, affected universe of facilities, policy responses, the number of affected markets and the cost of compliance activities. The degree to which these and other factors affect the confidence placed in the social cost estimates should be carefully reported and quantified when appropriate and possible.

While some key uncertainties have implications for both benefits and costs (e.g., for the baseline or affected facilities), several are unique to social cost estimation. For instance, estimates of compliance costs are often "study-level" estimates, used by engineers to judge the economic feasibility of projects prior to engaging in a costly planning process, and are associated with an error (e.g., +/- 30%).[83] In some cases, more precise cost estimates, described by engineers as "scoping" or "detailed" estimates, may be available. When compliance costs are used to approximate or generate social cost estimates, qualitative and quantitative information available on the degree of precision in the underlying estimates should be prominently discussed to provide appropriate context.

Uncertainty regarding the costs of compliance will propagate through to the estimate of social costs when used in a partial or general equilibrium model. Estimates of social costs may also be subject to model and parameter uncertainty. Model uncertainty refers to uncertainty in a model's ability to accurately represent underlying processes relevant to understanding how an intervention affects the system of interest (for example, due to simplifications necessary to tractably model complex systems) (NRC 2009). As noted in Section 8.4.3, challenges in parameterizing models, including the choice of functional form, also may be a prominent source of uncertainty. Conducting sensitivity analysis or more sophisticated probabilistic analysis across a tractable range of identified uncertainties can provide information on the robustness of the central social cost estimates.[84]

As noted in Section 8.2.3.4, technical change and learning can have an important effect on future compliance costs. Estimates about the effect of innovation will be inherently uncertain and, in some cases, may not be available. Even so, the expectation is that technological change and learning generally leads to lower social costs over time compared to a scenario that assumes no innovation occurs; uncertainty in this case is asymmetric, as innovation is unlikely to increase future costs.

Uncertainty may also affect social cost estimates when projecting the costs of regulations that are implemented by local or state jurisdictions in the future. For example, in illustrative attainment analyses conducted for some National Ambient Air Quality Standards (NAAQS), once all identified control technologies have been applied, some areas of the country may still be modeled as out of compliance with the air quality standard. In these cases, it is uncertain how attainment will be achieved and at what cost. Similarly, in the case of deregulatory actions, how state and local jurisdictions respond — for example by potentially enacting protections in place of the forgone federal standards — can affect the ultimate cost (and benefits) of relaxing the federal standard. In these cases, sensitivity analysis is useful for understanding the robustness of social cost estimates to alternate assumptions.

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83 For example, EPA's Air Pollution Control Cost Manual (U.S. EPA 2018) notes that "costs and estimating methodology in this Manual are directed toward the "study-level" estimate with a probable error of +/-30 percent."

84 See Chapter 5 for further discussion of uncertainty and sensitivity analyses.

8.4.5 Potential for Double-Counting

Because a regulation may have multiple effects through the economy, the analyst should take particular care to avoid double-counting costs. 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. Also, when reporting private costs for certain groups, the portion of those costs that reflect social costs versus transfers to other groups should be clearly identified in the analysis.

Even in a general equilibrium analysis, analysts must take care in selecting an appropriate measure of social cost. Calculating social cost by adding together estimates of the costs in individual sectors can lead to double counting. Instead, focusing on measures of changes in final demand, so that intermediate goods are not counted, can avoid the double-counting problem.

When analysts rely on multiple models that take fundamentally different approaches to cost estimation, care should be taken to separately report and characterize each model's output to avoid double-counting. For example, if a technology-rich PE model is linked to a CGE model, the estimate of social costs comes from the CGE model. The social cost is not the sum of the costs from the CGE and PE models. Furthermore, the cost estimate from a compliance cost model, for example the increased expenditures on compliance activities in the sector, should not be reported as the social cost of the regulation without further elaborating what this cost estimate represents, why it provides a reasonable estimate of the social cost, and that it is not equivalent to the actual social cost of the rule.

Chapter 8 References

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Chapter 9 - Economic Impacts

A wide variety of economic impacts can occur as a consequence of environmental policy. Analysis of who will experience gains and who will be burdened by a regulation, and analysis of the nature and magnitude of regulatory impacts, provides important information for decision makers, stakeholders and the broader public. An *economic impact analysis* (EIA)[1] identifies and quantifies a wide range of regulatory impacts including market-based impacts such as changes in employment, prices, profitability and plant closures; as well as impacts outside the marketplace (e.g., impacts on state and local governments). An EIA identifies specific groups that may benefit or be burdened by a policy and assesses the impacts they experience. Affected groups may include consumers, industries, small businesses, workers, communities, tribes and governments. Using this definition of an EIA, this chapter discusses issues relevant to estimating the economic impacts of EPA policies. An EIA can be tailored to improve understanding of specific regulatory impacts. However, in some instances, EPA has been directed to conduct an EIA, as explained in Section 9.2 of this chapter. Subsequent sections begin with a review of frameworks that provide a general understanding of economic impacts, followed by guidance for assessing each impact category.

This chapter primarily focuses on market impacts due to compliance costs. However, Section 9.5.6 is a discussion of the impacts of benefits (changes in environmental quality and public health) and several other sections, such as Section 9.5.1.5, briefly discuss specific beneficial impacts. Impacts on governments and non-profits are discussed in Section 9.5.4; and a consideration of economy-wide impacts from both costs and benefits is discussed in Section 9.5.5. Chapter 10, “Environmental Justice and Life Stage Considerations,” complements the current chapter by discussing how regulation might change the distribution of environmental quality and health risks across minority and low-income populations, and by life stage.

1 At the EPA, an EIA differs from a Regulatory Impact Analysis (RIA). The latter Is frequently used interchangeably with "economic analysis" and may contain analyses of benefits, costs and economic impacts; in other words, an EIA is often contained within an RIA. For more information, see Chapter 1.

9.1 Background

Analyzing economic impacts sheds light on the distribution across groups of costs, transfers,[2] benefits and other economic outcomes induced by regulation. An EIA may include a broad range of measures including monetized metrics such as profit or price changes, as well as non-monetized metrics such as changes in employment or the likelihood of plant closures. The crux of an EIA is understanding these changes experienced by specific groups. In contrast, a BCA focuses on measuring aggregate social net benefits and is concerned with economic efficiency which requires that benefits outweigh costs, irrespective of to whom net benefits accrue. Thus, the two types of analyses use different measures. Unlike aggregate benefit and cost measures calculated for a BCA, the impact measures included in an EIA need not be mutually exclusive. For example, an impact that appears simultaneously in two related markets, such as costs in the regulated sector and revenues in the pollution control sector, can be included and appear as two impacts in an EIA. In BCA, where the focus is on aggregate efficiency, transfers which, by definition, shift money from one group to another will not impact estimates of net benefits. However, because transfers affect who experiences gains or burdens from a policy, they may be key within an EIA (OMB 2023).

Despite these important differences, analyses of economic impacts in an EIA and of social benefits and social costs in a BCA are complementary, as both shed light on the consequences of regulation. When conducted for the same policy, both types of analyses should use a consistent baseline and set of assumptions. Generally, both analyses have similar scopes; that is, if it is appropriate for the analysis of social costs to extend to markets beyond the regulated industry then it would likely be appropriate for the EIA as well. Both analyses should explain underlying assumptions, explore the sensitivity of results to assumptions and inputs, strive for transparency and include documentation and references (OMB 2023).

Whether regulatory consequences are measured in terms of economic impacts, changes in social welfare or both, ultimately the focus is on how *people* are affected. An EIA that analyzes profitability, for example, is studying potential impacts on the income of firm owners or shareholders. Analysis of employment impacts sheds light on impacts on workers. An EIA that estimates changes in prices is concerned about impacts on consumers. To complicate matters, many impacts estimated in an EIA give insight into changes that might affect multiple groups. For example, an increased likelihood of plant closure affects both firm owners and workers.

9.2 Statutes and Policies

Multiple statutes and policies contain directives for an EIA that are applicable across media.[3] The following statutes and executive orders (EOs), described more fully in Chapter 2, directly address economic impacts.

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2 Transfers are shifts of money or resources from one part of the economy to another such as tax payments. See Section 8.2.2.2 for a discussion of compliance costs and transfers. Circular A-4 defines a transfer as "… a shift in money (or other item of value) from one party to another. More generally, when a regulation generates a gain for one group and an equal-dollar-value loss for another group, the regulation is said to cause a transfer from the latter group to the former." (OMB 2023)

3 The EPA's Action Development Process (ADP) Library is a resource for analysts who wish to access relevant statutes, EOs or Agency policy and guidance documents. Besides the broadly applicable statutes and directives

- Regulatory Flexibility Act (RFA), as amended by the Small Business Regulatory Enforcement Fairness Act (SBREFA) (1996);
- Unfunded Mandates Reform Act (UMRA) (1995);
- EO 12866, Regulatory Planning and Review (1993) as amended by Executive Order 14094, Modernizing Regulatory Review (2023);
- EO 13132 (1999), “Federalism;"
- EO 13175 (2000), “Consultation and Coordination with Indian Tribal Governments;”
- EO 13211 (2001), “Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution or Use.”

Together with OMB's Circular A-4, these directives highlight features of affected entities that may be relevant for EIAs. Table 9.1 lists the features identified by these directives and offers examples of potentially affected groups.

Table 9.1 - Features of Potential Relevance to Economic Impact Analyses as Identified by Statutes, Executive Orders and Guidance

Feature	Statute, Order or Directive	Examples of Potentially Affected Economic Groups
Sector	UMRA; EO 12866; EO 13132; EO 13175; OMB Circular A-4	Producers; industries; state, county, local, territorial, or tribal governments.
Entity size	RFA/SBREFA; UMRA; EO 12866, OMB *Circular A-4*	Businesses, governmental jurisdictions, not-for-profit organizations. Analyze small entities separately.
Time; Dynamics	OMB Circular A-4	Groups (e.g., consumers, workers, producers, firms, industries) experiencing transitional or long-run impacts.
Geography	UMRA; EO 12866; OMB Circular A-4	Regions, states, counties, non-attainment areas, local or regional markets.
Energy	EO 13211	Energy sector (i.e., developers, distributors, generators, or users of energy resources).

9.3 Connections between Economic Impacts and Frameworks of Distributional Effects

Virtually any economic measure of the consequences of a regulation may be included in an EIA.[4] To accommodate this degree of flexibility, an EIA is not constrained or governed by an operating framework. However, there are several conceptual frameworks in the economics literature that

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discussed in this section, there are also environmental statutes with specific applicability that require consideration of impacts on certain populations (such as impacts on labor; see Section 9.5.1.4), or that may require analysis of impacts for facilities potentially eligible for regulatory variances.

4 For textbook discussions of the meaning and usefulness of impact analysis, see Field and Field (2005) and Tietenberg (2006).

provide insight into the meaning and interpretation of impact categories. Section 9.3.1 provides a summary.

While a worthwhile analytic objective for environmental policy is to estimate the net welfare changes experienced by each affected group in an economy, the EPA does not currently conduct such analyses. The information needed to distribute shares of regulatory costs, benefits and transfers among groups and estimate each group's net welfare change is not available. This is explained In Section 9.3.2.

9.3.1 Conceptual Frameworks

A deeper understanding of economic impacts can be achieved by drawing connections to conceptual frameworks of distributional effects. These frameworks, often presented in terms of welfare effects, are useful for understanding parts of an EIA because they illustrate the different pathways through which regulatory costs are distributed across population groups.[5] Expenditures are incurred by regulated entities to comply with environmental mandates, standards, permit requirements, taxes and so on. Compliance expenditures may be passed on partially or fully to other groups.[6] For example, costs may be experienced by firm owners or shareholders through lower profits, or passed on to consumers through higher prices. Or, costs may be passed on to workers through changes in labor compensation, and/or on to the owners of other factors of production through reduced rates of return to land and capital.[7] The portion of the cost experienced by these different groups depends on a variety of factors including the time-frame under consideration, the characteristics of the regulated market such as the elasticity of demand relative to the elasticity of supply and whether there are barriers that prevent new firms or imports from entering the market. Some costs may trickle through to related markets. While in practice economists cannot always measure the extent of cost pass-through, existing frameworks help shed light on the variety of ways that costs percolate through the economy.[8]

A framework developed by Harberger (1962) to better understand the distributional effects (incidence) of taxation provides insight into who bears the costs of environmental regulation. Effects are separated into two broad categories: those falling on the *sources* of income including owners of firms, labor, capital or land; and those falling on the *uses* of income, or consumption, due to changing prices. Harberger's simple two sector, two good model representing a perfectly competitive closed economy with perfectly mobile factors of production suggests that a tax on one input could lead to either, or both, source-side and use-side effects. Adapting the model to

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5 If the regulated entity is not a profit-maximizing firm, then the principles discussed in this section are likely not relevant. We address impacts on governments and non-profits in Section 9.5.4.

6 For a more detailed discussion, see Tietenberg (2002, 2006), which is the basis for the discussion in this paragraph. Useful textbook discussions are also provided by Kolstad (2000) and Field and Field (2005). For a review of the empirical literature, see Bento (2013). For a discussion specifically of the effects of command and control regulations, see Fullerton and Heutel (2010).

7 Throughout this chapter, all factors of production are represented by either land (natural resources), labor (human resources) or capital (man-made resources).

8 The following sources provide frameworks for understanding distributional impacts of environmental regulation: Christiansen and Tietenberg 1985; Baumol and Oates 1988; Field and Field 2005; Tietenberg 1992, 2002, 2006; Serret and Johnstone 2006; Kristrom 2006; Fullerton 2009, 2011; Robinson et al. 2016; Fullerton and Heutel 2010; Fullerton and Muehlegger 2019.

represent an environmental tax shows a use-side burden on purchasers of the commodity in the taxed sector; and a source-side impact on factors affected by the tax (Fullerton and Muehlegger 2019). Many other existing frameworks also categorize distributional effects according to the route through which the effect is transmitted (product prices, profits, shifts in factor compensation) which is then traced to the group on which the effect falls (consumers; owners of firms, land or capital; workers).

Figure 9.1 illustrates how Robinson et al. (2016) conceptualize one set of pathways through which regulatory compliance costs may eventually be distributed across population groups. These pathways help contextualize metrics that often appear in an EIA. The groups experiencing economic impacts as described in Section 9.5 (producers, workers, other factors of production, consumers, communities and the overall economy) are related to one or more of the three routes through which regulatory compliance costs flow.[9] The groups themselves, however, do not always align perfectly with the three groups identified in the figure (consumers, employees and owners). For example, the figure does not directly represent "producers," yet impacts on producers are commonly analyzed at the EPA, and directives to consider some producer impacts are given by statute or EO.[10] Impacts on producers will ultimately be felt by all the people who together make up affected firms (owners and shareholders, workers and other owners of productive factors).[11] Other impact categories discussed in Section 9.5, such as impacts on labor or employees, are more directly represented by Figure 9.1.[12] The right-hand box conceptualizes how costs might be experienced across different population groups; for example, among regions or among households with different demographic characteristics. This is a common endpoint for an EIA as explained in the sections below on specific impact categories — for example, Section 9.5.2 explains how price increases might be experienced differently by high- versus low-income consumer groups.

Fullerton (2016) offers a more nuanced framework for disaggregating regulatory consequences. He identifies the following *potential* cost-related effects on the regulated market:[13] (1) an increased cost of production results in an increase in the price of the regulated good affecting people who purchase the good; (2) decreased production reduces revenues and changes relative returns to

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9 Government and non-profit organizations are also discussed In Section 9.5, but they are structured differently than private firms and are not well represented by Figure 9.1.

10 For example, RFA/SBREFA and EO 13211 (2001) direct agencies to consider impacts on small firms, and on energy producers, respectively.

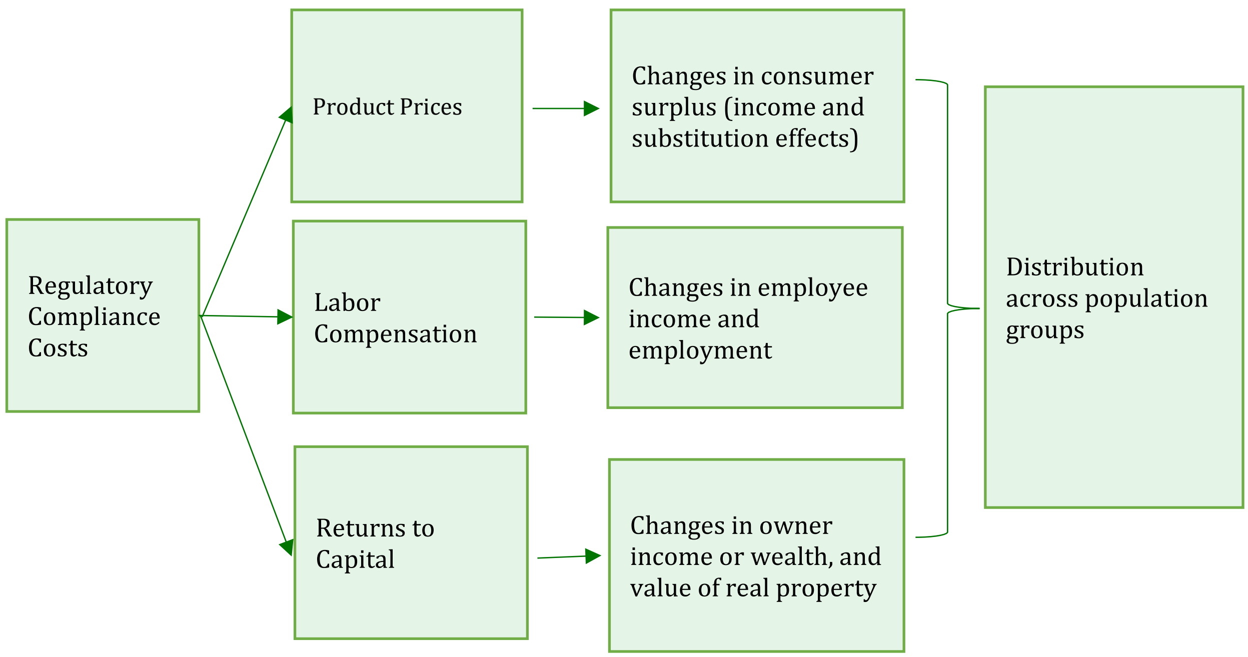
11 Through impacts on producers, regulatory costs could also affect upstream suppliers of inputs (e.g., coal) by leading them to lower their prices, thinking that if they do not, the regulated facilities (e.g., power plants) could shut down.

12 Some "changes" in Figure 9.1 may be measured as economic impacts, welfare changes or possibly both. For more context on Figure 9.1's "changes in employee income and employment," see Text Box 9.1 on labor impacts and benefit cost analysis.

13 Some of these effects may be negligible or may not occur at all. Fullerton (2016) also identifies channels through which distributional effects can occur on the benefits side. For example, asset prices can be affected by environmental quality improvements (e.g., improvements could be capitalized into land and housing prices (and some households could be dislocated due to higher rents). See Sections 9.5.1.5 and 9.5.3; and Chapter 10 for more discussion.

workers and firm owners and factors of production; (3) restrictions on pollution create scarcity rents[14] for owners of firms, capital, and/or land; (4) transitional impacts occur as the economy adjusts to a new equilibrium, for instance, if workers must search for new jobs; and (5) gains and losses are capitalized into asset prices such as corporate stock prices rising due to an expected future flow of scarcity rents.[15]

Figure 9.1 - Example Framework to Map Distribution of Compliance Costs (Robinson et al. 2016)[16]



A few key insights for EIA can be gleaned from these frameworks:

- **Differentiating between impacts that occur in the short- and long-run is important.** The short-run refers to the period in which only some factors of production are variable (e.g., labor) while others are fixed (e.g., capital equipment), and consumers are constrained by existing household assets, commitments, and information. In policy contexts, the short-run is sometimes referred to as a transition period. The long-run refers to the period in which all factors of production are variable, the aforementioned consumer constraints are relaxed, and the economy returns to equilibrium (i.e., all prices and quantities have fully

14 Scarcity rents represent a measure of welfare: "This producer's surplus which persists in long-run competitive equilibrium is called scarcity rent." (Tietenberg 2006). For a discussion of scarcity rents created by environmental regulations through pollution restrictions and captured by firms in the form of higher profits, see Fullerton and Metcalf (2001). See Buchanan and Tullock (1975) for a discussion of the potential for scarcity rents under a quota or a cap-and-trade policy where permits are distributed for free.

15 For an interesting example, see Fullerton (2011) where this framework is applied to a specific environmental policy (a carbon permit system) by linking measurable outcomes to welfare changes.

16 Reproduced with author permission.

adjusted to the new regulation). There are likely to be different implications for the economic impacts of a policy in the short-run compared to the long-run. For example, in the long-run, consumers are better equipped to switch to substitute goods, and firms are better equipped to switch to producing different outputs and to make entry and exit decisions. These time frames also have different implications for workers (see Section 9.5.1.4).

- **The distribution of impacts among market participants depends on the nature of the affected market(s).** Market characteristics including the extent of competition and the elasticity of demand relative to the elasticity of supply determine the allocation of impacts among consumers, labor and owners of firms, capital, and other resources. All things equal, competitive markets pass regulatory costs through to consumers to a greater extent than markets in which firms have monopoly power. Firms in very competitive markets do not earn excess profit and have no choice but to pass on costs if they want to stay in business. Of course, the reduced quantity demanded at higher prices may force them to close. Firms with market power have incentive to absorb a portion of regulatory costs since raising the price they charge reduces the quantity consumers demand of their products and reduces profits.[17] Relative elasticities are also important. In an imperfectly competitive market, the portion of the cost borne by producers increases with a greater elasticity of demand relative to elasticity of supply (and the portion borne by consumers increases with a greater elasticity of supply relative to elasticity of demand).[18]
- **Impacts may differ within market participant categories.** Substantial heterogeneity of a regulation's impacts is often experienced within groups. In practice, firms and their circumstances are not identical, so compliance may be more burdensome for some firms than for others.[19] For example, small firms may have fewer units of production over which to spread compliance costs, or some firms may have technologies that are more expensive to adapt to regulatory requirements. Similarly, consumers and their circumstances are not identical. People purchase varying bundles of goods and therefore will not be uniformly affected by price changes. Also, the same incremental change in consumption will affect individuals differently depending on their baseline levels of consumption; those with higher levels will value a small change in consumption less (referred to as the diminishing marginal utility of consumption). Industries, factors of production and other market participant categories can be affected differently as well. In Section 9.5, we discuss the conditions associated with divergent impacts for each impact category.

This section has discussed frameworks that shed light on the potential distribution of compliance costs. Several papers also consider the distribution of health benefits or environmental quality (e.g., Fullerton 2016; Robinson et al. 2016; Pearce 2006). For example, Robinson et al. trace the effects of hazard reduction on changes in human risks and the valuation of those changes. See Section 10.2.1 in Chapter 10 for a discussion of this literature.

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17 For a discussion of economic impacts on a representative firm and on the market, in a supply and demand model with perfect competition and under monopoly, see Tietenberg (2006), pp. 510-516.

18 See Fullerton and Metcalf (2002).

19 Heterogeneity in impacts may also be the result of regulatory design (e.g., differentiation of standards by facility vintage). This possibility is discussed in Section 9.5.1.

9.3.2 Disaggregated Welfare Effects

An analysis that disaggregates welfare effects (social costs and benefits) and transfers across relevant groups is a worthwhile goal. The analyst could estimate the net welfare changes experienced by each affected group in an economy, which in principle, if all regulatory consequences by group were fully described, might obviate the need for an EIA. In practice, however, many obstacles prevent a complete distributional analysis of welfare effects. For instance, at the EPA it is typical that the social costs and benefits of an environmental regulation are estimated for different groups. The former is usually estimated for firms that must comply with regulatory requirements, but the ultimate incidence of those compliance costs among owners, workers, and consumers (as costs are passed through to profits and prices, for example) is not typically estimated. Social benefits are estimated for individuals experiencing changes in environmental risks or conditions. Sparse information regarding the overlap between the groups bearing the costs and experiencing the benefits makes calculating disaggregated net welfare effects particularly challenging.

A different possibility to achieve disaggregated welfare effects would be converting the economic impacts included in an EIA experienced by different groups into welfare changes and summing across effects. Unfortunately, current models and data prevent such a detailed exercise. Consider business closures, for example. They might decrease profits to owners and upstream firms and cause workers to become unemployed. One would need to have information about effects on upstream firms (i.e., those that would be affected and by how much) as well as information on affected workers (e.g., the forgone wages of unemployed workers, the length of time they remain unemployed and their wages once they are successfully re-employed). Such detailed information is typically not available. Text Box 9.1 discusses the inherent difficulties of estimating social welfare effects associated with employment impacts.

Finally, to estimate group specific social benefits, analysts would need group-differentiated estimates of willingness to pay for the variety of environmental quality changes caused by EPA rules. While the existing literature contains evidence of variability in willingness to pay for public environmental goods among income (and other) groups, it does not contain a full suite of such estimates[20]; and the use of any specific estimate would be controversial without significant public review.

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20 For discussion and examples, see Banzhaf et. al (2019) and Chapter 10 on willingness-to-pay in the environmental justice literature; Banzhaf and Walsh (2008) for empirical evidence of household sorting in response to toxic air emissions; and Ito and Zhang (2020) for evidence of variable WTP for clean air in China.

Text Box 9.1 - Labor Impacts and Benefit Cost Analysis

In a benefit-cost analysis, some portion of changes in employment may also affect social welfare, but there are many theoretical and practical challenges to accounting for them. One challenge is how to estimate transition costs to workers experiencing involuntary job loss and unemployment. Including all resulting earnings losses would overstate social costs if they are transfers of economic rents - for example, if displaced workers were highly paid relative to their productivity (Hall 2011).

In addition to earnings losses, workers may incur transition costs due to relocation across labor markets, health impacts or other impacts on well-being that are not well-measured (Smith 2015; Kuminoff, Schoellman, and Timmins 2015). Transition costs may be higher during a recessionary period, when overall labor demand is already reduced due to nationwide declines in production, which can lengthen the time needed to locate new employment (Bartik 2015). These costs may be higher for certain categories of workers such as those whose skills are specially adapted for the sector experiencing reduced labor demand. For example, effects may differ by workers' age. For involuntary job loss, older workers with more human capital may face larger earnings losses for fewer years of remaining labor force time in their careers than otherwise similar workers who are young. Older workers experiencing involuntary job loss may have access to more resources from lifetime earnings, private insurance or access to social programs. Otherwise, similar younger workers may face larger costs because capital market imperfections prevent borrowing against their future lifetime earnings.

Likewise, quantifying changes in health or welfare due to an environmental regulation that affects workers, for example by improving their productivity or their ability to work, is challenging. An emerging literature documents these benefits; for reviews see Aguilar-Gomez et al. (2022) and Graff Zivin and Neidell (2013, 2018). These are just some of the issues to consider regarding potential welfare effects of labor impacts. Economists do not yet have a unified theory that incorporates employment impacts measured as social welfare effects into benefit cost analysis. For discussions, see Hall (2011), Ferris and McGartland (2014), and Smith (2015) who conclude that more work is needed in this area.

With caution, we also mention an analytic construct for further considering net welfare by detailed groups. A Social Welfare Function (SWF) establishes criteria under which efficiency and equity outcomes are transformed into a single metric, making them directly comparable. To do this, SWFs make assumptions regarding how society places different values on incremental changes in measures of well-being across individuals or groups (see Adler 2012, 2019, for a discussion). OMB (2023) outlines an option to implement a SWF in which individual- or group-specific WTP estimates are weighted differently. The weights assign lower values to incremental increases in consumption accruing to individuals with higher baseline consumption relative to people with lower baseline consumption (to account for diminishing marginal utility of consumption).[21] Implementation of this approach requires estimates of costs and benefits for each individual or each income group conditional on their baseline income and cannot rely on estimates of the average WTP across the whole population. Such average estimates are common in analyses of environmental regulations - EPA's estimate of the value of statistical life is an example. OMB's optional approach reflects one possible SWF; however, given its subjective nature, there is no clear

21 Please see OMB (2023) Section 10.e. for a detailed explanation.

consensus in the literature regarding how to value different distributions of welfare improvements. For these reasons, SWFs are not currently recommended when conducting regulatory analysis at the EPA.

Despite an inability to estimate the net welfare effects experienced by different groups affected by regulation, estimates of economic impacts improve understanding of the pathways through which welfare changes can occur, e.g., through business closures, or by restructuring markets or by increasing housing values in a community. Impact measures may also be useful for identifying individuals who might be strongly affected — for example the firms likely to close; whereas net welfare changes among groups might average out such strong effects so that their severity is overlooked. In addition, certain impact categories are examined to respond to statutory and executive order directives. Instead of focusing directly on welfare effects, this chapter provides information for qualitatively and quantitatively assessing changes in a wide variety of economic impacts that are expected to have an effect on welfare.

9.4 Analytic Components of an Economic Impact Analysis

An EIA should develop a profile of baseline conditions among groups expected to experience important effects of the rule. These are the conditions occuring in the absence of the rule or policy over the period of analysis. For example, the profile could include the number of regulated firms, their average size, and their average profitability. These metrics would be estimated for the year the rule takes effect and for the remaining timeframe of analysis. An EIA may also include two additional components: a preliminary analysis to screen for the magnitude of incremental impacts and an in-depth examination of expected important impacts. For each component of an EIA, analysts should highlight key analytic limitations and uncertainties. This section discusses the baseline profile, the preliminary analysis, and the in-depth examination, and identifies potentially useful data sources.

9.4.1 Baseline Profile

An EIA should develop a baseline profile that describes the industries, consumers, workers, or other groups that are expected to experience important incremental effects of a regulation.[22] The profiles will overlap with baseline profiles developed for other components of a regulatory analyses, such as the cost analysis.

The effects of some regulations may extend beyond participants in directly regulated markets, affecting, for instance, upstream or downstream markets, or complementary or substitute product markets. Often the markets involved in pollution control activities are affected. We will refer to the latter as the environmental protection sector and note that it may overlap with upstream markets.

The following information can contribute to an industry profile:

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22 For more about how to define and describe baselines, see Chapter 5. For more about developing a baseline for governments or non-profit organizations, see Section 9.5.

- The affected North American Industrial Classification System (NAICS) industry codes (NAICS is the standard used by federal statistical agencies in classifying business establishments);[23]
- Industry summary statistics, including total employment, revenue, costs, number of establishments, number of firms, size of firms, and race and gender profile of firm owners and workers;
- Baseline industry structure, including competitive structure, market concentration and degree of vertical integration within the industry;
- Characteristics of supply and demand (e.g., relative elasticities);
- Industry trends including growth rates, expected changes in technology and financial conditions;
- Openness to and reliance on international trade;
- Pre-existing environmental and other regulations and associated compliance behavior;
- Barriers to entry; and
- Diversity of production technologies among firms.

The baseline socioeconomic characteristics of groups expected to experience consequential economic gains or burdens due to a regulation are also important and may include consumers, workers, business owners, shareholders, renters, community members and others. Attributes to consider include:

- Income and poverty levels;
- Age distribution;
- Employment status;
- Community characteristics such as unemployment rate;
- Geographic location and mobility; and
- Pollution burdens.

The potential relevance of these market conditions and socioeconomic characteristics within the context of a specific impact category is discussed in Section 9.5.

9.4.2 Preliminary Analysis

During the early stages of regulatory analysis, a preliminary analysis to explore the potential for important impacts can be useful and may be as simple as systematically thinking through the expected impacts of a regulation and qualitatively describing them. When data are sparse, it may still be possible to roughly estimate some regulatory impacts. For example, to screen for significant impacts on small businesses, analysts can compare a rule's estimated annualized costs per regulated facility to estimated annual revenues of affected small facilities to determine whether the ratio of regulatory costs to facility revenues violates established thresholds.[24]

While the EPA has established thresholds that suggest when impacts on small entities are significant, in most cases the criteria for when an impact warrants additional analyses are not well defined and may depend on the condition of the economy. For example, during an economic

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23 For more information on classifying industries by NAICS codes, see https://www.census.gov/eos/www/naics/.

24 See Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act (U.S. EPA 2006a).

recession, impacts on workers may be a concern. Or, the timing of regulatory impacts may be relevant, including the period of anticipation of an upcoming compliance date, and whether effects are expected to grow or diminish or affect different groups over time. The context or location within which economic impacts are experienced is important. For example, reduced demand for labor in a small town with declining job opportunities might have bigger labor market impacts than in a larger city with abundant work opportunities. Or, when a trade exposed industry is the subject of regulation, there may be concerns regarding potential loss of domestic market share. Finally, if analysts suspect important impacts beyond directly regulated industries, the scope of analysis can be broadened, even if data and tools permit only qualitative assessments.

9.4.3 In-Depth Examination

Analysts may conduct an in-depth examination of the impact categories identified as likely to be important by the preliminary analysis. Substantial adverse impacts deserve special attention. If possible, a partial equilibrium analysis of affected markets can yield greater insights into impacts relative to an engineering cost analysis alone.[25] For example, with information on demand and supply elasticities in affected markets, analysts can move to a more refined analysis that examines the pathways through which costs would travel (e.g., consumer prices versus producer profits and input prices including wages). With regional- and firm-specific demand and supply information, analysts might also be able to shed light on how impacts vary across regions and firms. It may also be possible to link together several sector-specific partial equilibrium models with a multi-market model to examine linked impacts on regulated and related markets. If appropriate, a general equilibrium model can offer insights into impacts on a broad spectrum of markets and groups across the economy (see Section 9.5.5).

9.4.4 Data

Analysts may have access to proprietary data or detailed plant-level data (which may be confidential business information) collected through the rulemaking process that can be leveraged in an economic impact analysis. However, often data must be sought elsewhere. Table 9.2 describes available data sources that might be useful for analyzing economic impacts. The right-hand column gives examples of groups or impact categories under analysis for which each data source might be useful. Note that quantitative estimates of some economic impacts may not be possible because of inadequate household-, firm- or community-specific data (including elasticity estimates). Data that are available are often aggregated to the sector, or jurisdiction, level.

25 For a discussion of partial equilibrium and other market and engineering models, see Chapter 8 on Analyzing Costs.

Table 9.2 - Examples of Available Data Sources for Analyzing Economic Impacts of EPA Regulations

Source	Examples of types of data	Examples of relevant groups/impact categories
U.S. Bureau of Labor Statistics Consumer Expenditure Survey: https://www.bls.gov/cex/	Expenditures, income and demographic characteristics of U.S. consumers.	Consumers, Communities.
U.S. Bureau of Labor Statistics - Current Employment Statistics: https://www.bls.gov/ces/	Establishment-level estimates of nonfarm employment, hours and earnings by industry.	Sectors or Industries, Producers, Labor, Communities.
U.S. Bureau of Labor Statistics – Current Population Survey: https://www.bls.gov/cps/	Household level data on employment, unemployment, persons not in the labor force, hours of work, earnings and other characteristics.	Labor Communities.
U.S. Bureau of Labor Statistics – Producer Price Index: https://www.bls.gov/ppi/	Index of producer output prices, by detailed industry.	Sectors or Industries, Producers.
U.S. Census Bureau – Longitudinal Employer-Household Dynamics: https://lehd.ces.census.gov/	Statistics on employment, earnings and job flows at detailed levels of geography and industry for different demographic groups.	Sectors or Industries, Labor, Producers, Government entities.
Published research specific to an industry or sector.	Demand and supply elasticities, regional supply and demand information, and other specific estimates of interest.	Sectors or Industries, Consumers, Producers.
University of Wisconsin – Wisconsin National Data Consortium: http://windc.wisc.edu/	Open-source datasets for economic analysis, for U.S. states and counties, with state, sector and region economic activity.	Sectors or Industries, Consumers, Producers, Government entities.
U.S. Census States & Local Areas: https://data.census.gov/all?g=010XX00US$0400000	Demographic and socioeconomic information.	Consumers, Government entities.
U.S. Census State and County Quickfacts: https://www.census.gov/quickfacts/fact/table/US/PST045221	Demographic and socioeconomic information.	Consumers, Government entities.
U.S. Census Bureau – American Housing Survey: https://www.census.gov/programs-surveys/ahs.html	Data on the housing and construction industry, homeownership, and characteristics of homes.	Housing and Construction Industry, Consumers, Government entities, Communities.
U.S. Department of Housing and Urban Development Aggregated USPS Administrative Data on Address Vacancies: https://www.huduser.gov/portal/datasets/usps.html	Occupancy status.	Communities, Government entities.

Source	Examples of types of data	Examples of relevant groups/impact categories
U.S. Census Bureau – American Community Survey: https://www.census.gov/programs-surveys/acs	Detailed population and housing information, by community	Sectors or Industries, Labor, Producers, Consumers, Government entities, Communities.
Trade Publications and Associations	Market and technological trends, sales, location and ownership changes.	Sectors or Industries.
U.S. Census Statistics of U.S. Businesses: https://www.census.gov/programs-surveys/susb.html	National and subnational economic activity by enterprise size and establishment industry.	Producers, Small businesses, Non-profits, Government entities.
U.S. Bureau of Economic Analysis: https://www.bea.gov/data	Economic statistics on U.S. production (e.g., GDP), consumption, investment, exports and imports, and income and saving. National, Regional, Industry and International economic accounts	Sectors or Industries, Producers, Labor, Consumers, Government entities, Communities, International competitiveness.
U.S. Census Bureau – Annual Survey of Manufacturers: https://www.census.gov/programs-surveys/asm.html	Statistics for manufacturing establishments Discontinued after 2021, transitioned to the Annual Integrated Economic Survey: https://www.census.gov/programs-surveys/aies.html	Manufacturing sector, Producers.
U.S. Census Bureau – Economic Census: https://www.census.gov/programs-surveys/economic-census.html	Sector-level sales, value of shipments, number of employees and establishments, value added, cost of materials, capital expenditures, household and community characteristics	Sectors or Industries, Producers, Consumers, Communities.
U.S. Department of Commerce Industry & Trade Outlook Periodically published book – most recently in 2000	Industry, trends, international competitiveness and regulatory events.	Sectors or Industries.
New York University. Margins by Sector: http://pages.stern.nyu.edu/~adamodar/New_Home_Page/datafile/margin.html	Profit margins: gross income and net income based.	Sectors or Industries, Producers, Businesses.
Internal Revenue Service. Statistics of Income Bulletin https://www.irs.gov/pub/irs-soi/16winbul.pdf	Tax receipts, deductions and profits.	Sectors or Industries, Producers, Businesses.
Dun & Bradstreet Information Services: www.dnb.com	NAICS code, address, facility and parent firm revenues and employment.	Sectors or Industries, Producers, Businesses.

Source	Examples of types of data	Examples of relevant groups/impact categories
Standard & Poors: www.standardandpoors.com	Quarterly financial information for publicly held firms, line-of-business and geographic segment information and Standard and Poor's (S&P) ratings.	Sectors or Industries, Producers, Businesses.
Value Line Industry Reports: http://www.valueline.com/Stocks/Industries.aspx	Industry overviews, company descriptions and outlook, and performance measures.	Sectors or Industries, Producers, Businesses.
Securities and Exchange Commission Filings and Forms: https://www.sec.gov/edgar.shtml	Income statement and balance sheet, working capital, cost of capital, employment, regulatory history, foreign competition, lines of business, ownership and subsidiaries, and mergers and acquisitions.	Sectors or Industries, Producers, Businesses.
U.S. Energy Information Administration – Electricity Data: https://www.eia.gov/electricity/data.php	Statistics on electric power plants, capacity, generation, fuel consumption, sales, prices and customers.	Energy sector and subsectors (e.g., oil, natural gas, coal, nuclear energy sources), Customers.
United States Utility Rate Database (URDB)[26] https://openei.org/wiki/Utility_Rate_Database	Rate structure information for electric utilities in the United States. The URDB includes rates for utilities based on the authoritative list of U.S. utility companies maintained by the U.S. Department of Energy's Energy Information Administration.	Energy sector and subsectors (e.g., oil, natural gas, coal, nuclear energy sources), Customers.
U.S. Department of Commerce Pollution Abatement Costs and Expenditures Survey: https://www.census.gov/econ/overview/mu1100.html	Pollution abatement costs for manufacturing facilities by industry, state, and region. Data is limited to annually from 1973 to 1994, with the exclusion of 1987; and 1999 and 2005.	Sectors or Industries, Producers, Businesses.
S&P, Moody's and Fitch state and city bond ratings.	Financial strength indicator.	Government entities.
U.S. Department of Commerce Census of Governments: https://www.census.gov/econ/overview/go0100.html	Revenue, expenditures debt, employment, payroll, assets for counties, cities, townships and school districts.	Government entities.
United Nations, International Trade Statistics Yearbook.	Foreign trade volumes for selected commodities and major trading partners.	Sectors or Industries, Producers, Businesses.

26 Rates are posted annually by the National Renewable Energy Laboratory (NREL), under funding from the U.S. Department of Energy's Solar Energy Technologies Program, in partnership with Illinois State University's Institute for Regulatory Policy Studies.

Source	Examples of types of data	Examples of relevant groups/impact categories
U.S. International Trade Commission: https://www.usitc.gov/research_and_analysis.htm	Investigative Reports.	Sectors or Industries, Producers, International Trade
Global Trade Analysis Project: https://www.gtap.agecon.purdue.edu/databases/default.asp	Global data base describing bilateral trade patterns, production, consumption and intermediate use of commodities and services.	Sectors or Industries, Producers, International trade.

9.5 Impact Categories

This section provides guidance for assessing specific impact categories. Categories discussed are not mutually exclusive; rather, they have a high likelihood of overlap. For example, impacts on producers (employees and owners) likely overlap with impacts on the communities where they are located. Impact categories discussed in this section are:

- Producers and factors of production.
- Consumers.
- Communities.
- Governments and non-profits.
- Economy-wide.
- Benefits of improved environmental quality or health.

The discussion that follows usually considers the impacts of new compliance activities. However, it is also relevant to reductions in compliance activities which generally would produce impacts going in the opposite direction.

9.5.1 Impacts on Producers and Factors of Production

Compliance activities typically increase production costs to regulated industries. This may affect many different impact categories which are listed below and discussed in this section:

- Production.
- Profitability and plant closures.
- Small businesses.
- Labor.
- Land and capital.
- Related markets.
- Energy sector.
- Competitiveness.

Effects may vary by industry or firm characteristics, production technologies, pollution intensities, policy design and more. There may be different effects in the long-run versus the short-run, and according to whether one-time, ongoing, or transitional costs are being considered. Ongoing costs are to maintain the newly achieved state of environmental quality. Transitional costs stem from adjusting from one state of environmental quality to another (Baumol and Oates 1988).

Consideration of the effect on small businesses is mandated by statute; consideration of impacts on the energy sector is directed by executive order.[27]

If regulatory costs are small and/or distributed widely, there may be negligible impacts on producers. However, even if the average impact across firms is small, some producers, such as those facing the highest abatement costs, may be substantially affected. The following subsections discuss how to assess impacts on producers and factors of production.

9.5.1.1 Impacts on Production

In response to substantial regulatory costs, the supply curve in the directly regulated market may shift upward in the area near the market price which typically leads to higher prices and lower output.[28] Reductions in industry output are usually driven by a mix of increased and lowered operating rates at existing plants, closure of some plants and/or reduced future growth in production relative to the baseline. This section discusses circumstances that influence changes in output at the firm or facility (for firms that own more than one plant) level. Such changes can be combined with industry characteristics such as the number and size or regional distribution of firms to assess total changes in production.

At least two conditions can cause environmental regulation to have different impacts across firms, and lead to changes in both the number and size of the average firm (Tietenberg 2006). The first is significant heterogeneity in firm or facility cost structures; the second is regulatory requirements that differ depending on firm characteristics.

Variability in cost structures can cause variation in the magnitude of regulatory costs and, while not always the case, can lead to differences in the magnitude and direction of changes in output across producers. For example, total industry output may decline or shift from the highest cost plants to more efficient competitors. To better understand the extent of heterogeneity in how firms might adjust production in response to regulatory requirements, a profile of baseline conditions is useful. If available, detailed industry, firm or plant-level information may provide insights into how production processes and baseline costs might vary across facilities and how this variation might lead to different incremental costs of a regulation. For example, the ease with which facilities can accommodate pollution control equipment may vary, or there could be variability in the ability to substitute less hazardous chemicals for more toxic ones. Some firms may have to finance abatement equipment and activities. For such firms, the cost and availability of financing can affect production decisions.[29] Ultimately, what analysts will need are the differences across firms in post-regulatory costs. Firms may be able to maintain or even increase production levels if after absorbing compliance costs, their production costs fall below the highest cost firms. Or they may decrease

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27 See Chapter 2 and Section 9.2 which refers to the RFA as amended by the SBREFA, and to EO 13211.

28 In the post-policy equilibrium, if the production costs of the marginal firm are not notably affected by the regulation, then it is possible that the production and price effects can be de minimis even if inframarginal firms face notable compliance costs.

29 Analysts should carefully consider private market interest rates and other financing costs that firms might face. A detailed consideration is presented in chapter 10 of the documentation for EPA's Integrated Planning Model (IPM) for the power sector. Financing costs are represented as the weighted average cost of capital in which firms finance projects with a combination of debt and equity. Merchant power providers are assumed to face higher financing costs than utilities (U.S. EPA 2024a). See also Section 6.4 of these Guidelines on selecting private discount rates.

production if, after absorbing compliance costs, their production costs are among the highest in the market.

The second cause of variable impacts across firms are variable regulatory requirements. Vintage-based regulations that vary with the age of facilities may differentiate between existing and future pollution sources, with future sources regulated more stringently. In other cases, firms in regions with high baseline pollution may face stricter emission controls.[30] In general, regulatory requirements that vary by firm characteristics will shift economies of scale and can affect the distribution of output among firms as well as firms' average level of output. For example, firms may respond to policies that differ across plant locations by relocating production to a less-regulated area within the U.S. The greater the degree to which firms take advantage of this ability to shift production across space to reduce compliance costs, the more likely it is that overall domestic production does not change substantially. The outcome could be plant closure(s) and accompanying plant opening(s) due to relocations, with distributional effects on affected areas.[31] Shifts in production from domestic to foreign sources can also occur and are discussed in more detail in Section 9.5.1.8.

9.5.1.2 Impacts on Profitability and Plant Closures

Regulatory costs can reduce profits and increase the possibility of plant closures. The industry profile (see Section 9.4) describes baseline industry growth and financial conditions at regulated firms. To assess changes in profits due to a regulation, analysts should compare the expected change in market price to the change in production costs after accounting for compliance activities. This increment should be multiplied by expected changes in output to estimate how profits change.

Industries and firms that are relatively profitable in the baseline will be better able to absorb any new compliance costs that are not passed on to consumers. In cases where facilities have different baseline pollution controls or different production technologies, those with lower costs after meeting a new environmental standard will be better able to maintain profitability relative to other firms and may increase their market share. These firms may even be able to increase profitability if their costs of compliance increase by less than the increase in market price.

Discussing the likelihood of baseline closures improves understanding about the likelihood of closures attributable to the regulation.[32] Note that vertically or horizontally integrated facilities might not be viable as stand-alone operations but may continue to operate based on their contribution to the business line.

If pollution restrictions limit production of industry output, profitability may be affected. There may be different profitability impacts for new versus existing firms. This may be the case, for example, with vintage-differentiated regulation that imposes less rigorous pollution controls on

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30 The firms experiencing less-stringent regulation might be more likely to see expanding market shares relative to their counterparts, though some empirical evidence suggests this is not the case (Tietenberg 2006 citing Pashigian 1984 and Pittman 1981; Greenstone 2002).

31 Shadbegian and Wolverton (2010) survey the plant location literature which suggests that firms reallocate production (Gray and Shadbegian 2010), plant entry (List et al. 2003), or plant exit (Kahn 1997) in response to environmental regulations.

32 For example, the EPA's documentation for its power sector model, IPM, includes detailed information on power plants that have made public announcements of future closures, and this information can inform a baseline analysis (U.S. EPA 2024a).

existing relative to new firms.[33] If market demand is increasing, new firms can enter but face higher costs which negatively impact profitability. Existing firms can benefit through newly created scarcity rents, with positive impacts on profitability. Over the long run, the likelihood of plant closures may change if older plants with higher emissions are kept in operation for longer than was expected in the baseline scenario.

Analysis of impacts on regulated firms' financial conditions involves the use of available financial data. Impacts can be assessed by examining direct compliance costs as a percent of a firm's average revenues, profits, or sales. An upper-bound assumption is that compliance costs are borne entirely by the regulated industry (i.e., none are passed through to consumers). When data allow, assessing the ratio of regulatory costs to profits is useful.[34] Due to data limitations, analysts may only have access to industry average revenues or sales. Calculating the ratio of full compliance costs to average firm revenues gives some sense of the magnitude of compliance activities relative to production activities without directly addressing the effect on profitability. When data on firm profits are available, the ability of firms to pass costs through to prices should be considered.

Additional challenging issues affect ex-ante analysis of the effect of compliance spending on profitability. First, economic models are simplified representations of complex economic systems. They can be useful for estimating effects on groups but often are not reliable predictors of firm or facility-level decisions.[35] Second, common simplifying assumptions about firm decision-making include *perfect foresight*, where agents know precise values for all economic variables in all future years, and *perfect information*, where precise values drive decision-making so that a one-cent difference between costs and revenues can be the difference between continued operation versus closure.[36] Such assumptions may perform well when describing aggregate behavior, but they often run counter to the everyday complex and uncertain decision-making by managers, which is remarkably difficult to model.[37] There is typically little information regarding the economic decision maker's expectations about the future (e.g., the firm's profitability, costs, revenues and market conditions) and how those expectations respond to new conditions, such as a new regulation. Indeed, many decisions are multi-faceted.

For example, management decisions about plant closures often result from the cumulative effect of multiple factors, such as financial distress, unfavorable market conditions and aging equipment, rather than any single factor such as a new environmental regulation. Finally, facility-specific, rather than firm-specific, financial information is preferred for assessing profitability and particularly for assessing the likelihood of plant closures. However, it is often difficult to find. For instance, while financial data for publicly held companies is available, it is often too aggregated to shed light on specific business practices or management decisions. For these reasons it is important

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33 See Tietenberg (2006) Chapter 21 for more discussion and for references to literature finding evidence of a new-source bias in environmental regulations.

34 Several sources in Table 9.2 provide information on industry profitability. See the table entries labeled, "New York University, Margins by Sector," and "Internal Revenue Service, Statistics of Income."

35 Some models use "model plants" to represent specific plant or unit types and solve a linear programming problem by choosing compliance strategies to minimize costs across the model plants (see Section 8.4.3).

36 This is referred to in the literature as the "penny-switching effect." See Krey and Riahi (2009).

37 The financial literature points to managers' individual characteristics and biases that can affect corporate decision-making, e.g., risk aversion, confident or pessimistic approaches, misestimation of financial market data, or loss aversion. For a brief survey of the literature on behavioral corporate finance, see Malmendier and Tate (2015).

for analysts to describe the main limitations of the analysis when evaluating the incremental impact of a regulation on firm profitability or the likelihood of plant closures.

9.5.1.3 Impacts on Small Businesses

The RFA requires agencies to define small business according to the Small Business Administration's (SBA) small business size standard regulations.[38] As another option, the RFA authorizes any agency to adopt 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. If adopted, the agency must publish the alternative definition in the Federal Register. The analytical tasks associated with complying with the RFA include a screening analysis for "significant economic impacts on a substantial number of small entities" (SISNOSE). The small businesses to be included in the analysis are those that are directly regulated; that is, those that are subject to the rule's requirements. If a small business does not have an obligation imposed directly by the regulation, then EPA guidance is that it should be excluded from the analysis.

Care should be exercised when distributing regulatory costs experienced by small businesses over multiple years. The annualization of compliance costs should rely on an estimate of the private discount rate that reflects the cost of capital. In general, the private discount rate will reflect the risk associated with the regulated entity in question. The cost of capital will also be affected by the ability of affected firms to deduct debt from their tax liability.

Some small businesses may be liquidity constrained and find it challenging to spread costs over multiple periods as they may face difficulty in raising external capital, including external debt. This issue may differentially affect women-owned, minority-owned, rural small businesses and very small businesses (firms with revenues less than $100,000 annually) (Federal Reserve Banks 2023, 2024, Morazzoni and Sy 2022, Fairlie et al. 2020, Cole 2020). For example, the Federal Reserve Banks (2023) analysis finds that even though startups by people-of-color are just as likely to apply for financing through financial institutions/lenders as are startups by White individuals, they are less likely to receive the requested funds. Analysts should consider whether the costs faced by liquidity-constrained small businesses are best modeled as being fully incurred during the year in which they are borne.

In order to determine SISNOSE, the EPA conducts a screening analysis for both proposed and final rules based on a percentage of sales as an economic impact for small businesses (a "sales test") (U.S. EPA 2006a).[39] While the analytic objective includes better understanding the effect of regulatory costs on profitability, on the likelihood of plant closure or plant cutbacks, and so on, in practice sparse data on profitability often limits an analysis to examining compliance costs as a percent of average firm revenues or sales. As discussed in Section 9.5.1.2, ex-ante analysis of the effect of compliance spending on profitability presents a difficult challenge.

"Small Entities" are defined by the RFA but "substantial number" is not specified. The EPA has broad guidelines including example thresholds for determining SISNOSE certification, but generally recommends three factors in determining "significant impact" and "substantial number":

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38 See U.S. Small Business Administration (2022) for SBA's size standards.

39 See also Chapter 2. For a discussion of the screening analysis for small governments and small non-profits, see Section 9.5.4.

1. Magnitude of economic impact that may be experienced by regulated small entities;
2. Total number of regulated small entities that may experience the economic impact; and
3. Percentage of regulated small entities that may experience the economic impact.

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. Further analysis examining other types of impacts, as discussed elsewhere in this chapter, in relation to small businesses, may provide additional information for decision makers.[40]

9.5.1.4 Impacts on Labor

Evaluation of employment impacts is required by many of the major environmental statutes.41 Impacts can vary according to baseline labor market conditions; employer and worker characteristics such as industry, occupation, skill-level and region; and the type of workforce adjustment or job transition. Employment impacts may occur in the regulated and environmental protection sectors, in upstream or downstream sectors, or in sectors producing substitutes or complements. As economic activity shifts in response to a regulation, typically there will be a mix of declines and gains in employment in different parts of the economy over time. This section focuses on labor demand[42] and on employment impacts measured as changes in employment levels. An employment impact analysis will describe both positive and negative changes in employment to present a complete picture. For most situations, employment impacts are assessed as part of an EIA, and should not be included in the formal BCA.[43] See Text Box 9.1, above, for a discussion of social costs and employment effects within BCA.

When the economy is at full employment as in long-run equilibrium, a regulation may reallocate employment among economic activities rather than affect the general employment level, and in the short-run may lead to transitional employment effects, such as workers involuntarily separated from their jobs (Arrow et. al. 1996, Hafstead and Williams 2020).

Economic theory of labor demand indicates that employers affected by environmental regulation may increase their demand for some types of labor, decrease demand for other types, or for still other types, not change it at all. A variety of papers have provided frameworks for understanding the employment impacts of regulation. Morgenstern et al. (2002) decompose the labor consequences in a regulated industry facing increased abatement costs. They identify three separate components. First, there is a demand effect caused by higher production costs raising market prices. Higher prices reduce consumption (and production) reducing demand for labor within the regulated industry. Second, there is a cost effect: as production costs increase, plants use more of all inputs including labor to produce the same level of output. For example, pollution abatement activities that require additional labor services to produce the same level of output.

40 See EPA Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act (U.S. EPA 2006a) for details on complying with the RFA.

41 Relevant statutes include the Clean Air Act, section 321(a); the Clean Water Act; section 507(e); the Toxic Substances Control Act, section 24; the Solid Waste Disposal Act, section 7001(e); and the Comprehensive Environmental Response, Compensation and Liability Act, (section 110(e).

42 See Section 9.5.6 and Chapter 7 Section 7.2. for examples of how environmental regulation may also affect labor supply through changes in worker health and productivity (e.g., Graff Zivin and Neidell 2012, 2013, 2018).

43 Except to the extent that labor costs are part of total costs in a BCA.

Third, there is a factor-shift effect: post-regulation production technologies may be more or less labor intensive (i.e., more/less labor is required per dollar of output). A different paper, Deschênes (2014), describes environmental regulations as requiring additional capital equipment for pollution abatement that does not increase productivity. This can be included in a labor demand model as an increase in the rental rate of productive capital. These higher production costs induce regulated firms to lower output and decrease labor demand (an output effect) as well as shift away from the use of more expensive capital toward increased labor demand (a substitution effect).[44] Berman and Bui (2001) discuss how affected firms' overall labor demand could increase, decrease or remain unaffected, depending, in part, on the labor-intensity of environmental protection activities needed for regulatory compliance compared to the labor-intensity of producing output. To study labor demand impacts empirically, a growing literature has compared employment levels at facilities subject to an environmental regulation to employment levels at similar facilities not subject to that environmental regulation; some studies find no employment effects, and others find significant differences. For a review of recent empirical evidence, see Gray et al. (2023).

In practice, an EIA evaluates potential changes and shifts, positive and negative, in employment levels by industry or other affected groups, and describes transitional employment effects for affected groups of workers. While employment impacts are measured as changes in employment levels by industry or affected group, workers affected by changes in labor demand due to regulation may experience a variety of transitional effects including job gains or involuntary job loss and unemployment (Smith 2015; Schmalensee and Stavins 2011; Congressional Budget Office 2011; and OMB 2015). Transitional, or adjustment, costs may occur as workers shift out of current employment and into other, potentially less desirable jobs (for example, jobs that are lower paying or in a less desirable location); or into unemployment; or exit the labor force sooner than otherwise (Walker 2013). Workers involuntarily displaced from declining industries or occupations, with long job tenure, or living in areas where labor mobility is low or unemployment is high, may be especially likely to face challenges in finding comparable re-employment (Baumol and Oates 1988). If displaced workers' job search challenges are significant and keep them from employment, then from a resource perspective, their labor is underutilized, similar to a stranded asset.[45] Involuntary job loss can lead to significant earnings losses for displaced workers, and may involve periods of unemployment as well as other impacts, such as negative health effects (Jacobson, LaLonde and Sullivan 1993; Sullivan and von Wachter 2009a, 2009b).[46] Text Box 9.2 discusses involuntary job loss, unemployment impacts and health and wealth effects.

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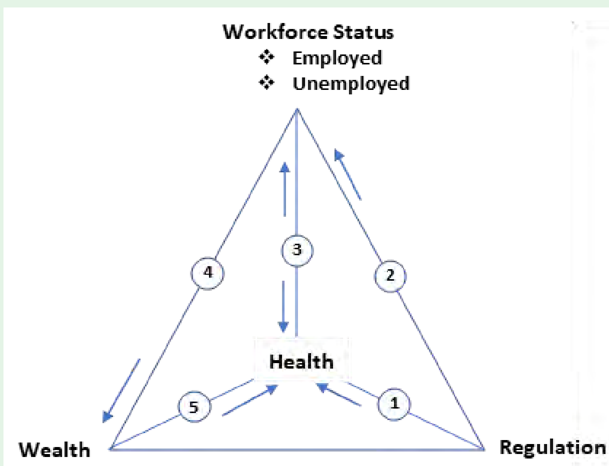
44 For an overview of the neoclassical theory of production and factor demand, see Chapter 9 of Layard and Walters (1978). For a discussion specific to labor demand, see chapter 4 of Borjas (1996). When using this theoretic framework, authors have conceptualized regulation as an increase in the price of pollution (Greenstone 2002; Holland 2012), an increase in the price of capital (Deschênes 2014), an increase in energy prices (Deschênes 2012), an increase in pollution abatement costs (Morgenstern et al. 2002), or with pollution abatement requirements modeled as quasi-fixed factors of production (Berman and Bui 2001).

45 Worker displacement can ultimately affect communities' provision of public services. See Morris et al. (2019) and Black et al. (2005) for examples of coal mining-reliant counties In Appalachia. See Section 9.5.3 for discussion of impacts on communities.

46 Involuntary job loss refers to job displacement that results from employer decisions and that is unrelated to worker performance, e.g. plant closings, mass layoff events and other firm-level employment reductions (Farber 2017; Sullivan and von Wachter 2009b; Chan and Stevens 2001).

Text Box 9.2 - Unemployment Impacts, Health, and Wealth

Empirical studies indicate unemployment is associated with increased mortality risk for those in their early and middle careers, but "whether unemployment is causally related to mortality remains an open question … and recent research has begun to focus on possible confounding, mediating and moderating factors" (Roelfs et al. 2011, p. 2). The figure below shows the complex relationships most related to *environmental* regulation between workforce status, regulation, wealth and health. As line (3) indicates, a bi-directional relationship exists between unemployment and health. Causality is difficult to identify for the unemployed population: increased mortality risk may be caused by unemployment itself, independent of pre-existing health status, or it may be caused by a decline in health resulting from a workforce status change (e.g., job loss, unemployment). The first causal pathway is potentially informative for regulatory analysis, but many studies lack detail to isolate it.



1. *Environmental regulation protects human health.*
2. *Employment impacts (e.g., workforce adjustment).*
3. *Workforce status affects health; health affects workforce status.*
4. *Workforce status affects wealth (e.g., unemployment reduces wealth).*
5. *Wealth affects health.*

A nascent economic literature uses detailed worker data to explore the effect of plant closures or mass layoff events on health. Sullivan and von Wachter (2009b) find increased mortality rates among displaced male workers with long job tenure in Pennsylvania and, in a study of displaced Austrian male workers, Kuhn et al. (2009) find that involuntary job loss negatively affected mental health. A study of plant downsizing in Norway found that displaced workers were more likely to utilize disability pensions than comparable workers in non-downsized plants (Rege et al. 2009). In a meta-analysis of studies on unemployment and health, Picchio and Ubaldi (2023) find on average a small negative effect of unemployment on health and, when the identification strategy relies on exogenous unemployment shocks like plant closure, the effect becomes smaller. Positive health impacts of moving from unemployment to a job may also exist (e.g., decreased depression) (van der Noordt et al. 2014).

The economics literature has found connections between wealth and health (indicated by line (5)). Sullivan and von Wachter (2009a) find that higher variability of earnings is associated with increased mortality. Dobkin et al. (2018) find that adverse health events measured by hospital admissions can lead to reduced earnings and increased risk of bankruptcy for those without health insurance.

The utility of these findings for regulatory analysis depends on whether involuntary job loss and unemployment are expected impacts. The prospect of such an impact is shown by line (2). If expected, the analysis may describe the likelihood of plant closures and employment impacts for affected workers. But analysts should use caution transferring published empirical estimates on adverse health impacts. Some studies use samples that may not correspond well to affected workers in the policy scenario and some lack detailed data on key worker characteristics (e.g., "involuntariness" of job separation (Sullivan and von Wachter 2009b); if job loss was health-related (Burgard et al. 2007).

While regulatory analyses may estimate employment impacts of regulations, it is challenging to identify associated job displacement at the firm- or plant-level. Both Curtis (2018) and Hafstead and Williams (2018) find workforce adjustments occur through reduced hiring rates rather than increased job separations. Reduced hiring rates could still imply that workers spend more time unemployed, though this may have a smaller impact than increased job separations. In a survey of firms experiencing mass layoffs, government regulation is rarely a stated reason (U.S. BLS 2011). More research is needed.

Workforce adjustments can be costly to firms as well as workers, so employers may choose to adjust their workforce gradually over time through natural attrition (retirements, voluntary separations) or reduced hiring, rather than incur costs associated with job separations (layoffs or other firm-level employment reductions). Curtis (2018) estimates changes in industry employment levels over time due to an environmental regulation and finds that changes occurred slowly through reduced hiring rates, and not through increased job separations. Hafstead and Williams (2018) find a similar result for the regulated sector, of employment levels decreasing through slow hiring and natural attrition rather than increased separations, when modeling a carbon tax.

As a result of shifts in the demand for labor, environmental regulation might also induce wage effects. However, firms generally avoid adjusting existing employees' wages downward (Walker 2013; Curtis 2018). Nominal wage rigidity has been attributed to many causes, not least is the potential impact of lowering wages on employee morale (Howitt 2002). Another factor suggesting very limited wage impacts in the specific context of environmental regulation, is that regulated firms are often a fraction of employers in affected labor markets and thus are not influential enough to affect industry wage rates (Berman and Bui, 2001).

The remainder of this section describes practical approaches to employment impacts analysis.

Estimating Labor Impacts: An employment impact analysis provides a baseline profile of potentially affected employers and workers, labor market conditions and possibly potentially affected communities. The analysis discusses or estimates potential changes or shifts in employment due to a regulation. Both positive and negative employment changes should be examined, including for example, possible employment impacts in the regulated sector as well as the environmental protection sector. When feasible, analysts can describe direct changes expected in the use of labor by the regulated sector for compliance requirements.[47] In cases where impacts are anticipated, and if data and modeling allow, analysts can describe employment impacts due to changes in production, revenues or expenditures by the regulated sector and potentially also by related sectors.

A baseline employment profile may include the size of the affected labor force, the degree to which affected labor markets are concentrated among few employers, the amount of labor mobility, job turnover, job search rates and the affected workers' regional or occupational unemployment

47 These labor costs (in dollars) are already included in the cost analysis of an RIA as they are costs to regulated firms (see Chapter 8 for more information). They can also be described within an employment impact analysis, and may be converted from dollar value labor costs to numbers of employees, or annual full-time equivalent (FTE), etc.

rates.[48] Recent employment trends may be relevant. Characteristics of affected workers, such as sector, industry, occupation, earnings, experience and job skills, may be described. If employment impacts are expected to be concentrated in certain communities, those communities could be characterized. Table 9.2 lists examples of possible data sources that may be helpful in developing a baseline employment profile.

To examine the incremental impacts of a regulation on employment, analysts should keep in mind that labor demand may be affected differently in the short-run compared to the long-run. For example, the RIA for the 2024 New Source Performance Standards for Greenhouse Gas Emissions from Certain Units includes employment impact estimates for the power sector both for short-run effects (e.g., construction-related employment needs) as well as long-run or recurring non-construction employment due to shifts in the use of fuels in electricity generation (U.S. EPA 2024b).

For many regulations, assessing employment impacts will be limited to a qualitative discussion. It will include the baseline profile described above, and the likely direction of change of employment levels in affected sectors and occupations. A discussion of any concentrated employment impacts, regionally or otherwise, would be useful. Information on the ability or limitations of workers to respond to shifts in labor demand should be considered.

A quantitative analysis may project changes in employment in affected sectors by occupation or among other groups of workers (e.g., by region). The quantitative estimates can use information from the compliance cost analysis if the labor requirements for expected compliance activities are provided. Examples of compliance activities include installation, operation and maintenance of pollution control equipment; as well as monitoring, inspecting, reporting and recordkeeping. For example, the RIA for the EPA's Safer Communities by Chemical Accident Prevention Final Rule included estimates of changes in the number of labor hours required for compliance activities among different occupations and for different sized facilities.[49] Its analysis of labor impacts examined how many total labor hours on average per year would be required for certain provisions, and whether new workers would likely be hired. The analysis discussed which rule provisions would likely require additional labor hours, the occupations of workers needed, and whether the work was short- or longer-term.

In quantitative analyses, aggregated labor hours should be converted to estimates of annual average job-years or full-time equivalents (FTEs).[50] When these estimates are small relative to average employment at a representative facility or firm, a reasonable assumption may be that existing employees or contractors would take on the tasks for regulatory compliance rather than a facility or firm adjusting the size of its workforce.

48 See, for example, Smith (2015) on local labor market conditions and unemployment, and Baumol and Oates (1988) Chapter 15, on reemployment prospects and consideration of workers in communities characterized by one or two large employers.

49 See U.S. EPA (2023a).

50 A job-year is not an individual job and is not necessarily a permanent or full-time job. Instead it is the work performed by one FTE employee in one year. For example, 20 job-years may represent 20 full-time jobs or 40 half-time jobs in a given year, or any combination of full- and part-time workers such that the total is equivalent to 20 FTE employees. In practice, for example, if the cost analysis for a regulation estimates a need for 1 million labor hours per year in the regulated sector to conduct compliance activities, this could be converted to approximately 480 job-years by dividing 1 million by the annual work hours for a full-time employee, which equals 2,080.

While transparent, the quantitative approach just outlined only addresses a subset of employment impacts as it does not address shifts in labor demand associated with potential changes in output in the regulated, or related, sectors. When a regulatory cost analysis project shifts in output due to compliance costs or shifts in the composition of production within the regulated sector (e.g., shifts in the electricity generation fuel-mix) a more detailed analysis may be possible. In these specific cases, analysts can estimate employment impacts by multiplying the change in output by the average amount of labor per unit of output (or per value of shipments) in the sector. This gives an approximation of the output effect, a potentially important type of employment impact.[51] The U.S. Census and U.S. Bureau of Labor Statistics (BLS) provide estimates of the units of labor associated with expenditures (or value of output/sales) at the industry-level. A limitation of this type of analysis is that in practice producer-level employment impacts will likely differ from aggregate, industry-level employment impacts. For example, relatively more efficient firms may expand output (and employment) to pick up the slack as less efficient producers contract (Jaffe et al. 1995; Tietenberg 2002; and Christiansen and Tietenberg 1985).

Detailed industry information is useful to develop disaggregated employment estimates for related sectors. For example, as part of estimating labor impacts in regulatory analyses of air pollution regulations affecting the electric power sector, the EPA examined coal mining by region.[52] The EPA combined estimates of changes in coal demand with detailed estimates of coal supply and regional coal mining productivity data available from the U.S. Energy Information Administration (U.S. EIA). Labor productivity differed significantly across geographic regions, e.g., in 2018 labor productivity in Virginia was 2.07 short tons of coal per labor hour, in Texas it was 6.73, and in Wyoming, it was 26.63 (U.S. EPA 2023b). This level of detail informed the analysis of employment impacts.

Approaches for estimating the employment impacts of environmental regulation are evolving. Analysts are encouraged to engage the EPA's National Center for Environmental Economics early in the process when developing a strategy for evaluating the employment impacts of a regulation. Analysts should describe the methods used in a quantitative employment impacts analysis – whether it analyzes changes in pollution abatement activities alone or combined with changes in production – and explain analytical limitations, which might include:

- Use of an estimation approach that produces *partial* employment impacts and does not fully measure all potential changes in regulated and related sectors.
- Application of average labor-to-cost or labor-to-output ratios instead of the change in labor expected in response to incremental increases or decreases in costs or production.
- Estimation of labor-to-cost, or labor to-output, ratios at the industry-level that reflect the labor component of pre-regulation costs or production rather than post-regulation costs or production. This is a limitation because such ratios can be influenced *by* the regulation.
- Use of available labor ratio data that may be for industrial sectors not well-aligned with the affected sectors.
- Heterogeneity of firm- or facility-level responses to regulation, especially those of marginal facilities operating at the tail end of productive efficiency, may be glossed over by labor ratio data typically available at the sector level only.

51 Data on labor per unit of output would be a proxy for the overall effect on labor demand in the regulated sector. These data are based on past production processes and therefore are not directly useful for measuring a substitution effect between labor and other productive inputs when compliance activities are required in the regulated industry.

52 U.S. EPA (2023b), "U.S. EPA Methodology for Power Sector-Specific Employment Analysis."

Cautionary Notes: Analysts should proceed with caution regarding the following approaches sometimes used to estimate quantitative employment impacts of regulation.

Transferring Certain Empirical Estimates: Morgenstern et al. (2002) estimated the effect of pollution abatement expenditures on the quantity of labor in four highly polluting and regulated industries. However, a later attempt to replicate and extend this research failed. Analysts should not rely on the empirical estimates from Morgenstern et al. (2002). Likewise, analysts should not rely on the estimates from Belova et al. (2013, 2015) as the authors "recommend that the EPA refrain from using these results until the underlying cause(s) for the implausibly large estimates in the employment effects found in Belova et al. (2013a) are uncovered and resolved."[53] We highlight Morgenstern et al. (2002) because of its prominence in the prior edition of the EPA's *Guidelines for Preparing Economic Analysis* (2010). The theoretical model in Morgenstern et al. (2002) remains valid.

Input-Output Analysis: As described in Section 8.3.4.1, input-output analysis can provide employment impact estimates. This type of analysis is most suitable for analyzing detailed sectoral impacts of regional, state, or local policies in the short term. In general, input-output models should not be used for estimating impacts of national regulations because they do not allow prices, production processes or technologies to adjust over time. As a result, they represent a very short-term response to regulation and are better equipped to represent the response of a single region to a small regulatory change which is not expected to affect prices.[54] They are of limited use for analyzing large regulatory changes or regulations that are national in scope.[55]

Plant Closures and Employment*:* Section 9.5.1.2 discusses difficulties in assessing the likelihood of plant closures given a dearth of data and a limited ability to model key factors, such as expectations of future profitability. Even in cases when estimates of the likelihood of plant closures are available, estimating employment impacts from them can be difficult. Employment impacts associated with plant closures may differ from the projected decline in plant output. Firms face labor adjustment costs, and, for example, multi-plant firms may choose to transfer workers, potentially those more skilled and experienced, to other locations (Ferris, Shadbegian and Wolverton 2014). Or, as noted above, production and employment may shift between firms, away from higher cost plants towards more efficient competitors. Such heterogeneity implies that employment impacts at the firm or plant-level can differ in direction from industry-level employment impacts. Analysts should consider these possibilities.

9.5.1.5 Impacts on Other Productive Factors: Land and Capital

In addition to labor impacts, environmental regulation can lead to changes in the demand for, and value of, other factors of production employed by regulated firms. Economists label these other factors of production as land (any natural resource), and capital (any man-made resource). In general, environmental regulation is expected to have varying effects across factors, and tracing

53 Quote is from Belova et al. (2015). Note that Belova et al. (2013a) inside the quote is identical with Belova et al. (2013), cited above.

54 Even for regional analyses, input-output models tend to overestimate impacts. "They typically include exogenous multipliers that magnify direct effects on output and employment based on the assumption that all new economic activity will recirculate within the regional economy. Input-output models tend to ignore displacement of workers or resources that might occur outside the region under analysis" (U.S. EPA 2011).

55 The underlying data can be useful for identifying related sectors, e.g., upstream and downstream.

impacts back to specific factors is difficult (Fullerton 2009). Estimating changes in the quantities demanded of broad categories of land and capital is more practical.[56] There are two separate and valid ways to represent the value of factors of production: earnings per period (also called rates of return) or asset values. The latter is the discounted present value of the future stream of earnings generated by the productive factor.

The relationship between changes in regulated firms' price and quantity of output, and changes in their factor demands or factor returns, can be complicated. In response to stricter environmental regulation, factors used intensively by the regulated industry might experience reduced demand and/or returns. If a unit of capital is not perfectly mobile, or a type of natural resource is taken out of production, it may lose value and impose a burden on the owners (Fullerton and Muehlegger 2019). For example, if a regulation induces firms to switch from high-carbon coal to lower emitting natural gas, then the value of coal will decline, and the stock value of coal-intensive businesses could decline as well. How fast an asset may return to production will affect the extent of burden. A coal mine that closes may become valueless as the land may be quite difficult to switch to a different use. It could even become a liability. Factors that are complements to pollution abatement might experience an increase in demand or returns; while those that are highly mobile with similarly valued alternative uses should hold their value. There are two general expectations for the long-run response to environmental regulation. One is for land and capital to shift away from high-emission activities toward lower emitting ones, including the environmental protection sector; another is for land and capital to shift towards less regulated uses. Regionally differentiated impacts on capital and land are possible when the stringency of pollution control varies by region.

To estimate how the costs of compliance are passed through to and distributed across productive factors, analysts need the cross-price elasticities between these factors. When this is not available, analysts can examine current production practices and the input biases of anticipated abatement activities to inform a qualitative discussion of likely impacts on productive factors.

In general, income earned from ownership of land and capital (or of firms) tends to make up a greater proportion of earnings for higher-income households. Thus, an increase in regulatory costs passed through to households via lower returns to capital tend to be progressive, placing a greater share of the burden on wealthier households.[57] The magnitude of the impact on owners and investors depends on the proportion of their portfolio affected by the change.

A different impact on factors of production stems from improved environmental quality which can be capitalized into the price of nearby land, and buildings (including housing). The increase in property or asset values accrues to the owners at the time of the improvement.[58] The degree to which the land and buildings are owner-occupied versus rented, and the degree to which the increased value is passed on in the form of higher rents, will influence who experiences positive versus negative impacts of the environmental improvement. If landlords increase rents to the point of forcing out renters, then the renters may experience transitional impacts from relocation activities. Identifying how owners and renters respond to improved environmental quality is a complicated exercise and quantitative analysis is challenging. A qualitative discussion can be useful. Related literature and modeling challenges are discussed in the final paragraphs of Section 10.2.1.

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56 Land and capital may also be rented or supplied under contract. When not owned by the regulated firm, the impacts are considered upstream, as discussed in Section 9.5.1.6.

57 For more details, see Rausch et al. (2011) or Fullerton and Metcalf (2002).

58 If land improvements are concentrated and substantial, there could be community-wide effects. See Section 9.5.3 for a discussion of impacts on communities.

9.5.1.6 Impacts on Related Markets

An environmental regulation may affect markets other than those that are directly regulated. Related markets may be positively affected, such as those in the environmental protection industry or those producing substitutes; or negatively affected, such as those producing complements, or those who are up- or downstream from the regulated industry (note that the environmental protection sector may overlap with upstream markets). 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 ones will be burdened. Consumers in the related markets may experience impacts as well (see Section 9.5.2). Downstream impacts may accrue to firms who purchase the regulated firms' outputs. In general, when analyzing related markets, analysts should consider the same potential impacts as for directly regulated markets.

If substantial impacts on related industries are expected, it will be useful to include firm sizes, profit margins, growth rates and more, in a baseline profile of the related industries. For instance, when the regulated sector sells an intermediate good or service (e.g., electricity), questions that might be relevant include: What proportion of the purchasing firms are small or face narrow profit margins? Are substitute inputs readily available? What proportion of the purchasing firms' spending goes to the regulated firms?

Partial equilibrium models that represent significantly affected, related markets may be useful, although sparse data and resources may limit their use. For regulations that are expected to substantially affect many related markets, an economy-wide model as described in Section 9.5.5 might be considered, though the additional conditions described there should also be satisfied.

9.5.1.7 Impacts on Energy Supply, Distribution or Use

EO 13211 (2001) directs agencies to prepare a Statement of Energy Effects for "significant energy actions," which are defined as significant regulatory actions (under EO 12866) that also are "likely to have a significant adverse effect on the supply, distribution or use of energy."[59] OMB guidance suggests that adverse effects could include any of the following:

- Reductions in crude oil supply in excess of 10,000 barrels per day;
- Reductions in fuel production in excess of 4,000 barrels per day;
- Reductions in coal production in excess of 5 million tons per year;
- Reductions in natural gas production in excess of 25 million mcf per year;
- Reductions in electricity production in excess of 1 billion KWH per year or in excess of 500 MW of installed capacity;
- Increases in energy use required by the regulatory action that exceed any of the thresholds above;
- Increases in the cost of energy production in excess of 1%;
- Increases in the cost of energy distribution in excess of 1%; or
- Other similarly adverse outcomes.

A regulatory action also may have adverse effects if it is likely to:

- Adversely affect, in a material way, productivity, competition or prices in the energy sector;

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59 See Section 2.1.6 and especially see OMB (2001).

- Adversely affect, in a material way, energy productivity, competition or prices within a region;
- Create a serious inconsistency or otherwise interfere with an action taken or planned by another agency regarding energy; or
- Raise novel legal or policy issues adversely affecting the supply, distribution or use of energy arising out of legal mandates, the President's priorities or the principles set forth in EOs 12866 (1993) and 13211 (2001).

For actions that may be significant under EO 12866 (1993), particularly for those that impose requirements on the energy sector, analysts must be prepared to examine the energy effects listed above.

9.5.1.8 Impacts on Domestic and International Competitiveness

Competitiveness impacts are regulatory impacts that change the distribution of market power among firms or sectors, either domestically or internationally. Unfair advantage may accrue to producers that are free from regulatory constraints, or that face less expensive regulation. Or, high fixed costs that are incurred to comply with environmental regulation may cause production to become concentrated among fewer firms, enhancing their monopoly profits over the long run. Regulatory constraints may differ among specific subsets of sectors or firms: existing versus new, or small versus large.[60] If some firms find it less costly to comply with a regulation, they may benefit competitively at the expense of other regulated firms. Analysts may wish to consider the extent to which production is shifted toward plants with higher-than-average productivity (Jaffe et al. 1995).

As with other impact categories, the extent to which a regulation leads to effects on competitiveness depends on the interaction between the regulated firms' absorption of compliance costs and their market structure.[61] In general, greater compliance flexibility is expected to reduce competitiveness effects.[62]

A first step to gauge the potential for competitiveness effects is the baseline profile of affected industries. The profile should identify which domestic and international firms compete with regulated entities, and their basic market structures. Do competitors face expensive environmental regulation? Is the output produced by regulated firms differentiated from that of competitors, potentially reducing impacts on competition? The literature suggests an increased likelihood of competitiveness effects for industries in which compliance costs are high relative to total production costs.[63]

Consideration of the impact of new environmental regulation in three key areas is particularly germane to competitiveness effects. First, lack of access to debt or equity markets to finance market entry, including regulatory costs, can represent significant barriers to entry.[64] Over the long run, this can change market structures and reduce competitiveness. Second, a regulation may have an

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60 Section 9.5.1.2 discusses the impacts of differentiated regulation that occurs when existing firms are regulated with greater leniency than new firms.

61 The importance of this interaction is discussed by Iraldo et al. (2011).

62 Evidence for this is presented by Iraldo et al. (2011) and Jaffe et al. (1995).

63 See Iraldo et al. (2011).

64 See the discussion about small business access to credit In Section 9.5.1.3.

impact on market concentration. A potentially useful measure of concentration is the Herfindahl-Hirschman index (HHI), which is the sum of the squares of the market shares of each firm in a given market. The U.S. Department of Justice uses the HHI to estimate changes in market concentration due to mergers and acquisitions. Post-merger HHI values that are below 1,000 are considered "unconcentrated," between 1,000 and 1,800 are regarded as moderately concentrated and above 1,800 are considered highly concentrated.[65]

Finally, the impact of regulation on the market position of domestic firms relative to their foreign counterparts is important. Domestic environmental regulations may have global economic implications because the costs of domestic producers increases relative to foreign producers.[66] Analyses of impacts on international competitiveness have been concentrated on the most pollution- or energy-intensive and most trade-exposed industries because they are most likely to face regulatory requirements and least able to pass compliance costs to consumers.[67] For example, in the context of unilateral climate policy, proposed legislation has focused on potential competitiveness impacts on trade-exposed domestic energy firms.[68] Quantifying these effects can be complex and may require a multi-country computable general equilibrium model. There are three classes of indicators of impacts on international competitiveness: the degree to which net exports change, the degree to which production shifts overseas (i.e., pollution haven effect), and the relative change in investment from domestic (regulated) producers to producers in other countries (Jaffe et al. 1995).

9.5.2 Impacts on Consumers

Measuring impacts on consumers is straightforward when environmental policy regulates consumer behavior. Requirements for automobile emissions tests or product bans such as the Final Rule on Methylene Chloride in Paint and Coating Removal for Consumer Use (U.S. EPA 2019) have impacts on consumers through time costs and fees. More frequently, environmental regulatory requirements are imposed on producers. In these cases, there is a less obvious potential impact on consumers as a result of producers passing through or transferring regulatory costs to purchasers of their products through increased prices. To understand cost pass-through to consumers, analysts typically examine the expected impacts of a regulation on prices of final goods. Also relevant are the characteristics of consumers purchasing the goods. Of course, firms may also be consumers of regulated products and as such are covered in Section 9.5.1.6 "Impacts on Related Markets."

New environmental requirements typically raise the cost of production in directly regulated industries, causing an upward shift in the market supply curve (that is, an increase in the price

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65 For more information, see https://www.justice.gov/atr/herfindahl-hirschman-index.

66 A related literature examines how differences in environmental regulation across countries, states, or sectors may result in increased emissions in less regulated countries, also called emissions leakage. For instance, see Bohringer et al. (2012) and Fischer and Fox (2012).

67 Carbone and Rivers (2017) discuss the impacts of environmental regulation on international competitiveness. In general, the literature has found relatively small effects (Jaffe et al. 1995; Aldy and Pizer 2015; Carbone and Rivers 2017). Jaffe et al. (1995) point out that concerns about industry competitiveness may also ultimately affect consumers as net exports decline and in the long-run imported goods become more expensive as the economy returns to balanced trade.

68 For a survey of the literature on competitiveness impacts of unilateral climate change policy, see Carbone and Rivers (2017). For a policy relevant discussion, see U.S. EPA (2016).

producers require for each quantity supplied). In response, consumers will do without or with less of the product, and/or pay a higher price, thus bearing some of the burden of regulatory costs.

A good starting point to analyze potential impacts on consumers purchasing output from the regulated sector is to gather information on the determinants of the elasticity of demand relative to the elasticity of supply for the affected goods. To gauge elasticity of demand, a useful consideration is whether the product is considered necessary by the purchaser, has many substitutes or its purchase makes up a substantial portion of the consumer budget.[69] Consumer impacts may be smaller if there are good substitutes that are comparably priced causing a high demand elasticity and smaller price change. There also may be small changes in output prices if compliance expenditures are low relative to total production costs.

To gauge elasticity of supply, analysts should assess how easily firms can increase or decrease production quantities. Information on the flexibility of capital equipment and buildings for shifting into different types of production would be useful; for example, understanding whether excess capacity can be used to produce comparably valued output.

The characteristics of the regulated industry also influence the share of costs passed on to consumers.[70] Under noncompetitive conditions, when firms in the regulated industry have market power, less cost-pass-through via prices is likely.[71] All else equal, if the same compliance requirements are placed on two markets that differ in terms of the degree of competition among firms, the one with less competition (e.g., due to barriers to entry such as restricted access to a scarce natural resource) will generally bear a higher share of those costs than the more competitive market. For firms with market power, raising price will lower sales; therefore, these firms will generally absorb some portion of regulatory costs (Tietenberg 2006). A market consisting of producers that have different cost structures, perhaps because they use different technologies or are of different sizes or ages, will lead to heterogeneity in the degree of pass-through of compliance expenditures. Finally, the structure of related markets may affect cost pass-through. For example, Preonas (2023) finds that distortions in the rail industry (an upstream market to the regulated one) led railroads to reduce coal markups when downstream power plant demand for coal declined. This suggests that regulatory costs faced by an industry may sometimes be partially absorbed by related markets, shielding consumers from price increases.

A qualitative discussion of the factors that can affect impacts on consumers may be useful. However, analysts may be able to locate empirical estimates of demand and supply elasticities. If possible, analysts should select elasticity estimates that reflect the focus of analysis. For example, to understand potential differences in the pass-through of regulatory costs into prices over time, analysts should examine estimates of elasticity in the short-run compared to the long-run; to understand differences in cost-pass-through across communities, analysts should examine regional demand elasticity measures.

Combining an estimated price increase with information on the share of the consumer's budget spent on the product will improve understanding of the impacts on households. There is a possibility that budget shares may vary substantially across consumers. Even if price increases are small, specific groups of consumers may still be affected if the product is a necessity for which low

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69 For more information about the determinants of elasticities, see Appendix A: Economic Theory, Section A.4.1 Elasticities.

70 For cases when government or non-profit organizations are the producers, see Section 9.5.4.

71 For example, see Ganapti et. al. (2020) on incomplete pass-through of energy input costs and imperfect competition in the manufacturing sector.

income households spend a substantial portion of their budget. For example, the share of income spent on energy or water by low income households is larger than for others, so energy or water price increases may affect them more.[72] This effect may be strengthened by the flexibility among higher income households to purchase substitutes with substantial upfront costs such as efficient appliances. However, it is also important to consider whether existing government programs may help mitigate the impact of price increases on consumers.

If consumer impacts are expected to be nonnegligible, information on affected consumers such as their age distribution, income level or residential location should be gathered to contribute to a baseline profile. Nationwide averages of these variables may be appropriate if consumers are broadly distributed across the country.

In some cases, assessing the impact of a regulation on consumers can be complex.[73] Analyzing policies with limited use patterns such as pesticides or paint removers may be challenging due to inaccessible or sparse data. Other complicating factors are associated with goods for which price- or rate-setting is complex. For example, to explore the extent to which proposed air pollution control costs will be experienced by different electricity consumers, the analysis would need to include information on how the policy affects consumers served by cost-of-service utilities, compared to deregulated electricity providers. Any assistance available for low-income or other consumers to offset rate increases is also relevant; as is variability in consumption patterns among categories of customers. If regulatory costs are large, economy-wide models may lend additional insight into how impacts affect consumers across the economy (see Section 9.5.5). Such models may also examine the interaction with existing government transfer programs.[74]

9.5.3 Impacts on Communities

Environmental regulation may have significant impacts on some specific communities or neighborhoods. Facility closures or production curtailments provide an example of locally concentrated economic impacts that could be acute in areas with limited economic opportunities. Displaced workers who live in such communities may be especially challenged as they search for comparable re-employment. Out-migration by displaced workers and families may cause reductions in the demand for products in the local goods sector. Tax revenues may decline with negative impacts on the provision and quality of community public goods. As the local economy shrinks, property values may decline. For example, regulation on coal-fired power plants could have negative impacts on coal-dependent communities. Mine closures and employment cuts can affect others in the community as the economic base and local tax revenues decline (Baumol and

72 The share of income spent on energy falls as income increases. Some studies have found that policies that increase energy prices are regressive, placing a greater burden on lower income households (e.g., Burtraw et al. 2009; Hassett et al. 2009; Williams et al. 2015). Other studies account for the indexing of transfer payments to inflation and find that the burden of a carbon tax is roughly proportional to permanent income, and so is neither regressive, nor progressive (Cronin et al. 2019). See Deryugina et al. (2019) for a discussion of some of these energy policy studies.

73 Cory and Taylor (2017) conduct a detailed analysis of spending by low-income households and explore the potential impacts on health spending caused by price changes induced by safe drinking water standards.

74 Some government transfer payments like Social Security are indexed to inflation and may provide some protection of purchasing power for lower income households.

Oates 1988; Black et al. 2005; Morris et al. 2019).[75] Impacts of changing business conditions that spread across industries in the same community are often approximated by "local multipliers" (Moretti 2010; Osman and Kemeny 2022). Such multipliers measure the broader changes in employment and wage income across communities. When appropriate, analysis of baseline economic conditions at the community level can help identify where the regulated industry is a key driver of the local economy, signaling the potential for multiplier effects.

Community-level health impacts can be exacerbated by a combination of localized concentrations of emissions from one or more sources, and community-wide exposure to other stressors. Locations with such combinations of risks are often referred to as "hot spots." and may reflect baseline conditions or be caused or aggravated by environmental regulation. Relevant issues to consider may include proximity to multiple pollution sources, specific exposure pathways, and drivers of differential susceptibility. For a full discussion see EPA's Technical Guidance for Assessing Environmental Justice in Regulatory Analysis (2024d).

Localized improvements in environmental quality, such as hazardous site cleanup, can reduce health risks and improve local property values thereby increasing the local tax base, and potentially in the long run, increasing investments in local public and private goods. If low-income residents are largely renters, then they could be burdened by increases in land values and subsequent increases in rent due to improved environmental quality, while at the same time property owners could enjoy higher rent payments. Property owners who reside in their own homes may be burdened through property tax increases. The higher property taxes and rental payments may cause some residents to move. The turnover may cause cost-of-living increases that further burden remaining low-income residents even beyond increased rents and property taxes. Low-income residents who relocate face transactions costs and do not experience the benefits of improved environmental quality.[76]

When localized impacts of environmental policies are expected, a baseline profile of affected communities will be informative.[77] Data on the unemployment rate, average income level, the poverty rate, whether the community is rural or urban, and its growth rate can help inform policy makers as to the relative disadvantages faced by affected communities.[78]

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75 Historic funding levels are being directed to coal mining and power plant communities to help rebuild and diversify their economies. The Interagency Working Group on Coal & Power Plant Communities & Economic Revitalization presents information on funding eligibility and more at https://energycommunities.gov/priority-energy-communities/#.

76 When a community that has experienced improved environmental quality undergoes a widescale turnover to higher income households, this Is described as environmental gentrification. For further discussion of gentrification in housing markets, see Section 10.2.1 of this guidance document, Section 8.2.5.1 of EPA's Handbook on the Benefits, Costs, and Impacts of Land Cleanup and Reuse (U.S. EPA 2011), and/or Banzhaf and McCormick (2012).

77 For details regarding examining environmental justice communities, see Chapter 10.

78 For a discussion on contributors to higher susceptibility, see EPA's Technical Guidance for Assessing Environmental Justice in Regulatory Analysis (U.S. EPA 2024c), Section 4.2, which addresses susceptibility or vulnerability within groups such as communities.

9.5.4 Impacts on Governments and Non-Profits

State and local governments and their residents, and non-profit organizations may incur costs or bear the burden of costs from EPA regulations. The frameworks and impacts discussed above apply to private markets. Governments and non-profits are distinctive because they are not motivated by profits. Analysts should consider potential impacts to governments and non-profits, including short- and long-run impacts.[79] Useful measures for evaluating impacts on these types of entities include assessments of the difficulty of paying regulatory costs and of continuing to provide services.

Examples of important impacts on government include water treatment costs paid by municipally-owned water authorities to comply with water quality standards. Air pollution controls required of power plants may affect municipally-owned electric companies. Implementation and enforcement costs associated with a variety of environmental regulations may impose costs on state or local government. If regulation affects the local tax base, then there may be impacts on government revenues or expenditures that may affect the provision of local public or private goods and services. For example, some coal-mining counties in the United States derive a significant portion of their budgets from coal-related revenues. Policies to restrict carbon pollution that reduce coal production could significantly affect such communities causing the loss of local public goods and lowered property values.[80]

To understand economic impacts on state, local and tribal governments, analysts should develop a baseline profile potentially including the following relevant factors:

- Size of the population in the community;
- Property values;
- Household income levels (e.g., median and/or income range);
- Age distribution;
- Unemployment rate;
- Foreclosure rate; and
- Revenue amounts by source.

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 may be important. If a government entity serves multiple communities, such as a regional water or sewer authority, then information for all the communities in the service area may be relevant.

To gain insight into the ability of governments to finance new regulatory costs, U.S. EPA's *Clean Water Act Financial Capability Assessment Guidance* (U.S. EPA 2024d) suggests examining baseline financial capability by exploring indicators of debt, socioeconomic conditions and success regarding financial management.[81] Analysts can obtain the community's bond or credit rating, which is itself determined by an assessment of financial health. For governments that rely on property taxes for

79 In some cases, EPA has been directed to consider impacts on government and non-profits. For example, UMRA requires assessment of impacts to state, local and tribal governments. The RFA as amended by SBREFA requires assessment of impacts to small entities including governments and non-profits (see Section 9.2 and Chapter 2).

80 Morris et al. (2019) study three counties with high labor shares engaged in coal mining and conclude that a third or more of their budgets may be funded with coal-related revenue.

81 The EPA uses U.S. EPA (2024d) to assess implementation of CWA requirements. The assessments affect negotiations for Clean Water Act compliance schedules.

income, analysts might consider the amount of debt that must be repaid through property taxes (known as net debt) per capita; or the net debt relative to the value of taxable properties. Property tax revenues relative to full market value of properties may be a useful indication of the property tax burden (U.S. EPA 2024d). Table 9.3 provides thresholds used by the Office of Enforcement and Compliance Assurance (OECA) and the Office of Water (OW) to indicate weak, mid-range or strong financial wellbeing of government entities.[82]

To screen for significant impacts on governments, analysts may wish to consider new regulatory costs per capita, the ratio of per capita costs to median household income and lowest quintile income, the latter especially in communities with households that have difficulty paying for their water services. Depending on these values, further analysis might be desirable.83 Further analysis should consider a government entity's options for funding new costs or how new process requirements could change operating procedures. For example, what is the availability of new loans or grants and user fees? Are there other viable routes for increasing funds available to finance new regulatory costs? Do new processes alter the quality or quantity of goods and services provided to residents? Other factors that are potentially relevant are the historic trend in government revenues; the capability of the revenue sources to shoulder additional financial burdens; and the magnitude of the benefits from the rule enjoyed by citizens.

Finally, indirect impacts on state, local and tribal government may be important if a policy changes local property values or employment rates or has other community-wide impacts. For example, brownfield grants to assess or clean up land may cause small increases in local property values which could raise property tax revenues (Sullivan 2017). On the other hand, a policy that exacerbates unemployment, for example, could cause more spending on assistance programs.

EPA regulations may also affect non-profit organizations. For example, non-profit hospitals face costs from hazardous waste disposal requirements. A baseline profile for non-profits should consider:

- Entity size and size of community served;
- Goods or services provided;
- Operating costs; and
- Amount and sources of revenue.

If the entity is raising its revenues through user fees or charging a price for its goods or services (such as university tuition), then the income levels of its clientele are relevant. If the entity relies on contributions, then it would be helpful to know the financial and demographic characteristics of its contributors and beneficiaries. If it relies on government funding (such as Medicaid) then possible future changes in these programs would be informative.

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82 For another source that explores approaches for assessing the health of a local government, see McDonald (2018).

83 For instance, when assessing regulatory costs, the EPA considers financial impact as low if costs per household are less than 1% of median household income, mid-range if it is 1-2% of median household income and high if it is greater than 2% (U.S. EPA 2024d). Also, see the discussion of financial and rate model analyses in Alternative 2 in U.S. EPA (2024d). Spreadsheet tools to help users evaluate the economic impacts of water quality decisions can be found at https://www.epa.gov/wqs-tech/economic-guidance-water-quality-standards#spreadsheet.

Table 9.3 – Indicators of Economic and Financial Well-Being of Government Entities

Indicator	Strong	Mid-Range	Weak
Bond Rating	AAA – A (S&P) or Aaa – A (Moody's) or AAA – A (Fitch Ratings)	BBB (S&P) or BAA (Moody's) or BBB (Fitch Ratings)	BB - D (S&P) or Ba – C (Moody's) or BB - D (Fitch Ratings)
Overall Net Debt as a Percent of Full Market Property Value	Below 2%	2% - 5%	Above 5%
Unemployment Rate	More than 1 Percentage Point Below the National Average	± 1 Percentage Point of National Average	More than 1 Percentage Point Above the National Average
Median Household Income	More than 25% Above Adjusted National MHI	± 25% of Adjusted National MHI	More than 25% Below Adjusted National MHI
Property Tax Revenues as a Percent of Full Market Property Value	Below 2%	2% - 4%	Above 4%
Property Tax Collection Rate	Above 98%	94% - 98%	Below 94%

Source: Table B-1 of U.S. EPA 2024d.

To screen for impacts on non-profits, analysts can compare regulatory costs to baseline revenues or operating expenses. Regulatory costs can also be compared to baseline asset values or, after accounting for debts, net asset values. If these ratios are large, insights would be gained from information on the relative importance, size and growth rate of the non-profit, the nature of the population being served and the vulnerability of revenues and donors.

Impacts on Small Governments and Small Non-Profits

Consideration of impacts on small governments and small non-profits is required by the RFA as amended by SBREFA.[84] 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. As with 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 activities 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 2006a). Under the RFA, economic impacts on small governments are included in the screening analysis for significant economic impacts on a substantial number of small entities (SISNOSE), and any required regulatory flexibility analysis. In order to determine

84 See Chapter 2 and Section 9.2 for more information.

SISNOSE for small governments, the EPA conducts a screening analysis for both proposed and final rules based on annualized compliance costs as a percentage of revenue (U.S. EPA 2006a).

The Unfunded Mandates Reform Act (UMRA) uses the same definition of small government as the RFA, with the addition of tribal governments. Section 203 of UMRA requires the Agency to develop a "Small Government Agency Plan" for any regulatory requirement that might "significantly" or "uniquely" affect small governments. In general, "impacts that may significantly affect small governments include — but are not limited to — those that may result in the expenditure by them of $100 million [adjusted annually for inflation] or more in any one year." Other indicators that small governments are uniquely affected may include whether they would incur higher per-capita costs due to economies of scale, a need to hire professional staff or consultants for implementation, or requirements to purchase and operate expensive or sophisticated equipment.[85]

The RFA requires separate consideration of regulatory impacts on small non-profits and defines one as a non-profit "enterprise which is independently owned and operated and is not dominant in its field." 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, direct economic impacts on small non-profit organizations are included in the SISNOSE screening analysis, and if required, the regulatory flexibility analysis for a rule. In order to determine SISNOSE for small non-profits, the EPA conducts a screening analysis for both proposed and final rules based on annualized compliance costs as a percentage of operating expenditures.[86]

9.5.5 Economy-Wide Impacts

The more interconnected a regulated sector is with the rest of the economy, the greater the likelihood that a regulation will affect related markets. If a regulation is expected to affect markets with (i) significant cross-price effects between markets, *and* (ii) significant pre-existing distortions, it may be appropriate to examine economy-wide impacts in a supplemental analysis (U.S. EPA 2017). Pre-existing market distortions that could be exacerbated by environmental regulations include taxes or subsidies on labor, energy or capital; monopoly or monopsony power; price controls; or other government regulations that change the way markets operate.

Computable general equilibrium (CGE) models are particularly effective at assessing long-run economy-wide impacts.[87] These include the allocation of employment or other factors of production across sectors, the distribution of output by sector and the distribution of income among households. For example, regulations in the power sector may cause electricity prices to increase. The price increase will affect all industries that use electricity as an input to production, as

85 Guidance on complying with Section 203 of UMRA, "Interim Small Government Agency Plan," is available on the EPA's intranet site, ADP Library.

86 See Table 1, "Recommended Quantitative Metrics for Economic Impact Screening Analyses" of U.S. EPA 2006a.

87 CGE models assume that for some discrete time period an economy can be characterized by a set of conditions in which supply equals demand in all markets. When the imposition of a regulation alters conditions in one market, the model determines a new set of relative prices that return the economy to its long-run equilibrium. While highly aggregate in nature, CGE models capture substitution possibilities between production, consumption and trade; interactions between economic sectors; and interactions with pre-existing distortions. Thus, they provide information on changes outside the directly regulated sector. See Chapter 8 for more discussion.

well as households. A CGE model can assess the distribution of consequent changes in production and consumption. By design, the basic capacity to describe and evaluate these sorts of impacts exists to some extent within every CGE model. More detailed impacts (e.g., effects on a certain type of facility or on an environmental endpoint such as drinking water) are difficult to capture in a CGE model due to model dimensionality and/or data constraints.

The simplest CGE models typically include a single representative consumer, a set of relevant production sectors, and a government sector within a single-country, static framework. Additional complexities can be specified. A CGE model can be solved dynamically over a longer time horizon, incorporating intertemporal decision-making on the part of consumers or producers. These decisions have implications for the treatment of savings, investment and the long-term profile of consumption and capital accumulation. Consumers can be divided into income quintiles or deciles, and producers disaggregated into a variety of regions and sectors, each producing a set of unique commodities. The government, in addition to implementing a variety of taxes and other policy instruments, may provide a public good or run a deficit. CGE models can be international in scope, consisting of many countries or regions linked by international flows of goods and capital. The behavioral equations that characterize economic decisions may take on simple or intricate functional forms.

While CGE modeling is complex, the effort may be worthwhile when impacts are likely to be substantial and widespread and when appropriate data (e.g., input-output tables, elasticities) are available. Text Box 5.3 and Chapter 8 discuss detailed criteria for judging model quality. Feedback from the Science Advisory Board (SAB) identified several guiding principles as to when economy-wide modeling is appropriate for assessing economic impacts of regulation (U.S. EPA 2017). Aspects of a CGE model that could affect suitability include degree of temporal, sectoral and geographic disaggregation; time horizon; the way in which firm and household expectations about the future are modeled; the types of impacts that can be forecast; and the approach for representing the policy instrument. CGE models may be useful as a supplement to other analytic approaches to evaluate sectoral effects (including shifts in labor or capital between sectors), impacts on energy supply and energy prices and effects on consumers. In some instances, linking a CGE model to sector models may be a useful way to leverage the relative advantages of both approaches in a single comprehensive framework (U.S. EPA 2017).[88]

CGE models have limitations. Many are not designed to illuminate certain types of impacts, such as short-run or transitional impacts. For example, a standard forward-looking CGE model that assumes full employment and instantaneous market adjustments is ill-suited to evaluate overall employment impacts or the potential for short-run disequilibria in labor and capital markets. Analysts interested in evaluating the short-run impacts of a policy should select a different framework for analysis. Finally, relatively few CGE models incorporate feedback from changes in pollution; instead, they mainly focus on private markets.

A partial equilibrium model of multiple markets that considers the interactions between a regulated market and other closely related markets may be a practical alternative to a CGE model. Such models require estimates of demand and supply elasticities and cross-price elasticities for included markets. Partial equilibrium models may be appropriate for regionally-based or sector-specific regulations that are too narrowly defined to be adequately captured in more aggregate CGE models.

88 See Text Box 8.1 for more discussion of model linking.

The SAB recommends that analysts apply the simplest model that is adequate to address the policy question at hand and consider a suite of models when possible (U.S. EPA 2017). A balance should be struck between capturing detail and complexity in the model versus transparency and tractability of the analysis.

As with all economic models, economy-wide and partial equilibrium models are simplified representations of complex economic systems built to assess relationships between economic factors. They are useful for estimating effects on groups but are not reliable predictors of firm or facility-level decisions. See Section 9.5.1.2 for further explanation of the common simplifying assumptions about firm decision-making.

9.5.6 Impacts of Benefits

Environmental benefits are generally nonmarket effects and as such pose special analytic challenges. As with costs, the benefits from improved environmental quality or health can accrue to, and may differ among, a wide variety of individuals. A key determinant of differential impacts is whether environmental improvements differ among affected groups (due to different exposure pathways, for example), or are uniform but have variable impacts due to differences in pre-existing factors such as baseline exposures or health (for more discussion, see U.S. EPA 2024c, especially Chapter 4).

The literature provides several potential frameworks for explicitly considering variability in the impacts of benefits across groups. Typically, these frameworks start with defining environmental damages as a function of exposure and individual susceptibility to environmental stressors, then they identify sources of susceptibility and finally they assess the impacts from environmental regulation (see, for instance, Hsiang et al. 2019; Gee and Payne-Sturges 2004; and Morello-Frosch and Jesdale 2006).

Useful information to improve understanding of the distribution of regulatory benefits includes:

- The types of health effects or other benefits;
- Population groups to whom the benefits are expected to accrue;
- How exposure varies across the affected groups; and
- How beneficial outcomes vary across population groups.

In addition to accruing to those who directly experience a reduced health risk, health and environmental quality benefits may also accrue to people who own homes near improved environmental quality, or to employers whose workers enjoy improved health and increased labor productivity, as well as to others.

Chapter 10 discusses how to analyze health effects and benefits for specific populations of concern (i.e., by income, race/ethnicity and age). The data and methods discussed there may be relevant for analyzing the distribution of benefits on other categories of people, on communities or on the general population. Sometimes analysts may wish to account not only for the ways in which changes in the regulated sector affect the distribution of benefits, but also how price and quantity responses across the economy affect the distribution of benefits, or how changes in environmental quality affect prices and quantities. Absent a partial-equilibrium or economy-wide model that explicitly incorporates benefits, relatively rare in the literature, these indirect impacts are difficult to evaluate.

Chapter 9 References

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Chapter 10 - Environmental Justice and Life Stage Considerations

Instead of focusing on quantifying and monetizing total benefits and costs, an evaluation of the impacts of a regulation examines how a regulation allocates benefits, costs, transfers and other outcomes across specific groups of interest. Chapter 9 describes approaches to quantify economic impacts across a wide array of groups that may be of interest to decisionmakers. This chapter overlaps with Chapter 9 in some respects — many of the economic impact categories it discusses are also potentially relevant here — but it is distinct in several ways. First, this chapter specifically considers the possible impacts of a regulatory action on people of color, low-income or Indigenous populations (i.e., the focus of environmental justice) and on children and older adults (i.e., life stages) due to their increased vulnerability to health effects from pollution. Second, while variation in the benefits and costs of regulation across these population groups is a significant consideration, this chapter also discusses the importance of characterizing changes in human health endpoints and environmental risk.

10.1 Executive Orders, Directives and Policies

Consideration of how economic and human health effects vary across specific population groups and life stages arises from several executive orders (EOs), directives and other documents.[1,2] The Agency also has developed separate guidance to provide direction to analysts on conducting environmental justice analyses. Together, these orders, directives and policies provide a solid foundation for considering effects on population groups from an environmental justice (EJ) and life stage standpoint in the rulemaking process.

In addition to the general guidance in the Office of Management and Budget's (OMB's) Circular A-4 (OMB 2023) regarding distributional analysis, several EOs, described more fully in Chapter 2, directly address different types of effects for population groups of concern: [3]

1 EPA's Regulatory Management Division's Action Development Process Library (http://intranet.epa.gov/adplibrary/adp) is a resource for accessing relevant statutes, executive orders and EPA policy and guidance documents in their entirety.

2 Some environmental statutes also identify population groups that may merit additional consideration. See EPA Legal Tools to Advance Environmental Justice (U.S. EPA 2022) for a review of legal authorities under the environmental and administrative statutes administered by the EPA.

3 This chapter addresses analytical components of EOs 12898 and 14096 and does not cover other components such as ensuring proper outreach and meaningful involvement.

- EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" (1994), calls on each federal agency to make achieving EJ part of its mission to the greatest extent practicable and permitted by law "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 in the United States."
- EO 14096, "Revitalizing Our Nation's Commitment to Environmental Justice for All" (2023), supplements EO 12898 and calls on agencies to, as appropriate and consistent with applicable law, identify, analyze, and address:
    - Disproportionate and adverse human health and environmental effects…, including those related to climate change and cumulative impacts of environmental and other Burdens on communities with environmental justice concerns;
    - Historical inequities, systemic barriers, or actions related to any Federal regulation, policy, or practice that impair the ability of communities with environmental justice concerns to achieve or maintain a healthy and sustainable environment; and
    - Barriers related to Federal activities that impair the ability of communities with environmental justice concerns to receive equitable access to human health or environmental benefits, including benefits related to natural disaster recovery and climate mitigation, adaptation, and resilience.
- EO 13045, "Protection of Children from Environmental Health Risks and Safety Risks" (1997), states that each federal agency shall ensure that its policies, programs, activities, and standards address disproportionate risks to children that result from environmental health or safety risks.
- EO 13175, "Consultation and Coordination with Indian Tribal Governments" (2000), calls on federal agencies to have "an accountable process to ensure meaningful and timely input by tribal officials in the development of regulatory policies that have tribal implications."
- EO 12866, "Regulatory Planning and Review" (1993), explicitly allows for consideration of "distributive impacts" and "equity" when choosing among alternative regulatory approaches, unless prohibited by statute.[4]
- EO 14094, "Modernizing Regulatory Review" (2023), supplements, reaffirms, and amends 12866, and confirms that "[r]egulatory analysis, as practicable and appropriate, shall recognize distributive impacts and equity, to the extent permitted by law."

10.2 Environmental Justice

This section offers a high-level summary of analytic expectations and recommendations for evaluating environmental justice concerns for EPA regulatory actions, consistent with *Technical Guidance for Assessing Environmental Justice in Regulatory Analysis* (*EJ Technical Guidance*) (U.S. EPA 2024a). Analysts should consult the *EJ Technical Guidance* for additional detail on analytic approaches and considerations.

4 EO 13563, issued in January 2011, supplements and reaffirms the provisions of EO 12866.

An analysis of EJ concerns for regulatory actions should address three questions:[5]

- **Baseline:** Are there existing EJ concerns associated with environmental stressors affected by the regulatory action for population groups of concern?[6]
- **Regulatory options:** For the regulatory option(s) under consideration, are there potential EJ concerns associated with environmental stressors that are affected by the regulatory action for population groups of concern?
- **Mitigation or exacerbation of effects:** For the regulatory option(s) under consideration, are EJ concerns exacerbated, mitigated, or unchanged compared to the baseline?

These questions provide the framework for analyzing the effects of a regulatory action on population groups of concern. The extent to which an analysis can address all three questions will vary due to data limitations, time and resource constraints, and other technical challenges. These challenges will vary by media and regulatory context, including the availability of information generated from human health risk and exposure assessments, or other components of the regulatory analysis.

The EPA encourages analysts to document key reasons why a particular question cannot be addressed to help identify future priorities for filling key data and research gaps. A lack of existing data and methods does not mean there is no EJ concern, so identifying such gaps is important. Regardless of the approach taken, the highest quality and most relevant data should be applied in a manner consistent with the OMB and EPA data quality guidelines (OMB 2019; U.S. EPA 2012; U.S. EPA 2002) and the *Peer Review Handbook* (U.S. EPA 2015).

The term "disproportionate" is used here to refer to differences in effects or risks that are extensive enough that they may merit Agency action and should include consideration of cumulative impacts or risks where appropriate and consistent with applicable law (U.S. EPA 2022). In general, the determination of whether a difference in effects or risks is disproportionate is ultimately a policy judgment which, while informed by analysis, is the responsibility of the decision-maker.[7] The terms "difference" or "differential" indicate an analytically discernible (or measurable) distinction in effects or risks across population groups. It is the role of analysts to assess and present differences in anticipated effects across population groups in the baseline and for the regulatory options, using the best available information (both quantitative and qualitative) to inform the decision-maker and the public.

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5 An EJ concern is the actual or potential lack of just treatment or meaningful involvement of any population group, community, or geographic area (e.g., associated with differences in income, race, color, national origin, Tribal affiliation, or disability status) in the development, implementation, and enforcement of environmental laws, regulations, and policies. For analytic purposes, this concept refers specifically to disproportionate and adverse health and environmental effects that may exist prior to or be created by the regulatory action.

6 The term environmental stressor encompasses the range of chemical, physical, or biological agents, contaminants, or pollutants that may be subject to a regulatory action.

7 A finding of disproportionate and adverse effects is neither necessary nor sufficient for the EPA to address them. The Agency's statutory and regulatory authorities provide a broad basis for protecting human health and the environment and do not require a demonstration of disproportionate effects to protect the health or environment of any population.

10.2.1 Population Groups of Concern

At an early stage of the analysis, analysts need to identify the population groups of concern relevant to a specific regulatory context.[8] The concept of vulnerability can be used to help identify population groups of concern.[9] For example, analysts can combine available data on baseline health, demographic, socioeconomic, or other relevant indicators (including those related to cumulative impacts, historic inequities and systemic barriers, and lack of access) to identify characteristics in affected communities that correlate with increased vulnerability to environmental exposure or lack of opportunity for public participation (Fann et al. 2011).

While the EPA does not have rigid criteria for identifying population groups of concern, E.O.s 12898 and 14096 reference race, ethnicity, national origin, low-income, disability status, Tribal affiliated and Indigenous populations, and those engaged in cultural or subsistence practices. Note that population groups of concern may be clustered within specific communities or geographically dispersed (e.g., unhoused populations, migrant workers). Underserved communities or populations also may warrant consideration.[10] See the *EJ Technical Guidance* (U.S. EPA 2024a) for more in-depth discussion.

It may be useful in some contexts to analyze population groups in combination or to evaluate additional aspects of diversity within a specific population group of concern (e.g., by life stage, gender), particularly when some individuals within a population group may be at greater risk for experiencing disproportionate and adverse effects (e.g., due to unique exposure pathways). Analysts should rely on the Office of Management and Budget (OMB) or other federal statistical agencies (e.g., U.S. Census Bureau), when available, to define relevant population groups (or combinations thereof) for a specific regulatory action. Note that analysis of additional population groups is not a substitute for examining the population groups explicitly mentioned in the E.O.s.

10.2.2 Main Steps of an EJ Analysis

Conducting a preliminary analysis may be a useful first step to identifying what level of assessment is feasible and appropriate to support the regulatory action. In addition, it can help identify the extent to which a regulatory action may raise EJ concerns that need further evaluation. Feasibility is informed by a technical evaluation of available data and methods, including:

- Scientific literature that discusses the effects of the stressor(s) being regulated on population groups of concern;
- Information received via public comments, technical reports, press releases, or other documentation discussing the environmental and health effects of the stressor(s) being regulated for population groups of concern, including information on other relevant environmental or non-environmental stressors;

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8 The term population groups of concern is used instead of the term subpopulations to include "population groups that form a relatively fixed portion of the population (e.g., based on ethnicity)." See the EPA's Early Life Stages website: http://www.epa.gov/children/early-life-stages.

9 Note that specific terminology and definitions related to vulnerability may be provided by statute.

10 Examples of other characteristics that may be relevant in some regulatory contexts include linguistic isolation, occupation, rurality, and employment status, among others.

- Availability of spatially disaggregated data for population groups that may live, work, or play in close proximity to the stressor(s) being regulated, or may otherwise be affected by the stressor(s); or
- Availability of methods for conducting in-depth analysis (e.g., proximity-based approach, risk- or exposure-assessment, and mixed methods approach).

If the preliminary analysis reveals that the scientific literature and data are unavailable or of insufficient quality to pursue an in-depth analysis that characterizes how exposure, risk, or health effects are distributed across population groups, analysts are expected to explain why additional analysis is not possible. In particular, analysts are encouraged to discuss relevant evidence, key limitations, and sources of uncertainty highlighted in the published literature. Some impacts that cannot be quantified may still represent important effects that should be considered in the analysis.

When conducting further evaluation is determined to be both feasible and appropriate, an EJ analysis typically includes five main steps (Figure 10.1). We briefly describe each step below.

Figure 10.1 - Main Steps of an EJ Analysis

Identify Regulated Sources	Describe Environmental Stressors	Characterize Affected Populations	Compare Affected and Compariosn Populations	Conduct Sensitivity Analysis
• Where are regulated sources located? • Do health and environmental risks vary with source characteristics? • Do the regulatory options vary with source characteristics?	• Is the pollutant spatially distributed? • What is known about fate and transport? • What are the relevant health outcomes?	• What factors might drive higher exposure? • What factors might drive differential exposure? • How do health outcomes vary by population group or community?	• Given data and methods, are you only characterizing the baseline or also regulatory options? • How are you presenting and characterizing results?	• What are key uncertainties and limitations? • Are there specific pockets of concern? • What are they key assumptions?

Questions to Ask at Each Step

- How can meaningful engagement inform the EJ analysis?
- What data are available and at what spatial scale?
- What tools are available to model exposure and/or risk?
- How are the effects distributed across sources and communities?

(1) Identify the sources being regulated: Before analysts can identify the populations and communities being affected by a regulatory action, it is important to first characterize the regulated sources: where are they located? Are there particular characteristics of the regulated sources that contribute to higher exposure and/or risk of health effects? Do the regulatory options vary with these characteristics? For instance, are some sources subject to greater stringency or other regulatory requirements that would be important to account for in the EJ analysis?

(2) Describe the environmental stressor: The spatial distribution of health and welfare outcomes is a relevant consideration for some regulatory actions. In these cases, evidence on the fate and transport of the environmental stressor can help determine the populations and communities potentially exposed. In other cases, the regulatory action's effects may be more widespread. It is also important to understand which specific health effects are of greatest relevance for a given regulatory context. The benefits analysis and, when conducted, the human health risk and exposure assessments can be important sources for this information.

(3) Characterize affected populations: It is important to understand what factors may contribute to EJ concerns. How are individuals being exposed? Are there unique pathways or other factors that drive higher exposures for some population groups? Recognizing underlying contributors within a specific regulatory context is important for properly assessing EJ concerns and can aid in the design of regulatory options. This may include evidence of already overburdened communities, including the cumulative effects of exposure to multiple environmental or non-environmental stressors on human health and well-being.

(4) Compare the affected and comparison groups: To answer each of the three analytic questions, analysts need to characterize the exposure and risk of health effects for population groups of concern in the baseline and for the regulated action relative to a comparison population group (see Section 10.2.6). This allows analysts to gauge the extent to which effects for the affected population are similar or different than they are for the comparison group and how they vary across population groups.

(5) Conduct sensitivity analysis: Due to the inherent limitations and uncertainties associated with analyses of EJ concerns, conducting sensitivity analysis around key assumptions is particularly important for clearly communicating results to the public.

Figure 10.1 also identifies four overarching questions that are relevant throughout the EJ analytic process:

1. **How can meaningful engagement inform the EJ analysis?**

   Meaningful engagement can help analysts to identify and sometimes help fill information and data needs. It can also help analysts to identify factors such as unique pathways or pre-existing vulnerabilities that may contribute to exposure and/or risk for affected populations. See U.S. EPA (2024a) and U.S. EPA (2024b) for more information.

2. **What data are available and at what spatial scale?**

   The quality and availability of data are key determinants in the scope and complexity of the EJ analysis. In some cases, analysts will have data at the individual level for the environmental stressor being regulated, allowing for a detailed, rigorous analysis. In other cases, analysts may need to rely on proxies for individual-level effects. Data relevant to the EJ analysis may include, but are not limited to, the demographic and socioeconomic characteristics of populations that may be exposed to environmental stressors from regulated sources, what each regulated source is emitting or discharging, and pre-existing health conditions or other environmental and non-environmental stressors that increase the vulnerability and therefore the risk of experiencing a health effect for some population groups.

3. **What tools are available to model exposure and/or risk?**

   Analysts have a choice among several scientifically defensible methods to assess EJ concerns associated with a regulatory action, including proximity-based analysis, exposure and risk modeling, and combining qualitative and quantitative approaches. The choice of a specific analytic method for the EJ analysis is often driven by data availability. Together, the data and methods utilized directly influence what conclusions can be drawn regarding EJ concerns for specific population groups or communities. Chapter 6 of the *EJ Technical*

Guidance (U.S. EPA 2024a) discusses the main methods available and the potential advantages and disadvantages of each in more detail.[11]

4. **How are the effects distributed across sources and communities?**

   In some cases, extensive differences in effects among population groups of concern may occur in only a few geographic locations. Referred to as hot spots, these locations are typically exposed to localized concentrations of emissions from one or more sources along with other stressors. In these cases, it may be appropriate to tailor the analysis to evaluate effects in a few specific areas. Identifying the potential for hot spots early helps analysts develop appropriate sources of data and analytic approaches, which may differ from those used for a broader analysis (see Section 10.2.7.5).

10.2.3 Recommendations for Analyses of EJ Concerns

The *EJ Technical Guidance* (U.S. EPA 2024a) makes five overarching recommendations to ensure a high-quality EJ analysis, while also recognizing the need for flexibility to reflect policy considerations and technical challenges within a particular regulatory context. The recommendations are intended to bring greater consistency across EJ analyses as they strive to answer the three analytic questions but are not prescriptive and do not mandate the use of a specific approach. Analysts should use their best professional judgement to decide on the type of analysis that is feasible and appropriate within a specific regulatory context.

While these recommendations and best practices are intended as a starting point, they should not be interpreted as limiting the scope of the EJ analysis. It is recommended that analysts thoughtfully tailor their analysis to the rule context and incorporate new data and methods as they become available. Ultimately, the EPA strives to innovate and improve upon EJ analyses as the state of science continues to evolve. The five overarching recommendations are:

1. **When risks, exposures, outcomes, or benefits of the regulatory action are quantified, some level of quantitative EJ analysis is recommended.**
   a. Analysts should present information on estimated health and environmental risks, exposures, outcomes, benefits, or other relevant effects disaggregated by race, ethnicity, income, and other relevant demographic and socioeconomic categories when feasible and appropriate.
   b. When such data are not available, it may still be possible to evaluate potential risk or exposure using other metrics (e.g., proximity to affected facilities, cancer or asthma prevalence, or evidence of unique exposure pathways for specific population groups) in a scientifically defensible way.
   c. When health and environmental outcomes or benefits are not quantified or disaggregated by race, ethnicity, income, or other relevant demographic and socioeconomic categories, analysts should present available quantitative and/or qualitative information that sheds light on EJ concerns that may arise.

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11 For an overview of proximity-based analysis, including a discussion of various spatial analysis techniques used in the literature, see also Chakraborty et al. (2011), Chakraborty and Maantay (2011), and Mohai and Saha (2007).

2. **Analysts should integrate EJ into the planning of a risk assessment conducted for the regulatory action.**[12]
3. **Analysts should strive to characterize the distribution of risks, exposures, or outcomes within each population group, not just average effects.**
    a. In particular, analysts should pay attention to whether populations in the upper tail of the distribution face the highest adverse risks, exposures or health effects.
4. **Analysts should follow best practices appropriate to the analytic questions at hand (see Text Box 10.1).**
5. **As relevant, analysts should consider any economic costs or challenges that may be exacerbated by the regulatory action for relevant population groups of concern.**
    a. For instance, it may be appropriate to consider how low-income populations are affected by price changes or to consider the distribution of economic costs (i.e., private and social costs) more broadly from an EJ perspective.

10.2.4 Characterizing the Baseline and Regulatory Options for In-Depth EJ Analysis

The five main steps of an in-depth EJ analysis can be applied to characterize baseline conditions, evaluate the effects of the regulatory options, and make comparisons between the two to evaluate the three analytic questions, when data and methods allow.

The OMB (2023) defines the baseline as "an analytically reasonable forecast of the way the world would look absent the regulatory action being assessed, including any expected changes to current conditions over time." It includes the characteristics of current populations and how they are affected by pollutant(s) prior to the regulatory action under consideration. As the OMB definition implies, however, the baseline is not a static concept. In particular, the OMB notes that analysts may need to consider the evolution of the market, compliance with other regulations, and the future effect of current government programs and policies, as well as other relevant external factors to project future baseline conditions. As discussed in chapter 5, how future regulations or policies affect the baseline specification is complex and requires consideration of many factors. Ideally all potential influences on baseline conditions would be estimated, but it is generally not practicable to do so. Anticipated changes in baseline demographic composition may also be relevant in an EJ context. Per the recommendations in Section 10.2.2, the baseline for the EJ analysis, including the geographic scope, year of analysis, and health and other effects, should be consistent with how it is specified in other parts of the regulatory analysis.

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12 For more information on this recommendation, see Chapter 5 of U.S. EPA (2024a).

Text Box 10.1 - Current Best Practices for Evaluating EJ Concerns

- Use the best available science while relying on current, generally accepted Agency procedures for conducting risk assessment and economic analysis.
- Use existing frameworks and data from other parts of the regulatory analysis, supplemented as appropriate.
- Be consistent with the basic assumptions underlying other parts of the regulatory analysis, such as using the same baseline and regulatory option scenarios.
- Use the highest quality and most relevant data available. Discuss the overall quality and main limitations of the data.
- Identify relevant population groups of concern and discuss available evidence of factors that make them vulnerable to adverse effects (e.g., unique pathways; cumulative exposure to multiple stressors; behavioral, biological, or environmental factors).
- Consider unique pathways for individuals that rely on cultural or subsistence practices and relevance for Tribal or Indigenous populations, when practicable.
- Carefully select and justify the choice of comparison population group.
- Carefully select and justify the choice of the geographic unit of analysis and discuss any challenges or aggregation issues related to the choice of spatial scale.
- Analyze and compare effects in baseline and across policy scenarios to show differences in effects.
- Present summary metrics for each population group and the comparison population group and characterize differences between them.
- When data allow, characterize the distribution of risks, exposures, or outcomes within each population group, not just average effects.
- Disaggregate data to reveal important spatial differences (e.g., demographic information for each source/place) when feasible and appropriate.
- Clearly describe data sources, assumptions, analytic approaches, and results.
- Summarize the main conclusions of differences in exposure or health risk between analyzed population groups based on the available evidence.
- Discuss key sources of uncertainty or potential data biases (e.g., sample size, proximity as a surrogate for exposure) and how they may influence the results.
- When possible, conduct sensitivity analysis for key assumptions or parameters that may affect findings.
- Qualitatively describe behavioral responses not accounted for in the analysis that could affect the level or distribution of exposure or health risks (e.g., dynamic spatial or temporal effects, averting or adaptive behavior).
- Make elements of the EJ analysis as straightforward and easy for the public to understand as possible.

When data and methods allow, an EJ analysis can also examine the distribution of effects for each regulatory option – different configurations of the regulatory action being considered. This analysis is based on a prediction of the state of the world under the regulatory options. For the analysis of EJ concerns, analysts are encouraged to examine how the exposure, risk of health or environmental effects, or other outcomes of the regulatory action are distributed across population groups for the regulatory options being considered, where practicable. The EJ analysis can then evaluate the change in the exposure or risk of relevant environmental and health effects for each regulatory option compared to the baseline. In addition to identifying whether the regulatory action is expected to exacerbate, mitigate, or leave baseline EJ concerns unchanged, the analysis should shed light on the extent and distribution of these changes.

With these three sets of information – effects in the baseline, effects under the regulatory options, and a comparison of the two –analysts can characterize the distribution of environmental and health effects associated with a regulatory action, thus answering all three EJ analytic questions.

Note that a constant reduction in risk or exposure across population groups will likely not mitigate EJ concerns if there are differences in baseline environmental quality or health risk across population groups or communities (Maguire and Sheriff 2011). Conceptually, an EJ concern is only completely mitigated when there is no difference in the distribution of effects across population groups for the regulatory options being considered – i.e., everyone is experiencing the same environmental quality or health risk post-regulation.

10.2.5 Data and Information to Assess EJ Concerns

In general, the type of analysis that can be conducted depends on the availability and quality of data. In some cases, spatially resolved, individual-level data may be most appropriate and relevant for an analysis of EJ concerns. In other cases, distance from a regulated source may be the best available metric. At times, the best available information may be qualitative, including local knowledge from affected communities and Tribes (e.g., Indigenous Knowledge, also referred to as Traditional Ecological Knowledge). In all cases, analysts should use the highest quality and most relevant data and information.

When data are missing or incomplete, it is recommended that analysts document what specific types of data are unavailable or of insufficient quality, including but not limited to cases where the data are available but not of the desired granularity (spatially or temporally) and/or available for only subsets of the population. Text Box 10.2 illustrates how data quality may affect the level of analysis.

Recognizing the importance of data quality, data needed to conduct an EJ analysis may include:

- Demographic and socioeconomic characteristics (e.g., race, ethnicity, income);
- Location of pollution sources (e.g., latitude/longitude coordinates, zip code, county);
- Historical, current, and projected emissions or concentrations of stressor(s) relevant to the regulatory action;
- Prevalence of specific exposure pathways that may increase risk for some population groups;
- Health effects (e.g., hospital and emergency admissions, race and ethnicity-stratified mortality rates, race and ethnicity-stratified asthma or other morbidity rates);
- Other environmental or non-environmental stressors that may be risk- or effect modifiers (e.g., indoor air concentrations, vulnerability to effects of climate change);
- Risk coefficients stratified by population groups of concern (e.g., race, ethnicity, income); and
- Distribution of economic costs, when relevant (see Section 10.x).

Text Box 10.2 - Data Quality and Spatial Resolution in the Context of Air Quality Regulations

Analysts' ability to address how a regulatory action changes the distribution of risk across population groups depends on the quality and spatial resolution of the data available. Finer-scale air quality, health, and demographic data allow one to assess the distribution of effects across population groups and to have greater confidence in the conclusions drawn from these data. When air quality data are lacking or only available at a coarse level, the ability to assess change in risk across populations and other conclusions is more limited.

An example in limited data environments: Using race-stratified county-level mortality and morbidity data, analysts can calculate population-weighted mortality rates by county. Analysts can then use a highly aggregated baseline air quality modeling projection (e.g., 12 or 36 km) to identify population groups most exposed to air pollution. Using geospatial tools, it is possible to combine the two sources of data. The coarse geographic scale of air quality information may inhibit the analyst's ability to detect meaningful differences in effects among and between groups. When risk coefficients are unavailable, it is not possible to estimate health effects separately for each population group.

An example in data-rich environments: Using finely resolved air quality data, analysts can identify at a highly disaggregated level (e.g., 1 km) population groups that experience the highest exposure to air pollution. Analysts can also identify population groups that exhibit the highest baseline incidence or prevalence rates for air pollution health effects. Using geospatial tools, analysts can spatially combine the two data sources. Using race-specific or standard risk coefficients analysts can then estimate health effects for each population group.

Three types of information are frequently used as inputs into an EJ analysis: demographic and socioeconomic data, emission data for regulated sources, and data on pre-existing health conditions or other factors that may increase an individual's vulnerability when exposed to releases from regulated sources. The U.S. Census Bureau is the recommended source for demographic and socioeconomic data in an EJ analysis. It produces several national-level data products that report demographic and socioeconomic characteristics at relatively fine spatial scales (e.g., census tract, block group), including the decennial Census, the American Community Survey (ACS), and the American Housing Survey (AHS). See Section 6.3 of the EJ Technical Guidance (U.S. EPA 2024a) for a detailed discussion of these and other sources of data and information to assess EJ concerns.

10.2.6 Analytic Methods

A variety of scientifically defensible methods can be used to assess EJ concerns associated with regulatory actions. The choice of analytic method is most often driven by data availability. Analysts may also rely on a combination of methods when analyzing a regulatory action. The conclusions that can be drawn from the analysis will vary depending on the method used. See the *EJ Technical Guidance* (U.S. EPA, 2024a) for a discussion of specific analytic methods and their relative advantages and limitations when evaluating EJ concerns.

Considerable uncertainty may exist about key relationships and health outcomes, such as how a reduction in emissions or other types of releases from a given source translates into ambient environmental quality and how it, in turn, translates into the human health effects of interest. This is particularly problematic if uncertainties differ across population groups. For instance, if an overexposed population group is more responsive to exposure (i.e., individuals in the group experience greater adverse health effects per unit of exposure), then using exposure alone as a proxy will underestimate the health risk posed by a stressor to that group. On the other hand, if proximity to a pollutant source does not correlate with exposure, it could overstate potential differences in health risk. Analysts should select the method that is most appropriate for the available data, recognizing time and resource constraints.

Regardless of the analytic approach used, the EJ analysis should be presented in a transparent way and include the following:

- Information about the specific population groups and individuals affected by the regulatory action;
- Main exposure pathways and expected health and environmental outcomes;
- Evidence for why risk, exposure, or outcomes may vary by population group, including the role of other relevant environmental and non-environmental stressors;
- Relevant geographic scale;
- Descriptions of the main methods of analysis used;
- Descriptions of key data or modeling assumptions;
- Summary statistics for the baseline and each regulatory option (both the mean and distribution) by population group;
- An easy-to-understand description of what the summary statistics show;
- Conclusions based on the information available;
- Sensitivity analysis to examine the robustness of results across options presented; and
- Data quality, key sources of uncertainty, and limitations that affect conclusions regarding potential differential effects.

Analysts should follow best practices appropriate to the questions under consideration (see Text Box 10.1). If it is not feasible to follow a particular best practice, analysts should explain why this is the case.

10.2.7 Analytic Considerations

Regardless of the analytic approach taken, analysts make a number of key decisions that can have a substantial effect on the results of the analysis, including: the geographic and temporal scope of the analysis; how to specify the comparison population group; how to spatially identify and aggregate effects across affected and unaffected populations; whether to conduct analysis from a community and/or facility perspective; and how to evaluate underlying variability, including the potential for hotspots.

An important general strategy in analyzing EJ concerns is the use of sensitivity analysis. Due to the uncertainties associated with the analytic decisions discussed below, sensitivity analysis around key assumptions is often critical for clearly communicating results to the public.

10.2.7.1 Geographic and Temporal Scope

The geographic scope of analysis for an EPA regulatory action is often the entire United States since requirements typically apply nationwide. However, in some cases the effects of a regulatory action are expected to be concentrated in specific regions or states. In such cases, it may make sense to analyze and present differences in health and environmental outcomes across population groups at both a national and a sub-national level. Because the geographic scope can affect the results of the analysis (Baden et al., 2007), analysts should make certain that the scope is relevant for the regulatory action under consideration. In addition, it is important to keep in mind that differences in health and environmental outcomes in one region or state may not necessarily hold in other regions or states.

It may be important to evaluate regulatory action effects on both shorter and longer time horizons. For instance, while a regulatory action may result in near-term reductions in emissions, changes in health and other risks may occur on a longer timeframe. In some cases, effects may even be felt intergenerationally (e.g., climate change) and the analysis may accordingly extend beyond the current generation to include a robust discussion of far-future health effects and costs. In general, the period of time over which the analysis is conducted should also be consistent with other parts of the regulatory analysis.

The scope of the analysis should generally match the scope used in other parts of the regulatory analysis (e.g., benefit-cost analysis). However, in some situations, using a different time horizon or spatial scale may be appropriate when considering EJ. For example, phasing in of regulatory requirements or relocation of polluting activities in response to the regulatory action may result in EJ concerns due to effects that occur on a time horizon or spatial scale that differs from other effects considered in the regulatory analysis. If such situations arise, analysts should clearly articulate the reasons for considering an alternative time horizon.

Another aspect of characterizing temporal scope is adequately anticipating the long run dynamic effects of a regulatory action (Cain et al., 2024). The literature uses spatial sorting models to examine how regulations may affect residential location choice but typically focuses on a specific city or region (e.g., Kuminoff et al., 2015; Redding and Rossi-Hansberg, 2017).[13,14] Spatial sorting can occur when improved environmental quality is capitalized into housing values, attracting higher-income households and shifting renters and lower-income households to less expensive neighborhoods with lower environmental quality (Melstrom and Mohammadi, 2022). On the other hand, some residents may be more likely to move into high-risk zones due to differences in housing prices (Bakkensen and Ma, 2020). Given the challenges of modeling these types of effects on a national scale, it is recommended that analysts qualitatively discuss possible household responses based on the available literature, while acknowledging the limitations of the analysis.

10.2.7.2 Comparison Population Group

To evaluate differences in effects for population groups of concern, results need to be presented relative to another group, typically referred to as a comparison population group. How the comparison population group is selected has important implications for evaluating differences in health, risk, or exposure effects across population groups of concern. It is possible to define the comparison population group as individuals with similar socioeconomic characteristics in areas of the state, region, or nation unaffected by the regulatory action (i.e., within-group comparison) or as individuals with different socioeconomic characteristics within the affected areas (i.e., across-group comparison).

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13 One exception is Fan, et al (2018). They link spatial sorting and economy-wide models of the United States to explore where people migrate in response to increased risk of extreme temperatures, while accounting for wage and housing price feedbacks.

14 Likewise, while hedonic price methods may be useful for demonstrating how changes in environmental quality factor into housing prices, predicting the effect of such price changes on household migration by race or income may be infeasible.

Analysts should aim to define the comparison population group for an across-group comparison as similar as possible to the population group of concern, but without the socioeconomic characteristics defining the group of concern. For example, analysts could compare the proportion of low-income households within areas affected by the regulatory action to the proportion of non-low-income households within the same affected areas. If analysts have fate-and-transport information on emissions, they can compare the average concentrations faced by low-income households within the affected areas to those faced by non-low-income households living in the same areas. Thus, the results from an across-group comparison indicate how the likelihood of risk or exposure within the affected areas varies with demographic and socioeconomic characteristics.

A within-group comparison compares the likelihood of risk or exposure for a specific demographic or socioeconomic group in affected areas to the likelihood of risk or exposure for that same demographic or socioeconomic group elsewhere. Again, analysts should aim for the comparison group to be as similar as possible to the population group of concern but without the risk or exposure of interest. For example, analysts can compare the proportion of low-income households within areas affected by the regulatory action to the proportion of low-income households in unaffected areas. Similarly, if analysts have information on the fate and transport of emissions, they can compare the average concentrations faced by low-income households within the affected areas to those faced by low-income households living in areas unaffected by the regulatory action.

If a regulatory action is expected to differentially affect populations within a given area (e.g., communities living near regulated facilities or in a specific region), then a combination of within- and across-group comparisons can demonstrate whether there are differences between specific population groups of concern and the general population. In some contexts, it may make sense to define the comparison population group at a sub-national level to reflect differences in socioeconomic composition across geographic regions. See Section 6.5 of the *EJ Technical Guidance* (U.S. EPA, 2024a) for a more discussion of selecting the appropriate comparison group.

10.2.7.3 Spatial Identification and Aggregating Effects

The spatial distribution of health and welfare outcomes is a relevant consideration for some regulatory actions, such as those that reduce emissions from point sources that have fairly localized effects or when there is a differential distribution of associated health or environmental effects. In other cases, the regulatory action's effects may be more widespread, and spatial distribution is less relevant (e.g., when exposure to a chemical substance depends on its purchase, use, transport, or disposal).

When exposures, risks, or human health effects are spatially distributed, analysts need to determine how to spatially identify affected and unaffected populations. The nature of the stressor(s) should guide analysts' choices of the geographic area of analysis. Some air pollutants, for example, may be emitted out of tall stacks and travel long distances, affecting individuals hundreds of miles away from the sources and thereby making it appropriate to choose a relatively large geographic area. In contrast, water pollutants or waste facilities may have more localized effects, making it appropriate to select relatively small areas for analysis. Likewise, an assessment of local effects from point sources – including possible traffic, odors, and noise implications from changes in production – may call for more spatially resolved data than those that affect regional air quality.

Complications can arise when the spatial resolution of the analysis is either too refined or too coarse. See the *EJ Technical Guidance* (U.S. EPA, 2024a) for a discussion of these challenges. Analysts are encouraged to discuss the approach used to create buffers and aggregate geospatial data, as what is most appropriate will vary with the stressor(s) affected and data used in the analysis, and to provide a transparent justification of their choice. In some cases, it may be helpful to consider multiple buffers to evaluate the effects of a regulatory action, for instance, because of uncertainty regarding fate and transport of a specific environmental stressor or because the regulatory action affects environmental stressors that travel different distances.

10.2.7.4 Facility vs Community-Based Perspectives

Exposure to other environmental and non-environmental stressors can increase the vulnerability of individuals or population groups to negative health effects from exposure to a specific environmental hazard. While explicit modeling of these interactions is often not feasible, analysts can shed light on this issue by evaluating and presenting results using not only a facility, but also a community-based perspective.

An analysis with a facility-based perspective primarily considers who may be exposed to sources regulated by the specific action under consideration. For example, such an analysis would examine proximity, emissions, concentrations, or risk associated with each regulated source in conjunction with the demographic and socioeconomic characteristics of those most likely to be exposed.

However, communities may be affected by multiple sources of pollution relevant to characterizing risk for a specific regulatory action. An analysis that takes a community-based perspective considers proximity, emissions, concentrations, or risk to a given community from multiple nearby sources of pollution to which individuals are exposed, accounting for the possibility that certain communities face increased vulnerability due to a greater number of nearby pollution sources.

10.2.7.5 Evaluating Underlying Variability and Identifying Potential Hot Spots

In addition to presenting aggregate results for population groups of concern affected by the regulatory action, it is important to understand the extent to which there are heterogeneous effects, both within specific population groups as well as across communities, given that communities often vary widely in the risks they face from the affected sources as well as from other environmental and non-environmental stressors. When data allow, analysts should characterize the distribution of risks, exposures, or outcomes within each population group of concern, not just average effects, with particular attention paid to the characteristics of populations at higher risk of exposure. When relying on proximity-based analysis, differentiating results by key facility characteristics that may be correlated with risk (e.g., plant age, capacity, production levels, accident history, types of chemicals stored on site) can be useful.

It is also important to evaluate the potential for hot spots, with particular attention paid to the communities in the upper end of the distribution of exposure or risk. Hot spots refer to geographic areas with higher levels of localized concentrations of emissions from one or more sources within a larger geographic area with more “normal” environmental quality. Hot spots may result from baseline conditions, such as exposure to other pre-existing stressors within the community. It is also possible that hot spots may be created, exacerbated, or mitigated following a regulatory action. Relevant issues to consider may include proximity to multiple sources of pollution, specific exposure pathways, and other drivers of increased vulnerability. Qualitative or other sources of data may also help to identify specific population groups or communities where a more detailed analysis is warranted.

10.2.8 Characterizing Analytic Results

Once an EJ analysis has been conducted, analysts face choices about how to characterize and communicate the results. Analysts need to present summary metrics for relevant population groups of concern and the comparison population group and characterize the differences between them. This section discusses the way in which information from the analysis can be summarized and presented, including the choice of summary metrics, ways of displaying the results in tables, maps or other visual displays, and the distinction between statistical and policy significance when interpreting results.

10.2.8.1 Summary Metrics

Simple summary measures can be used to characterize the distribution of health and environmental effects in the baseline and for regulatory options relative to appropriate comparison population groups. Analysts should consider characterizing results of the EJ analysis using more than one type of summary metric to provide a richer picture of potential effects. For instance, relative ratios can facilitate comparisons across groups or locations because all ratios are in common units. However, without presenting information on the absolute levels of risk or exposure, it is not possible to determine if either group is at risk of experiencing a potential health effect. Analysts should also present information that communicates underlying heterogeneity in the data, such as the degree of spread in the data relative to the mean (i.e., standard deviation).

Counts of the number of sources or geographic areas where the percent of a specific population group living nearby exceeds a particular threshold (e.g., the state/national average or a specific percentile) are not recommended. Counts are hard to interpret because they do not account for differences in population size or density across geographic areas. It is more informative to display metrics that characterize the full population or risk distribution to understand the extent to which affected communities differ from the comparison group. See Section 6.6 of the *EJ Technical Guidance* (U.S. EPA, 2024a) for more discussion of summary metrics.

10.2.8.2 Displaying Results Visually

Tables, maps, and other visual displays help communicate a large amount of information in an organized way to facilitate comparisons, convey results, and support discussion. Careful thought should go into how information is presented, particularly when there are:

- Multiple comparison groups (e.g., state, U.S., rural areas);
- Different types of effects (e.g., pollutants, health effects, or other environmental metrics);
- Multiple categories of regulated facilities or types of sources;
- Many individual sources;
- Clustering of sources in specific geographic areas;
- Multiple scenarios (e.g., baseline, multiple regulatory options); or
- Sensitivity analysis around key analytic assumptions (e.g., buffer distance).

Analysts need to clearly explain how to interpret the information presented in tables, maps, or figures to properly contextualize results and guard against erroneous conclusions (e.g., a large percentage change from a small baseline value may not be a large change in absolute terms). Often more than one table is needed to present results. In addition, bolding or shading specific cells can ease navigation of a dense table of results. Table 10.1 illustrates how results for multiple types of sources and several distance buffers can be presented within a single table. This example also uses shading to indicate values above the national average.

Visually displaying information in maps or figures can also help demonstrate how sources, risks, and exposures are geographically distributed across population groups, including baseline conditions and spatial clustering of sources. Note that it can be difficult to visually discern differences between baseline and regulatory options in maps or figures unless differences are large.[15] However, differences not discernible on a map may still be important.

Additionally, it is important to consider how visual indicators can be used to characterize data uncertainty, how geographic boundaries relevant to the analytic context (e.g., watershed, state, or Tribal lands) are identified on a map, and how to select appropriate intervals for visually representing the distribution of the data. For this reason, visual displays are only suggestive of potential effects and should be accompanied by tables or other graphics that allow the reader to access the underlying statistical information.

10.2.8.3 Statistical Significance and Other Considerations

Tests of statistical significance can be used to examine whether the difference between the mean values of two groups is due to factors other than chance. This can be done for a pairwise comparison, which does not control for other factors, or via a regression approach, which allows analysts to assess the relationship between two variables while controlling for other factors.

It may also be useful to examine parts of the distribution further from the mean (e.g., quantile approaches) or to use approaches that can account for outliers, skewness or heteroscedasticity (varying levels of spread) in the data. Note that the ability to test statistical significance is predicated on having a sufficiently large sample size and, for parametric approaches, an assumed distribution (e.g., normality).

It is important to understand that a statistical difference does not necessarily indicate that the difference is meaningful from a policy perspective. For instance, analysts may find that low-income households are more likely to be located near a pollution source than wealthier households, and that this effect is statistically significant (i.e., the effect is statistically distinguishable from zero and not due to sampling error). However, the difference in likelihood between these types of households could still be quite small in magnitude. Analysts need to examine what the difference implies (e.g., how different poverty is across geographic areas), and summarize those differences in a manner appropriate for policy relevance.

15 For an overview of general mapping best practices to communicate EJ concerns, such as selecting a projection, avoiding unintentional misrepresentation, and choosing a color scale to represent values, see Stieb et al. (2019).

Table 10.1 - Example Summary Table for Proximity-Based Analysis Results

Race	Population within 1 Mile of Sites with Legacy CCR SIs	Population within 3 Miles of Sites with Legacy CCR SIs	Population within 1 Mile of Sites with CCRMUs	Population within 3 Miles of Sites with CCRMUs	U.S. Population
Asian	10.36%	4.66%	2.37%	2.82%	5.64%
Black or African American	13.47%	17.03%	8.37%	13.73%	12.53%
Native Hawaiian/Pacific Islander	.03%	.08%	.06%	.07%	.18%
Native American or Alaskan Native	.79%	.93%	.86%	.78%	.82%
Other	16.65%	15.82%	20.63%	18.94%	12.82%
White	58.47%	61.47%	67.71%	63.67%	68.01%
Ethnicity	**Population within 1 Mile of Sites with Legacy CCR SIs**	**Population within 3 Miles of Sites with Legacy CCR SIs**	**Population within 1 Mile of Sites with CCRMUs**	**Population within 3 Miles of Sites with CCRMUs**	**U.S. Population**
Hispanic (any race)	26.27%	22.0%	32.61%	27.02%	19.24%
People of Color	**Population within 1 Mile of Sites with Legacy CCR SIs**	**Population within 3 Miles of Sites with Legacy CCR SIs**	**Population within 1 Mile of Sites with CCRMUs**	**Population within 3 Miles of Sites with CCRMUs**	**U.S. Population**
People of Color	52.42%	46.31%	46.70%	46.59%	41.14%
Poverty Level	**Population within 1 Mile of Sites with Legacy CCR SIs**	**Population within 3 Miles of Sites with Legacy CCR SIs**	**Population within 1 Mile of Sites with CCRMUs**	**Population within 3 Miles of Sites with CCRMUs**	**U.S. Population**
Households below the poverty level	16.21%	16.11%	14.94%	14.9%	12.71%
Other Sociodemographic Factors	**Population within 1 Mile of Sites with Legacy CCR SIs**	**Population within 3 Miles of Sites with Legacy CCR SIs**	**Population within 1 Mile of Sites with CCRMUs**	**Population within 3 Miles of Sites with CCRMUs**	**U.S. Population**
Linguistically isolated households	9.23%	5.42%	9.38%	6.05%	4.84%
Less than a high school diploma	17.60%	14.02%	17.11%	15.57%	11.24%
Person with disability	15.53%	15.61%	14.66%	15.23%	12.70%

Source: Table 6-9. Estimated Percent of Key Sociodemographic Indicators Near Legacy CCR Surface Impoundment (SI) and CCR Management Unit (MU) Sites (U.S. EPA, 2024c).

Finally, it is important to address and characterize uncertainty. Point estimates alone do not provide information about whether estimates are robust to alternate assumptions, nor can they convey the full range of potential outcomes. When statistical analysis is used, information such as confidence intervals and variance should be presented. Sensitivity analysis can also play a role in understanding the robustness of outcomes to key assumptions. Where the analysis is sensitive to the choice of model or method used, this uncertainty should also be described. Uncertainty can also be discussed by highlighting limitations in the literature, identifying caveats associated with results, or highlighting gaps in the data. See Section 6.6 of the *EJ Technical Guidance* (U.S. EPA, 2024a) for additional discussion.

10.2.9 Assessing the Distribution of Costs and Other Effects

This section addresses when it may be appropriate to evaluate how economic costs or challenges are distributed across population groups, how compliance and enforcement may vary across regulatory options under consideration, and the evaluation of non-health effects. We refer to costs as defined Chapter 8.

10.2.9.1 Distribution of Economic Costs

Certain directives (e.g., E.O. 13175, E.O. 14008, and OMB Circular A-4) identify the distribution of economic costs or challenges as an important consideration in regulatory analysis. The economics literature also typically considers both costs and benefits when evaluating distributional consequences of an environmental policy to understand its net effects. In the context of EJ, the distribution of health or environment effects alone might convey an incomplete – and potentially biased – picture of the overall burden faced by population groups of concern. For instance, if costs are unevenly distributed such that low-income households bear a larger relative share, it is possible that they may experience net costs even after accounting for environmental improvements.

Fullerton (2011) discusses six possible types of distributional effects that may result from an environmental policy: costs to consumers via change in relative product prices; cost to producers or factors of production via changes in the relative returns to capital and labor; the distribution of scarcity rents (i.e., excess benefits due to restricted nature of a good, such as pollution permits); the distribution of environmental quality improvements; temporary costs of adjustment and transition (e.g., for capital and labor); and the capitalization of environmental improvements into asset prices (e.g., land or housing values). That said, the consideration of economic costs in an EJ context may be challenging, given a lack of data and methods in many instances.

Whether to undertake an analysis of economic costs as it pertains to EJ is a case-by-case determination. It will depend on the relevance of the information for the regulatory decision at hand, the likelihood that economic costs of the regulatory action will be concentrated among particular types of households, and the availability of data and methods to conduct the analysis.[16] Analysts should coordinate with economists from the Office of Policy when evaluating the potential relevance of economic costs for EJ and the degree to which they can be discussed or analyzed.

In many cases, analysis of economic costs from an EJ perspective will not substantially alter the assessment of distributional effects for population groups of concern. For instance, often the costs of regulatory action are passed onto consumers as changes in prices or wages that are spread fairly evenly across many households. When these price changes are small, the effect on an individual

16 Note that there may be other effects of a regulatory action (e.g., employment) beyond direct compliance and social costs but understanding how all effects vary across population groups may not be feasible. For example, data on the distribution of changes in employment across low-income households may be difficult to assess. See Chapter 9.

household also will likely be relatively small. In this case, further analysis is unlikely to yield additional insights.

However, in some circumstances further exploration of the distribution of economic costs may offer substantial insight because costs are expected to differentially affect specific population groups. For example, further analysis may be warranted when costs to comply with the regulatory action represent a noticeably higher proportion of income for some population groups; when some population groups are less able to adapt to or substitute away from goods or services with now higher prices; when changes in environmental quality or health and costs are likely to accrue to the same set of individuals; when costs are concentrated on some types of households (e.g., renters) or in specific geographic areas; when there are identifiable plant closures in or relocation of facilities away from or into communities in which population groups of concern reside and work; or when behavioral changes in response to the costs of the regulatory action leave population groups of concern less protected than other groups.

While the Agency continues to investigate ways to improve incorporation of economic costs into an analysis of EJ concerns, it recognizes that, even in cases where the information is relevant, data or methods may not exist for full examination of the distributional implications of costs. In these instances, the issue can be qualitatively discussed, and the limitations and assumptions associated with characterizing costs explained. See the *EJ Technical Guidance* (U.S. EPA, 2024a) for further discussion.

10.2.9.2 Considering Compliance and Enforcement

Evidence suggests that compliance with environmental regulations can vary widely across sources in ways that exacerbate pre-existing disparities (e.g., Allaire et al. 2018; Balazs et al. 2012; Fedinick et al. 2019; McDonald and Jones, 2018). Analysts may want to consider whether regulated sources have a history of significant non-compliance or enforcement actions taken against them under various statutes or how capacity for monitoring and enforcement may differ across communities, including those on Tribal lands. Past compliance issues may indicate pre-existing EJ concerns that warrant further investigation.[17]

Analysts are encouraged to consider differences in compliance and ease of enforcement across regulatory options in the EJ analysis. When there are pre-existing differences in risk or exposure, options consistent with applicable law that improve monitoring coverage or encourage compliance can reduce exposure in communities with EJ concerns (e.g., enhanced reporting requirements for higher risk sources). Collecting, processing, and making publicly available real-time monitoring or remotely sensed data may also be effective for enhancing public awareness and participation (U.S. EPA 2021).

10.2.9.3 Other Effects and Considerations

The distribution of non-health effects associated with environmental stressors affected by the regulatory action may also be important to consider. For instance, certain population groups may place a higher value on a cultural resource (e.g., spiritual or sacred sites). If a regulatory option affects those resources, then the groups with a higher value will experience a different effect than

17 There is also a literature that explores whether the intensity of enforcement activities for environmental regulations varies with demographics such as race and income (Konisky et al. 2021; Shadbegian and Gray 2012).

groups that do not place a value on the cultural resource. Likewise, some regulatory options may differentially affect access to specific recreational activities for some population groups.

Quantifying changes in non-health effects may be challenging. Often, data on the distribution of baseline conditions for non-health effects are not easily available or are difficult to quantify, and/or are not suitable for analyzing the effects of a regulatory action. For instance, data on some ecosystem services (e.g., cultural uses of specific ecosystems) in the United States are quite limited in availability compared to baseline health data, such as mortality incidence. Likewise, data and models to assess how various regulatory options affect non-health related endpoints may not be available.

10.3 Environmental Health for Children and Older Adults

Analysis may shed light on differential effects of regulation on children and older adults, both of which are life-stage defined groups characterized by a multitude of unique behavioral, physiological and anatomical attributes. EO 13045 requires that each federal agency address disproportionate health risks to children. In addition, EPA's Children's Health Policy (U.S. EPA 1995) requires the Agency to "consider the risks to infants and children consistently and explicitly as a part of risk assessments generated during its decision-making process, including the setting of standards to protect public health and the environment."[18]

There are two sets of important differences between children and adults regarding health effects. First, there are differences in exposure to pollutants and in the nature and magnitude of health effects resulting from the exposure. Children may be more vulnerable to environmental exposures than adults because their bodily systems are still developing; they eat, drink and breathe more in proportion to their body size; their metabolism may be significantly different — especially shortly after birth; and their behavior can expose them more to chemicals and organisms (e.g., crawling leads to greater contact with contaminated surfaces, while hand-to-mouth and object-to-mouth contact is much greater for toddler age children). In addition, since children are younger, they have more time to suffer adverse health effects from exposure to contaminants. Second, individuals may systematically place a different economic value on reducing health risks to children than on reducing such risks to adults. In part this is because children cannot provide marginal willingness to pay values for their own risk reductions, unlike adults, so children's health risk valuation necessarily requires some model, implicit or explicit, about household decision making. These models differ in their implications for valuation. The perceived or actual effects of a given health outcome, too, may differ across children and adults. Empirical evidence also suggests that parents value a given risk reduction to themselves differently than to their children, with willingness to pay (WTP) for own risks generally valued less than those for children.[19]

Older adults also may be more susceptible to adverse effects of environmental contaminants due to differential exposures arising from physiological and behavioral changes with age, disease status and drug interactions, as well as the body's decreased capacity to defend against toxic stressors.

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18 See https://www.epa.gov/children/epas-policy-evaluating-risk-children for the original 1995 policy and the 2013 and 2018 reaffirmation memos.

19 See Gerking and Dickie (2013) for a review of both household decision making models for children's health risk valuation and the empirical literature. U.S. EPA (2003) provides an overview of children's health valuation issues in applied analysis.

Generally, many of the approaches described earlier in this chapter to characterize the distribution of impacts may be adapted to evaluate environmental health risks by life stage.[20] For example, when proximity-based analysis is appropriate for evaluating EJ impacts, it might also be used to examine whether children or older adults are disproportionately located near facilities of concern. In such a case, the considerations described earlier about geography, defining the baseline and comparison groups, and use of summary statistics would all apply.

10.3.1 Age as a Life Stage

Evaluating the impacts of regulatory actions on children or older adults differs in an important way from evaluating the same impacts on population groups of concern for EJ. For instance, when the EPA evaluates disproportionate health risk impacts from environmental contaminants, it views childhood as a sequence of life stages from conception through fetal development, infancy and adolescence, rather than a distinct "subpopulation."

Use of the term "subpopulation" is ingrained in both EPA's past practices as well as various laws that the EPA administers such as the Safe Drinking Water Act Amendments. Prior to publication of revised risk assessment guidelines in 2005, the EPA described all groups of individuals as "subpopulations." In the 2005 guidelines, the Agency recognizes the importance of distinguishing between groups that form a relatively fixed portion of the population, such as those described in Section 10.2, and life stages or age groups that are dynamic groups drawing from the entire population.

The term "life stage" refers to a distinguishable time frame in an individual's life characterized by unique and relatively stable behavioral and/or physiological characteristics associated with development and growth. Since 2005, the EPA has characterized childhood as a life stage.[21]

10.3.2 Analytical Considerations

Assessing the consequences of policies that affect the health of children or older adults requires considerations that span risk assessment, action development and economic analysis. In each case, existing Agency documents can assist in the evaluation.

10.3.2.1 Risk and Exposure Assessment

Effects of pollution may differ depending upon age of exposure. Analysis of potentially disproportionate impacts begins with health risk assessment but also includes exposure assessment. Many risk guidance and related documents address how to consider children and older adults in risk and exposure assessment.

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20 In principle there is a potential distinction between factors that are fixed, such as race and sex, and those defined by lifestages. The latter raises the possibility, at least, of examining effects through the lens of differences in lifetime utility or well-being rather than focusing on a single life stage. See Adler (2008) for one proposal consistent with this approach.

21The 2005 Risk Assessment Guidelines "view childhood as a sequence of lifestages rather than viewing children as a subpopulation, the distinction being that a subpopulation refers to a portion of the population, whereas a life stage is inclusive of the entire population." (U.S. EPA 2005a).

A general approach to considering children and childhood life stages in risk assessment is found in *A Framework for Assessing Health Risks of Environmental Exposures to Children* (U.S. EPA 2006a). The framework identifies existing guidance, guidelines and policy papers that relate to children's health risk assessment. It emphasizes the importance of an iterative approach between hazard, dose response and exposure analyses. In addition, it includes a discussion of principles for weight-of-evidence consideration — that is, the critical evaluation of available and relevant data — across life stages.

EPA's 2005 *Guidelines for Carcinogenic Risk Assessment (Cancer Guidelines)* (U.S. EPA 2005a) explicitly call for consideration of possible sensitive subpopulations and/or lifestages such as childhood. The *Cancer Guidelines* were augmented by *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA 2005b). Recommendations from this supplement include calculating risks utilizing life stage-specific potency adjustments in addition to life stage-specific exposure values which should be considered for all risk assessments.

EPA's *Child-Specific Exposures Handbook* (U.S. EPA 2008) and *Highlights of the Child-Specific Exposure Factors Handbook* (U.S. EPA 2009c) help risk assessors understand children's exposure to pollution. The handbook provides important information for answering questions about life stage specific exposure through drinking, breathing and eating. EPA's guidance to scientists on selecting age groups to consider when assessing childhood exposure and potential dose to environmental contaminants is identified in *Guidance on Selecting Age Groups for Monitoring and Assessing Childhood Exposures to Environmental Contaminants* (U.S. EPA 2005c).

While there is no standard framework for including economic and human health effects on older adults in an analysis of the impacts of regulation, the EPA stresses the importance of addressing environmental issues that may adversely impact them.[22] These considerations are highlighted in EPA's *Exposure Factors Handbook* (U.S. EPA 2011) and have led EPA's Office of Research and Development to consider an exposure factors handbook specifically for the aging (see U.S. EPA 2007). Additionally, the toxicokinetic and toxicodynamic impacts of environmental agents in older adults have been considered in EPA's document entitled *Aging and Toxic Response: Issues Relevant to Risk Assessment* (U.S. EPA 2005d).

10.3.2.2 Action Development

Disproportionate impacts during fetal development and childhood are considered in EPA guidance on action development, particularly the *Guide to Considering Children's Health When Developing EPA Actions: Implementing Executive Order 13045 and EPA's Policy on Evaluating Health Risks to Children* (U.S. EPA 2006b). The guide helps determine whether EO 13045 and/or EPA's Children's Health Policy applies to an EPA action and, if so, how to implement the Executive Order and/or EPA's Policy. The guide clearly integrates EPA's Policy on Children's Health with the Action Development Process and provides an updated listing of additional guidance documents.

22 There is a lack of broad agreement about when this life stage begins. The U.S. and other countries typically define this life stage to begin at the traditional retirement age of 65, but, for example, the U.N. has It begin at age 60 (U.S. EPA 2005d).

10.3.2.3 Economic Analysis

While these *Economic Guidelines* provide general information on BCA of policies and programs, many issues concerning valuation of health benefits accruing to children are not covered. Information provided in the *Children's Health Valuation Handbook* (U.S. EPA 2003), when used in conjunction with the *Guidelines*, allows analysts to characterize benefits and impacts of Agency policies and programs that affect children.

The *Handbook* is a reference tool for analysts conducting economic analyses of EPA policies when those policies are expected to affect risks to children's health. The *Handbook* emphasizes that regulations or policies fully consider the economic impacts on children, including incorporating children's health considerations into BCA, as well as a separate analysis focused on children.

Economic factors may also play a role in other analyses that evaluate children's environmental health impacts. For example, because a higher proportion of children than adults live in poverty, the ability of households with children to undertake averting behaviors might be compromised.[23] This type of information could inform the exposure assessment.

Analysis of who bears the costs and benefits of a policy also is complicated by the fact that individual life stages change over time. For instance, because children eventually grow into adults, health and other benefits of a policy that initially accrue mainly to children will also likely affect them as adults. Likewise, while the costs of a policy are initially borne by current adults, they will eventually be borne by the current set of children as they themselves become adults.

10.3.3 Intersection Between Environmental Justice and Children's Health

The burden of health problems and environmental exposures is often borne disproportionately by children from low-income communities and minority communities (e.g., Arcury et al. 2021; Israel et al. 2005; Lanphear et al. 1996; Mielke et al. 1999; Pastor et al. 2006; Schwartz et al. 2015). The challenge for the EPA is to integrate both EJ and life stage susceptibility considerations, particularly for children but also for older adults, where appropriate when conducting analysis. This is especially true when short-term exposure to environmental contaminants, such as lead or mercury, early in life can lead to life-long health consequences.

23 U.S. Census Historical Poverty Tables: People and Families - 1959 to 2018. https://www.census.gov/data/tables/time-series/demo/income-poverty/historical-poverty-people.html (accessed on April 1, 2020).

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Chapter 11 - Presentation of Analysis and Results

This chapter provides some general guidance for presenting analytical results to policy makers and others interested in environmental policy development. Economic analyses play an important role throughout the policy development process. From the initial, preliminary evaluation of potential options through the preparation of a final economic analysis document, economic analysts participate in an interactive process with policy makers. The fundamental goal of this process is to collect, analyze and present information useful for policy makers.

Economic analysis is often motivated by a desire to find an optimal outcome, such as a degree of stringency in a regulation, or a level of provision of a public good that yields the largest possible net benefits. Environmental statutes sometimes mandate criteria other than economic efficiency, such as best available control technology or lowest achievable emission rate. Policy makers rely on quantitative analysis to promulgate these approaches. In particular, they rely on analyses that delineate the costs, benefits or other impacts of a wide range of control options.

This guidance for presenting inputs, analyses and results applies to *all* stages of this process, not only to the final document embodying the completed economic analysis. Conveying uncertainty effectively and reporting critical assumptions and key unquantified effects to decision makers is critical at all points in the policy-making process.

This chapter begins by providing general guidance on how to present the results of economic analyses, with a particular emphasis on presenting benefits and costs, including those that cannot be quantified and/or put into dollar terms. The chapter then discusses the components, or inputs, of an economic analysis, and how their effect on the economic analysis can best be communicated.

11.1 Presenting Results of Economic Analyses

The presentation of the results of an economic analysis should be thorough and transparent. The reader should be able to understand:

- What the primary conclusions of the economic analysis are;
- Which benefits arise from the statutory objective of the regulation and which do not;

- How the benefits and costs were estimated;
- What the important non-quantified and/or non-monetized effects are;
- What key assumptions were made for the analysis;
- What the primary sources of uncertainty are in the analysis; and
- How those sources of uncertainty affect the results.

An economic analysis of regulatory or policy options should present all identifiable costs and benefits that are incremental to the regulation or policy under consideration.

Benefits and costs should be reported in monetary terms whenever possible. In reality, there are often effects that cannot be monetized, and the analysis needs to communicate the full richness of benefit and cost information beyond what can be put in dollar terms. Benefits and costs that cannot be monetized should, if possible, be quantified (e.g., expected number of adverse health effects avoided or improved biodiversity). Benefits and costs that cannot be quantified should be presented qualitatively (e.g., directional impacts on relevant variables). Section 11.1 contains more detailed guidance on presenting this information in the U.S. Environmental Protection Agency's (EPA's) economic analyses.

Agencies are also required to provide OMB with an accounting statement reporting benefit and cost estimates when sending over each economically significant rule. Analysts should rely upon these *Guidelines* and Circular A-4 for developing these estimates. Circular A-4 (2023) provides a suggested format for this accounting statement.[1]

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

- **Results from BCA.** Estimates of the net social benefits should be presented based on the benefits and costs expressed in monetary terms. Non-monetized and unquantifiable benefits and costs should also be included and described in the presentation.
- **Results from cost-effectiveness analysis (CEA).** Under OMB Circular A-4, CEA should generally be performed for rules in which the primary effect is human health or safety. Results of these analyses should also be presented when they are conducted.[2]
- **Results from economic impact analysis (EIA) and distributional assessments**. Results of the EIA should be reported, including predicted effects on prices, profits, plant closures, employment and any other effects. Distributional impacts for particular groups of concern, including small entities, governments and environmental justice populations should also be presented.

The relative importance of these three sections will depend on the policy and statutory context of the analysis.

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1 The sample accounting statement is on p. 91 of Circular A-4 (2023).

2 The Institute of Medicine (IOM) (2006) issued recommendations to regulatory agencies on how to perform health-based CEA. Examples of CEA can be found in appendices of several RIAs including those for particulate matter (PM) National Ambient Air Quality Standards (NAAQS) [see Appendix G listed at http://www.epa.gov/ttn/ecas/ria.html (accessed March 13, 2011)] and the Ground Water Rule [see Appendix H listed at http://www.epa.gov/safewater/disinfection/gwr/regulation.html (accessed March 13, 2011)].

11.1.1 Presenting the Results of Benefit-Cost Analyses

When presenting the results of a BCA, the expected benefits and costs of all analyzed options should be reported, including the proposed or finalized option and any alternatives. OMB's Circular A-4 (2023) recommends studying alternative levels of stringency in addition to the proposed or finalized 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 would be expressed in monetary terms, but this is almost never possible because of data gaps, unquantifiable uncertainties and other challenges. It is important not to exclude an important benefit or cost category from BCA even if it cannot be placed in dollar terms. Instead, such benefits and costs should be expressed quantitatively if possible (e.g., avoided adverse health impacts, number of species added). If important benefit or cost categories cannot be expressed quantitatively, they should be discussed qualitatively. Of course, care should be taken to avoid overlapping categories of benefits and costs and to avoid double-counting.

Quantifiable benefits and costs, properly discounted, should be compared to determine a regulation's net benefits, even if important benefits or costs cannot be monetized. However, an economic analysis should assess the likelihood that non-monetized benefits and costs would materially alter the net benefit calculation for a given regulation.

Incremental benefits, costs and net benefits of moving from less to more stringent regulatory alternatives should also be presented. If a regulation has particularly significant impacts on population groups of concern, the various options' incremental impacts on these groups or source categories should be reported. This should include a discussion of incremental changes in quantified and qualitatively described benefits and costs.

Given the number of potential models presented in Chapters 7 and 8, the analyst should take care to clearly indicate the correspondence between the benefit and cost estimates. For example, the cost analysis may include results from a general equilibrium model, but the benefit analysis may only include partial equilibrium effects. In this case, the cost side of the equation includes general equilibrium feedback effects while the benefit side does not. This difference should be clearly presented and explained.

The tables at the end of this chapter contain templates for presenting information on regulatory benefits and costs, including those that cannot be quantified or put into dollar terms. The analyst's primary goal, using these tables, is to communicate the full richness of benefit and cost information instead of focusing narrowly on what can be put in dollar terms. Some guiding principles for constructing these tables follow.

- **All meaningful benefits and costs, including benefits arising from the statutory objective of the regulation as well as other welfare effects, are included in all of the tables even if they cannot be quantified or monetized.** Not only does this provide consistency for the reader, but it also maintains important information on the context of the quantified and monetized benefits.
- **The types of benefits and costs are described briefly in plain terms to make them clearer to the public and to decision makers, and they should be well-defined and mutually exclusive, to the extent possible.** Benefits should be grouped in a manner consistent with the categories in Table 7.1 of Chapter 7, although the order and specific characterization can be expected to vary by rule as needed.

- **The benefits are expressed first in natural or physical units (i.e., numbers) to provide a more complete picture of what the rule accomplishes.** These units are not discounted as they would be in a CEA because the goal here is to describe what might be termed the "physical scope" of the rule's benefits. It may be the case that physical or natural units are not relevant for presenting costs.[3]
- **Explanatory notes accompany each benefit and cost entry** and can be used to describe whatever the most salient or important points are about scientific uncertainty, the type of benefit or cost, how it is estimated or the presentation.

The benefit categories in these templates (e.g., improved human health, improved environment and other benefits,) will need to be revised to reflect the benefits categories for the rule under consideration. Likewise, cost categories may need to be revised to match the circumstances of the individual rule. Simpler analyses may need only the overview (Table 11.1) and the final summary (Table 11.4).

Table 11.1 is a quick-glance summary of regulatory benefits and costs, the extent to which they could be quantified and monetized, and a reference to where they are more fully characterized or estimated in the economic analysis. Some benefits may be described only qualitatively.

Table 11.2 reports benefits in non-monetary terms along with the units and additional explanatory notes. The goal of this table is to communicate the physical scope of the regulation's benefits rather than the dollar equivalent. Benefits here do not need to be discounted to present value, but the time associated with the quantities should be made clear (e.g., "annual" or "more than 10 years").

Table 11.3 reports benefits and costs in monetary terms along with totals for dollar-valued benefits and costs. Here it is important to specify the reference year for the dollars (i.e., real terms), the discount rate(s) used and the unit value and/or source.

Table 11.4 contains a template for bringing all this information together in summary that includes the type of benefit or cost, how it is measured, its quantity and dollar benefits. When multiple regulatory options are included in this table, it is appropriate for including in the regulatory preamble as requested by OMB.

Consistent with recommendations in these *Guidelines* for communicating uncertainty, quantitative entries should generally include a central or best estimate in addition to a range or confidence interval. The ability to do this, of course, may be limited by data availability.

The templates provided in Tables 11.1-11.4 presume that the regulatory action is designed to achieve health and environmental-protection benefits, albeit at some cost. In the case of a deregulatory action, the structure of the templates may need to be reversed.

3 Note that, as described in Chapter 6, the undiscounted stream of the non-monetized effects should be presented as they occur over time, and that these non-monetized effects generally should also still be discounted in benefit-cost analysis and cost-effectiveness analysis if they are aggregated over time. See Section 6.1.

Table 11.1 - Template for Regulatory Benefits and Costs Checklist

Benefits

Improved Human Health Benefits	**Effect can be Quantified?** (put in numeric terms)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Reduced incidence of adult premature mortality from exposure to PM2.5	✓	✓	e.g., see Section 5.2 of the economic analysis
Reduced incidence of fetal loss from reduced exposure to disinfection byproducts	✓	-	Notes and reference to section of the economic analysis
Unquantified human health benefit *with a brief description*	-	-	Notes and reference

Improved Environment Benefits	**Effect can be Quantified?** (put in numeric terms)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Fewer fish killed from reduced nutrient loadings into waterways	✓	✓	*Notes and reference*
Improved timber harvest from lower tropospheric ozone concentrations	✓	✓	*Notes and reference*
Other environmental benefit *with a brief description*	-	-	*Notes and reference*

Other Benefits	**Effect can be Quantified?** (put in numeric terms)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Reduced fuel expenditures from improved efficiency in automobiles and light trucks	✓	✓	*Notes and reference*
Other benefit *with a brief description*	-	-	*Notes and reference*

Costs

Compliance Costs (Fixed)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Research and Development investments to meet new standard	✓	*Notes and reference*
Capital Costs for new pollution control equipment	✓	*Notes and reference*

Compliance Costs (Variable)	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Operating Costs for pollution control equipment	✓	*Notes and reference*
Monitoring, reporting and recordkeeping costs associated with new requirements	✓	*Notes and reference*
Transaction costs	-	*Notes and reference*

Other Opportunity Costs	**Effect can be Monetized?** (put in dollar terms)	**More Information** (e.g., reference to section of the economic analysis)
Transition costs	-	*Notes and reference*
Reduced output in the regulated market	✓	*Notes and reference*
Other costs with brief description	-	*Notes and reference*

Table 11.2 - Template for Quantified Regulatory Benefits

Improved Human Health Benefits	**Quantified Benefits** (confidence interval or range)	Units	**More Information** (with possible reference to section of the economic analysis)
Reduced incidence of adult premature mortality from exposure to $PM_{2.5}$	estimate *(range)*	expected avoided premature deaths per year	e.g., range represents confidence interval
Reduced incidence of fetal loss from reduced exposure to disinfection byproducts	estimate *(range)*	expected avoided fetal losses per year	e.g., confidence interval cannot be estimated. Range based on alternative studies
Unquantified human health benefit *with a brief description*	*	*	e.g., data do not allow for quantification

Improved Environment Benefits	**Quantified Benefits** (confidence interval or range)	Units	**More Information** (with possible reference to section of the economic analysis)
Fewer fish killed from reduced nutrient loadings into waterways	estimate *(range)*	thousands of fish per year	Notes *(reference)*
Improved timber harvest from lower tropospheric ozone concentrations	estimate *(range)*	thousands of board feet per year	Notes *(reference)*
Other environmental benefit *with a brief description*	*	*	Notes *(reference)*

Other Benefits	**Quantified Benefits** (confidence interval or range)	Units	**More Information** (with possible reference to section of the economic analysis)
Fuel savings from improved efficiency in automobiles and light trucks	estimate *(range)*	millions of gallons of gasoline reduced per year	Notes *(reference)*
Other benefit *with a brief description*	*	*	Notes *(reference)*

*Note: * indicates the benefit cannot be quantified with available information.*

Table 11.3 - Template for Dollar-Valued Regulatory Benefits and Costs

Dollar-Valued Benefits

Improved Human Health Benefits	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information**
Reduced incidence of adult premature mortality from exposure to $PM_{2.5}$	$ estimate *($ range)*	e.g., $X based on Agency guidance	Notes *(reference)*
Reduced incidence of fetal loss from reduced exposure to disinfection byproducts	*	Not available	Notes *(reference)*
Unquantified human health benefit *with a brief description*	*	*	e.g., data insufficient to quantify *(reference)*

Improved Environment Benefits	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information**
Fewer fish killed from reduced nutrient loadings into waterways	$ estimate *($ range)*	e.g., $X based on WTP for recreational fishing	e.g., range reflects two different valuation approaches *(reference)*
Improved timber harvest from lower tropospheric ozone concentrations	$ estimate *($ range)*	e.g., change in consumer and producer surplus	e.g., estimated from market model across several species *(reference)*
Other environmental benefit *with a brief description*	*	*	Notes *(reference)*

Other Benefits	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information**
Fuel savings from improved efficiency in automobiles and light trucks	$ estimate *($ range)*	e.g., $X, based on net-of-tax average per gallon price	e.g., there is debate on how well fuel savings represent consumer benefits *(reference)*
Other benefit *with a brief description*	*	*Not available*	Notes *(reference)*
TOTAL Benefits that can be monetized ($ millions per year)	**$ estimate** *($ range)*	**$ estimate** *($ range)*	**$ estimate** *($ range)*

Costs

Compliance Costs (Fixed)	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information** *(with possible reference)*
R&D investments	$ estimate *($ range)*	e.g., $X based on industry survey	Notes *(reference)*
Capital Costs	$ estimate *($ range)*	-	e.g., *estimated from engineering cost models*
Compliance Costs (Variable)	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information** *(with possible reference)*
Operating Costs	$ estimate *($ range)*		e.g., *estimated from engineering cost models*
Monitoring, Reporting and Recordkeeping Costs	$ estimate *($ range)*	e.g., $X based on industry estimates	e.g., industry survey with 55% response
Transaction Costs	$ estimate *($ range)*		Notes *(reference)*
Other Opportunity Costs	**Dollar Benefits** *(millions per year)*	**Basis of Value**	**More Information** *(with possible reference)*
Transition Costs	$ estimate *($ range)*	-	Notes *(reference)*
Reduced output in the regulated market	$ estimate *($ range)*	-	Notes *(reference)*
Other Costs	$ estimate *($ range)*	-	Notes *(reference)*
TOTAL Costs that can be monetized ($ millions per year)	**$ estimate** *($ range)*	**$ estimate** *($ range)*	**$ estimate** *($ range)*

*Note: * indicates the benefit cannot be quantified with available information.*

Table 11.4 - Template for Summary of Benefits and Costs

Notes: e.g., "annual average numbers; millions 2022 dollars annualized at 3% discount rate," best estimate (with range).

Benefits

Improved Human Health Benefits	Option 1 *(#)*	Option 1 *($)*	Proposed or Finalized Option *(#)*	Proposed or Finalized Option *($)*	Option 3 *(#)*	Option 3 *($)*	Source, limitations or other key notes
Reduced incidence of adult premature mortality from exposure to $PM_{2.5}$	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	*highlight most important points, as needed*
Reduced incidence of fetal loss from exposure to disinfection byproducts	estimate *(range)*	*	estimate *(range)*	*	estimate *(range)*	*	e.g., no valuation data exist. Effects sensitive to dose-response model.
Unquantified human health benefit *with description*	*	*	*	*	*	*	e.g., risk data insufficient for quantification

Improved Environment Benefits	Option 1 *(#)*	Option 1 *($)*	Proposed or Finalized Option *(#)*	Proposed or Finalized Option *($)*	Option 3 *(#)*	Option 3 *($)*	Source, limitations or other key notes
Fewer fish killed from reduced nutrient loadings into waterways	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	*Notes*
Improved timber harvest from lower tropospheric ozone concentrations	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	*Notes*
Other environmental benefit *with description*	*	*	*	*	*	*	*Notes*

Other Benefits	Option 1 *(#)*	Option 1 *($)*	Proposed or Finalized Option *(#)*	Proposed or Finalized Option *($)*	Option 3 *(#)*	Option 3 *($)*	Source, limitations or other key notes
Fuel savings from improved efficiency in light duty vehicles	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	estimate *(range)*	*Notes*
Other benefit *with description*	*	*	*	*	*	*	*Notes*
TOTAL monetized benefits (annualized, millions $2022)	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	e.g., range may be overstated due to aggregation *(See Section 8.1)*

Costs

Compliance Costs (Fixed)	Option 1 *($ millions)*	Proposed or Finalized Option *(#)*	Option 3 *(#)*	Source, limitations or other key notes
R&D investments	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	Notes *(reference)*
Capital Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	e.g., *estimated from engineering cost models*

Compliance Costs (Variable)	Option 1 *($ millions)*	Proposed or Finalized Option *(#)*	Option 3 *(#)*	Source, limitations or other key notes
Operating Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	e.g., *estimated from engineering cost models*
Monitoring and Recordkeeping Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	e.g., industry survey with 55% response
Transaction Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	Notes *(reference)*

Other Opportunity Costs	**Option 1** *($ millions)*	**Proposed or Finalized Option** *(#)*	**Option 3** *(#)*	**Source, limitations or other key notes**
Transition Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	Notes *(reference)*
Other Costs	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	Notes *(reference)*
Reduced output in the regulated market	$ estimate *(range)*	$ estimate *(range)*	$ estimate *(range)*	Notes *(reference)*
TOTAL monetized Costs *(annualized, millions $2022)*	**$ estimate** *(range)*	**$ estimate** *(range)*	**$ estimate** *(range)*	-
TOTAL Net Benefits that can be monetized (annualized, millions $2022)	**$ estimate** (range)	**$ estimate** (range)	**$ estimate** (range)	-

*Note: * indicates the benefit cannot be quantified with available information.*

11.1.2 Presenting the Results of Cost-Effectiveness Analyses

When BCA is not possible, CEA may be the best available option. The cost-effectiveness of a policy 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.

The non-monetary measure of benefits used in a CEA must be chosen with great care to facilitate valid comparisons across options. The closer the chosen measure is to the variable that directly impacts social welfare, the better the measures will convey the weight of the consequences of the alternatives and the more robust a CEA will be. Consider the following steps that a typical environmental economic assessment follows:

- Changes in emissions are estimated (e.g., tons of emissions); then
- 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, and preferable, measure for CEA.

To illustrate, consider a typical air pollution scenario. Depending on where and when air pollutants are released into the atmosphere, a given ton of a particular pollutant can have widely divergent impacts on ambient air quality. Similarly, depending on when and where air quality changes, widely different levels of human health impacts may result. Particularly when different regulatory approaches are under consideration (e.g., regulation of different source categories in different locations), failing to standardize the analyses on the benefit measure that directly affects human health or welfare will significantly reduce the value of the analysis to decision makers (and the public).

When presenting the results of a CEA, the rationale for the selection of the non-monetary benefit measure must be described in detail. The presentation of results should also include a discussion of the limitations of the analysis, especially if an inferior measure, such as cost per ton of pollutant, must be used.

CEA is most useful when the policy or regulation in question affects a single endpoint. When multiple endpoints are affected (e.g., cancer and kidney failures), combining endpoints into a single effectiveness measure is impossible unless appropriate weighting factors exist for the multiple endpoints. The theoretically correct weights to apply are the dollar values associated with each endpoint, but generally it is the absence of these values that necessitates CEA. Therefore, it is not possible to compare a policy or regulation that reduces relatively more expected cancers, but fewer expected cases of kidney failure, with one that has the opposite relative effects. When this occurs, the effects of each option for each endpoint should be reported. A single endpoint may be selected for calculating cost-effectiveness, while other endpoints can be listed as ancillary benefits (or, if possible, their monetary value should be subtracted from the option's cost prior to calculating its cost-effectiveness) (OMB 2023).

The most cost-effective option — i.e., the option with the lowest cost per unit of benefit — is not necessarily the most economically efficient. Moreover, other criteria, such as statutory requirements, enforcement problems, technological feasibility or quantity and location of total emissions abated may preclude selecting the least-cost solution in a regulatory decision. However, where not prohibited by statute, CEA can indicate which control measures or policies are inferior options.

11.1.3 Presenting the Results of EIA and Distributional Analyses

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

The results of the EIA should also be reported for important sectors within the affected population — identifying specific segments of industries, regions of the country or types of firms that may experience significant impacts or plant closures and losses in employment.

Reporting the results in distributional assessments may include the expected allocation of benefits, costs or both for specific population groups of concern including those highlighted in the various mandates. These include minorities, low-income populations, small businesses, governments, not-for-profit organizations and vulnerable populations (including children). Where these mandates specify requirements that depend on the outcomes of the distributional analyses, such as the Regulatory Flexibility Act, the presentation of the results should conform to the criteria specified by the mandate.

11.2 Communicating Sources of Uncertainty

While guidance on performing uncertainty analysis is in Chapter 5, it is also important to consider how to communicate uncertainty in the 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.

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. Expert elicitation techniques, can help close knowledge gaps surrounding key relationships (see Chapter 5).

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.

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 that produces conclusions that can withstand close scrutiny is more likely to provide policy makers with the information they need to develop robust environmental policies.

11.2.1 Data

An economic analysis should clearly describe all important data sources and references used. Unless the data are confidential business information or some other form of private data, they should be available to policy makers, other researchers, policy analysts and the public. Providing documentation and access to the data used in an analysis is crucial to the credibility and reproducibility of the analysis.

EPA Order CIO 2105.0 (U.S. EPA 2023a and U.S. EPA 2023b) and the applicable federal regulations established a mandatory quality system for the 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.

In any economic analysis, there should be a clear presentation of how data are used and a concise explanation of why the data are suitable for the selected purpose. The data's accuracy, precision, representativeness, completeness and comparability should be discussed when applicable. When data are available from more than one source, a rationale for choosing the source of the data should be provided.

11.2.2 Model Choices and Assumptions

An economic analysis of an environmental regulation should carefully describe the models it relies on, the major assumptions made in running the models (to be discussed more fully below) and any areas of outstanding uncertainty. The analyst should take particular care to explain any results that might be viewed as counterintuitive. In particular, analysts should be careful not to accept model output blindly. Any model that is used without proper thought given to both its input and output may become a "black box" insofar as nonsensical results may result from a misspecified scenario, a coding error or any of a number of other causes.

In the process of conducting an economic analysis, it is sometimes necessary to bridge an information gap by making an assumption. Analysts should not simply note the information gap but should also justify the chosen assumption and provide a rationale for choosing one assumption over other plausible options. The analyst should take care not to overlook information gaps that are filled with a piece of information that is only slightly related to the desired information. Analysts are advised to keep a running list of assumptions. This will make it easier to identify “key assumptions” for the final report. The likely impact of errors in assumptions should be characterized both in terms of direction and magnitude of effect when feasible.

Maintaining a list of assumptions can benefit the analysis in several ways. In the short run, a list can serve to focus analysts’ attention on those assumptions with the greatest potential to affect net benefits, possibly leading to new approaches to bridging an information gap. In the long run, highlighting information gaps may encourage the EPA or others to devote attention and resources to generating that information.

Whenever the likely errors in a particular assumption can be characterized numerically or statistically, the factor is a good candidate for sensitivity analysis or uncertainty analysis, respectively. In many cases, only a narrative description of the impact of errors in assumptions is possible. The analyst should include a table that clearly lays out all of the key assumptions and the potential magnitude and direction of likely errors in assumptions in the summary of results.

11.2.3 Addressing Uncertainty Driven by Assumptions and Model Choice

Every analysis should address uncertainties resulting from the choices the analyst has made. For example, many economic analyses performed at the EPA include assessments of economic impacts expected to occur decades into the future. Estimates of the future costs and benefits of a regulation will be sensitive to assumptions about growth rates for populations, source categories, economic activity and technological change, as well as many other factors. Sensitivity analyses on key variables in the baseline scenario should be performed and reported when possible. This allows the reader to assess the importance of the assumptions made for the central case. Some of these variables may be affected by a regulation, particularly the assumed rate of technological innovation (see Chapter 5 for additional guidance on specifying baselines).

The impact of using alternative assumptions or alternative models can be assessed quantitatively in many cases through sensitivity analysis and presenting alternatives, as described in Chapter 5. 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 and choosing among them is similar to making an assumption. Implicit in the choice of a model are many factors. For example, one model may take long-run effects into account while another model does not. When possible, presenting results of an alternate model can inform the reader. When resource limitations prevent the use of an alternative model, it is still often possible to predict the direction and likely magnitude of the use of an alternate model, and the analyst should present this information to the reader.

11.3 Use of Economic Analyses

The primary purpose of conducting economic analysis is to provide policy makers and others with detailed information on a wide variety of consequences of environmental policies. One important element these analyses have traditionally provided to the policy-making process is estimates of social benefits and costs — the economic efficiency of a policy. For this reason, these *Guidelines* reflect updated information associated with procedures for calculating benefits and costs, monetizing benefits estimates and selecting particular inputs and assumptions.

Determining which regulatory options are best even on the restrictive terms of economic efficiency is often made difficult by uncertainties in data and by the presence of benefits and costs that can be quantified but not monetized, or that can only be qualitatively assessed. Even if the criterion of economic efficiency were the sole guide to policy decisions, social benefit and costs estimates alone would not be sufficient to define the best policies.

A large number of social goals and statutory and judicial mandates motivate and shape environmental policy. For this and other reasons, these *Guidelines* contain information concerning procedures for conducting analyses of other consequences of environmental policies, such as economic impacts and equity effects. This is consistent with the fact that economic efficiency is not the sole criterion for developing good public policies.

Even the most comprehensive economic analyses are but part of a larger policy development process, one in which no individual analytical feature or empirical finding dominates. The role of economic analysis is to organize information and comprehensively assess the economic consequences of alternative actions — benefits, costs, economic impacts and equity effects — and the trade-offs among them. Ultimately statutory requirements dictate if and how the analytic results are used in standard setting. In any case, these results, along with other analyses and considerations, serve as important inputs for the broader policy-making process and serve as important resources for the public.

Chapter 11 References

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Appendix A - Economic Theory

This appendix provides a brief overview of the fundamental theory underlying the approaches to economic analysis discussed in Chapters 3 through 9. The first section summarizes the basic concepts of the forces governing a market economy in the absence of government intervention. Section A.2 describes why markets may behave inefficiently. If the preconditions for market efficiency are not met, government intervention can be justified.[1] The usefulness of benefit-cost analysis (BCA) as a tool to help policy makers determine the appropriate policy response is discussed in Section A.3. Sections A.4 and A.5 explain how economists measure the economic impacts of a policy and set the optimal level of regulation. Section A.6 concludes and provides a list of additional references.

A.1 Market Economy

The economic concept of a market is used to describe any situation where exchange takes place between consumers and producers. Economists assume that consumers purchase the combination of goods that maximizes their well-being, or "utility," given market prices and subject to their household budget constraint. Economists also assume that producers (firms) act to maximize their profits. Economic theory posits that consumers and producers are rational agents who make decisions taking into account *all* of the costs — the full opportunity costs — of their choices, given their own resource constraints.[2] The purpose of economic analysis is to understand how the agents interact and how their interactions add up to determine the allocation of society's resources: what is produced, how it is produced, for whom it is produced and how these decisions are made. The simplest tool economists use to illustrate consumers' and producers' behavior is a market diagram with supply and demand curves.

The demand curve for a single individual shows the quantity of a good or service that the individual will purchase at any given price. This quantity demanded assumes the condition of holding all else constant (i.e., assuming the budget constraint, information about the good, expected future prices, prices of other goods, etc. remain constant). The height of the demand curve in Figure A.1 indicates the maximum price, P, an individual with Q_d units of a good or service would be willing to pay to acquire an additional unit of a good or service. This amount reflects the satisfaction (or utility) the individual receives from an additional unit, known as the *marginal benefit* of consuming the good. Economists generally assume that the marginal benefit of an additional unit is slightly less than that

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1 The EPA's mandates frequently rely on criteria other than economic efficiency, so policies that are not justified due to a lack of efficiency are sometimes adopted.

2 Opportunity cost is the next best alternative use of a resource. The full opportunity cost of producing (consuming) a good or service consists of the maximum value of other goods and services that could have been produced (consumed) had one not used the limited resources to produce (purchase) the good or service in question. For example, the full cost of driving to the store includes not only the price of gas but also the value of the time required to make the trip.

realized by the previous unit. The amount an individual is willing to pay for one more unit of a good is less than the amount she paid for the last unit; hence, the individual demand curve slopes downward. A market demand curve shows the total quantity that consumers are willing to purchase at different price levels (i.e., their collective willingness to pay (WTP) for the good or service). In other words, the market demand curve is the horizontal sum of all the individual demand curves.

Figure A.1 - Market and Total WTP

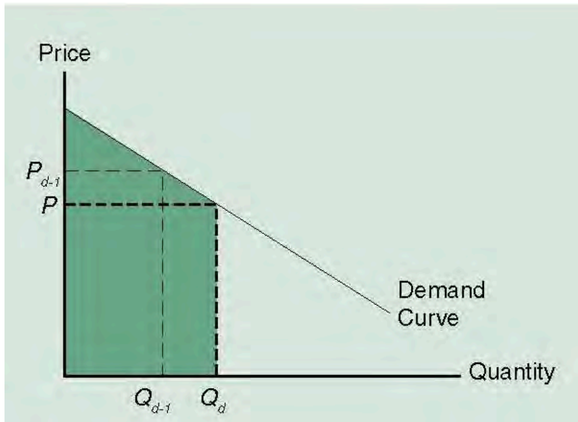
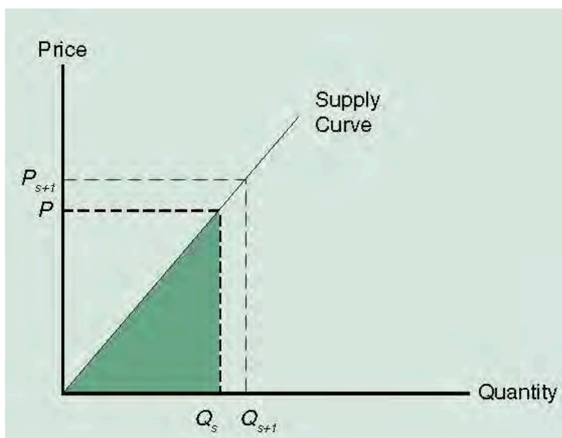


Figure A.2 - Marginal and Total Cost



The concept of an individual's WTP is one of the fundamental concepts used in economic analyses, and it is important to distinguish between total and marginal WTP. Marginal WTP is the additional amount the individual would pay for one additional unit of the good. The total WTP is the aggregate amount the individual is willing to pay for the total quantity demanded (Q_d). Figure A.1 illustrates the difference between the marginal and total WTP. The height of the demand curve at a quantity Q_{d-1} gives the marginal WTP for the Q_{d-1}th unit. The height of the demand curve at a quantity Q_d gives the marginal WTP for the Q_dth unit. Note that the marginal WTP is greater for the Q_{d-1}th unit. The *total* WTP is equal to the sum of the marginal WTP for each unit up to Q_d. The shaded area under the demand curve from the origin up to Q_d shows total WTP.

An individual producer's supply curve shows the quantity of a good or service that an individual or firm is willing to sell (Q_s) at a given price. As a profit-maximizing agent, a producer will only be willing to sell another unit of the good if the market price is greater than or equal to the cost of producing that unit. The cost of producing the additional unit is known as the *marginal cost*. Therefore, the individual supply curve traces out the marginal cost of production and is also the marginal cost curve. Economists generally assume that the cost of producing one additional unit is greater than the cost of producing the previous unit because resources are scarce. Therefore, the supply curve is assumed to slope upward. In Figure A.2, the marginal cost of producing the Q_sth unit of the good is given by the height of the supply curve at Q_s. The marginal cost of producing the Q_{s+1}th unit of the good is given by the height of the supply curve at Q_{s+1}, which greater than the cost of producing the Q_sth unit, and greater than the price, P. The *total cost* of producing Q_s units is equal to the shaded area under the supply curve from the origin to the quantity Q_s. The market supply curve is simply the horizontal summation of the individual producers' marginal cost curves for the good or service in question.

In a competitive market economy, the intersection of the market demand and market supply curves determines the equilibrium price and quantity of a good or service sold. The demand curve reflects the marginal benefit consumers receive from purchasing an extra unit of the good (i.e., it reflects their marginal WTP for an extra unit). The supply curve reflects the marginal cost to the firm of producing an extra unit. Therefore, at the competitive equilibrium, the price is where the marginal benefit equals the marginal cost. This is illustrated in Figure A.3, where the supply curve intersects the demand curve at equilibrium price P_m and equilibrium quantity Q_m.

A counterexample illustrates why the equilibrium price and quantity occur at the intersection of the market demand and supply curves. In Figure A.3, consider some price greater than P_m where Q_s is greater than Q_d (i.e., there is *excess supply*). As producers discover that they cannot sell off their inventories, some will reduce prices slightly, hoping to attract more customers. At lower prices consumers will purchase more of the good (Q_d increases) although firms will be willing to sell less (Q_s decreases). This adjustment continues until Q_d equals Q_s. The reverse situation occurs if the price becomes lower than P_m. In that case, Q_d will exceed Q_s (i.e., there is *excess demand*) and consumers who cannot purchase as much as they would like are willing to pay higher prices. Therefore, firms will begin to increase prices, causing some reduction in the Q_d but also increasing Q_s. Prices will continue to rise until Q_s equals Q_d. At this point no purchaser or supplier will have an incentive to change the price or quantity; hence, the market is said to be in equilibrium.

Economists measure a consumer's net benefit from consuming a good or service as the excess amount that she is willing to spend on the good or service over and above the market price. The net benefit of all consumers is the sum of individual consumer's net benefits — i.e., what consumers are willing to spend on a good or service over and above that required by the market. This is called the *consumer surplus*. In Figure A.3, the market demands price P_m for the purchase of quantity Q_m. However, the demand curve shows that there are consumers willing to pay more than price P_m for all units prior to Q_m.

Therefore, the consumer surplus is the area under the market demand (marginal benefit) curve but above the market price. Policies that affect market conditions in ways that decrease prices by decreasing costs of production (i.e., that shift the marginal cost curve to the right) will generally increase consumer surplus. This increase can be used to measure the benefits that consumers receive from the policy.[3]

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

Figure A.3 - Market Equilibrium

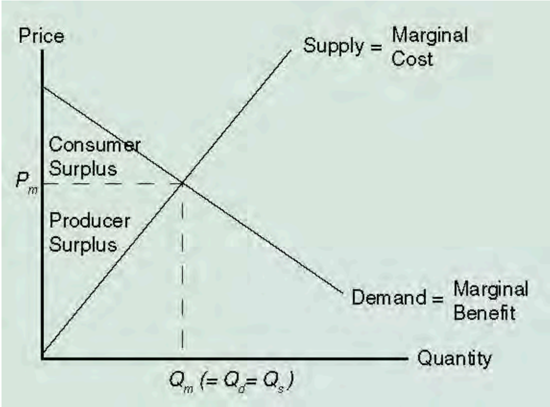
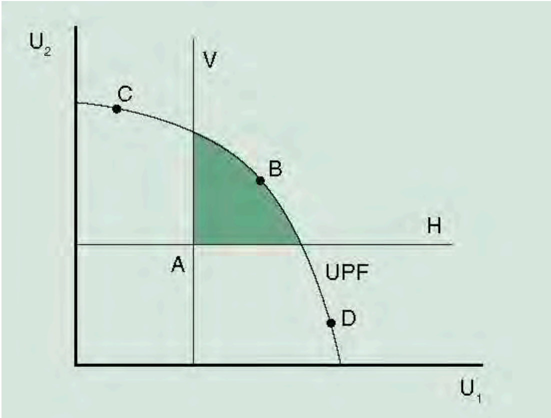


Figure A.4 - Utility Possibility Frontier



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 than the market price P_m. Hence, the net benefit to producers in this market, known as *producer surplus*, can be measured as the area above the market supply (marginal cost) curve but below the market price. Policies that increase prices by increasing market demand for a good (i.e., that shift the marginal benefit curve to the right) will generally increase producer surplus. This increase can be used to measure the benefits that producers receive from the policy.

Economic efficiency is defined as the maximization of social welfare. In other words, the efficient level of production is one that allows society to derive the largest possible net benefit from the market. This condition occurs where the (positive) difference between the total WTP and total costs is the largest. In the absence of externalities and other market failures (explained below), this occurs precisely at the intersection of the market demand and supply curves where the marginal benefit equals the marginal cost. This is also the point where total surplus (consumer surplus plus producer surplus) is maximized. There is no way to rearrange production or reallocate goods so that someone is made better off without making someone else worse off — a condition known as *Pareto optimality*. Notice that economic efficiency requires only that net benefits be maximized, *irrespective of to whom those net benefits accrue*. It does not guarantee an "equitable" or "fair" distribution of these surpluses among consumers and producers, or between sub-groups of consumers or producers.

Economists maintain that *if the economic conditions are such that there are no market imperfections* (as discussed in Section A.2), then this condition of Pareto-optimal economic efficiency occurs automatically.[4] That is, no government intervention is necessary to maximize the sum of consumer surplus and producer surplus. This theory is summarized in the two Fundamental Theorems of Welfare Economics, which originate with Pareto (1906) and Barone (1908):

1. **First Fundamental Welfare Theorem.** Every competitive equilibrium is Pareto-optimal.
2. **Second Fundamental Welfare Theorem.** Every Pareto-optimal allocation can be achieved as a competitive equilibrium after a suitable redistribution of initial endowments.

One graphical representation of these results is given in Figure A.4, which shows utility (welfare) levels in a two-person economy.[5] The curve shown is the utility possibility frontier (UPF) curve; the area within it represents the set of all possible welfare outcomes. Each point on the negatively sloped UPF curve is Pareto optimal since it is not possible to increase the utility of one person without decreasing the utility of the other. If the initial allocation is at point A, then the set of Pareto-superior (welfare-enhancing) outcomes include all points in the shaded area, bordered by *H*, *V* and the UPF curve.[6] If trading is permitted, the First Welfare Theorem applies and the market will move the economy to a superior, more efficient point such as *B*. Then the Second Welfare Theorem simply says that for any chosen point along the UPF curve, given a set of lump sum taxes and transfers, an initial allocation can be determined inside the UPF from which the market will achieve the desired outcome.[7]

A.2 Reasons for Market or Institutional Failure

If the market supply and demand curves reflect society's true marginal social cost and WTP, then a laissez-faire market (i.e., one governed by individual decisions and not government authority) will produce a socially efficient result. However, when markets do not fully represent social values, the private market will not achieve the efficient outcome (see Mankiw 2004, or any basic economics text); this is known as a *market failure*. Market failure is primarily the result of externalities, market power and inadequate or asymmetric information. Externalities are the most likely cause of the failure of private and public sector institutions to account for environmental damages.

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4 Technically, there are two types of efficiency. Allocative efficiency means that resources are used for the production of goods and services most wanted by society. Productive efficiency implies that the least costly production techniques are used to produce any mix of goods and services. Allocative efficiency requires that there be productive efficiency, but productive efficiency can occur without allocative efficiency. Goods can be produced at the least-costly method without being most wanted by society. Perfectly competitive markets in the long run will achieve both of these conditions, producing the "right" goods (allocative efficiency) in the "right" way (productive efficiency). These two conditions imply Pareto-optimal economic efficiency (see Varian 1992 or any basic economics text for a more detailed discussion),

5 Another, perhaps more commonly used, graphical tool to explain the First and Second Welfare Theorems is an Edgeworth box. See Varian (1992) or other basic economic textbook for a detailed discussion.

6 Note that efficiency could be obtained by moving along the vertical line V, which keeps utility of person 1 (U_1) constant while increasing utility of person 2 (U_2), or by moving along the horizontal line H, which only shows improvements in utility for person 1. Moving to point B improves the utility for both individuals.

7 Note that outcomes on the frontier such as C and D, although efficient, may not be desired on equity (or fairness) grounds.

Externalities occur when markets do not account for the effect of one individual's decisions on another individual's well-being.[8] In a free market producers make their decisions about what and how much to produce, taking into account the cost of the required inputs — labor, raw materials, machinery and energy. Consumers purchase goods and services taking into account their income and their own tastes and preferences. This means that decisions are based on the private costs and private benefits to market participants. If the consumption or production of these goods and services poses an external cost or benefit on those not participating in the market, however, then the market demand and supply curves no longer reflect the true marginal social benefit and marginal social cost. Hence, the market equilibrium will no longer be the socially (Pareto) efficient outcome.

Externalities can arise for many reasons. Transactions costs or poorly defined property rights can make it difficult for injured parties to bargain or use legal means to ensure that costs of damages caused by polluters are internalized into their decision making.[9] Activities that pose environmental risks may also be difficult to link to resulting damages and often occur over long time periods. Externalities involve goods that people care about but are not sold in markets.[10] Air pollution causes ill health, ecological damage and visibility impacts over a long time, and the damage is often far from the source(s) of the pollution. The additional social costs of air pollution are not included in firms' profit maximization decisions and so are not considered when firms decide how much pollution to emit. The lack of a market for clean air causes problems and provides the impetus for government intervention in markets involving polluting industries.

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8 More formally, an externality occurs when the production or consumption decision of one party has an unintended negative (positive) impact on the profit or utility of a third party. Even if one party compensates the other party, an externality still exists (Perman et al. 2003). See Baumol and Oates (1988) or any basic economics textbook for similar definitions and more detailed discussion.

9 A property right can be defined as a bundle of characteristics that confer certain powers to the owner of the right: the exclusive right to the choice of use of a resource, the exclusive right to the services of a resource and the right to exchange the resource at mutually agreeable terms. Externalities arise from the violation of one or more of the characteristics of well-defined property rights. This implies that the distortions resulting from an externality can be eliminated by appropriately establishing these rights. This insight is summarized by the famous "Coase theorem" which states that if property rights over an environmental asset are clearly defined, and bargaining among owners and prospective users of the asset is allowed, then externality problems can be corrected and the efficient outcome will result regardless of who was initially given the property right. The seminal paper is Coase (1960).

10 Often these are goods that exhibit public good characteristics. Pure public goods are those that are non-rivalrous in consumption and non-excludable (see Perman et al. (2003) for a detailed discussion of these, as well as congestible and open access resources — i.e., goods that are neither pure public nor pure private goods.) Because exclusive property rights cannot be defined for these types of goods, pure private markets cannot provide for them efficiently.

Figure A.5 - Negative Externality

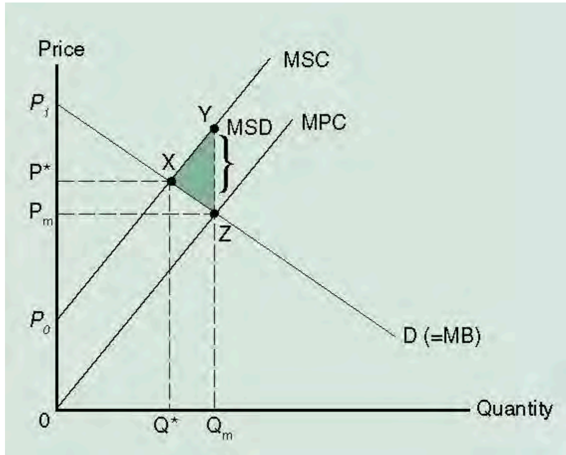


Figure A.5 illustrates a negative externality associated with the production of a good. For example, a firm producing some product might also be generating pollution as a by-product. The pollution may impose significant costs — in the form of adverse health effects, for example — on households living downwind or downstream of the firm. Because those costs are not borne *by the firm,* the firm typically does not consider them in its production decisions. Society considers the pollution a cost of production, but the firm typically will not. In this figure:

- *D* is the market demand (marginal benefit) curve for the product;
- *MPC* is the firm's marginal private real-resource cost of production, excluding the cost of the firm's pollution on households;
- *MSD* is the marginal social damage of pollution (or the marginal external cost) that the firm is not considering; and
- *MSC* is society's marginal social cost associated with production, including the cost of pollution (*MSC = MPC + MSD*).

In an incomplete market, producers pay no attention to external costs, and production occurs where market demand (*D*) and the marginal private real-resource cost (*MPC*) curves intersect — at a price P_m and a quantity Q_m. In this case, net social welfare (total WTP minus total social costs) is equal to the area of the triangle $P_0 P_1 X$ less the area of triangle *XYZ*.[11] If the full social cost of production, including the cost of pollution, is taken into consideration, then the marginal cost curve should be increased by the amount of the marginal social damage (*MSD*) of pollution.[12] Production will now occur where the demand and marginal social cost (*MSC*) curves intersect — at a price *P** and a quantity *Q**. At this point net social welfare (now equal to the area of the triangle, P_0P_1X, alone) is maximized, and therefore the market is at the socially efficient point of production. This example shows that when there is a negative externality such as pollution, and the social damage

11 Recall from Section A.1 that total WTP is equal to the area under the demand curve from the origin to the point of production ($0P_1ZQ_m$). Total costs (to society) are equal to the area under the MSC curve from the origin to the point of production ($0P_0YQ_m$).

12 When conducting BCA related to resource stocks, the MSD or marginal external cost is the present value of future net benefits that are lost to due to the use of the resource at present. That is, exhaustible resources used today will not be available for future use. These foregone future benefits are called user costs in natural resource economics (see Scott 1953, 1955). The marginal user cost is the user cost of one additional unit consumed in the present and is added together with the marginal extraction cost to determine the MSC of resource use.

(external cost) of that pollution is not taken into consideration, the producer will oversupply the polluting good.[13] The shaded triangle (*XYZ*), referred to as the *deadweight loss (DWL)*, represents the amount that society loses by producing too much of the good.

A.3 Benefit-Cost Analysis

If a negative externality such as pollution exists, an unregulated market will not account for its cost to society, and the result will be an inefficient outcome. In this case, there may be a need for government intervention to correct the market failure. A correction may take the form of dictating the allowable level of pollution or introducing a market mechanism to induce the optimal level of pollution.[14] Figure A.5 neatly summarizes this in a single market diagram. To estimate the *total* costs and benefits to society of an activity or program, the costs and benefits in each affected market, as well as any non-market costs or benefits, are added up. This is done through BCA.

BCA can be thought of as an accounting framework of the overall social welfare of a program, which illuminates the trade-offs involved in making different social investments (Arrow et al. 1996). It is used to evaluate the favorable effects of a policy action and the associated opportunity costs. The favorable effects of a regulation are the benefits, and the foregone opportunities or losses in utility are the costs. Subtracting the total costs from the total monetized benefits provides an estimate of the regulation's net benefits to society. An efficient regulation is one that yields the maximum net benefit, assuming that the benefits can be measured in monetary terms.

BCA can also be seen as a type of market test for environmental protection. In the private market, a commodity is supplied if the benefits that society gains from its provision, measured by what consumers are willing to pay, outweigh the private costs of producing the commodity. Economic efficiency is measured in a private market as the difference between what consumers are willing to pay for a good and what it costs to produce it. Since clean air and clean water are public goods, private suppliers cannot capture their value and sell it. The government determines their provision through environmental protection regulation. BCA quantifies the benefits and costs of producing this environmental protection in the same way as the private market, by quantifying the WTP for the environmental commodity. As with private markets, the efficient outcome is the option that maximizes net benefits.

The key to performing BCA lies in the ability to measure both benefits and costs in monetary terms so that they are comparable. Consumers and producers in regulated industries and the governmental agencies responsible for implementing and enforcing the regulation (and by extension, taxpayers in general) typically pay the costs. The total cost of the regulation is found by summing the costs to these individual sectors (see Section A.4.3 for an example, excluding the costs to the government). Since environmental regulation usually addresses some externality, the benefits of a regulation often occur *outside* of markets. For example, the primary benefits of drinking water regulations are improvements in human health. Once the expected reduction in illness and premature mortality associated with the regulation is calculated, economists use a

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

14 Chapter 4 discusses the various regulatory techniques and some non-regulatory means of achieving pollution control.

number of techniques to estimate the value that society places on these health improvements.[15] These monetized benefits can then be summed to obtain the total benefits from the regulation.

Note that in BCA gains and losses are weighted equally regardless of to whom they accrue. Evaluation of the fairness, or the equity, of the net gains cannot be made without specifying a social welfare function. However, there is no generally agreed-upon social welfare function and assigning relative weights to the utility of different individuals is an ethical matter that economists strive to avoid. Given this dilemma, economists have tried to develop criteria for comparing alternative allocations where there are winners and losers without involving explicit reference to a social welfare function. According to the Kaldor-Hicks compensation test, named after its originators Nicholas Kaldor and J.R. Hicks, a reallocation is a welfare-enhancing improvement to society if:

1. The winners could theoretically compensate the losers and still be better off; and
2. The losers could not, in turn, pay the winners to not have this reallocation and still be as well off as they would have been if it did occur (Perman et al. 2003).

While these conditions sound complex, they are met in practice by assessing the net benefits of a regulation through BCA. The policy that yields the highest positive net benefit is considered welfare enhancing according to the Kaldor-Hicks criterion. Note that the compensation test is stated in terms of *potential* compensation and does not solve the problem of evaluating the fairness of the distribution of well-being in society. Whether and how the beneficiaries of a regulation should compensate the losers involves a value judgment and is a separate decision for government to make.

Finally, BCA may not provide the *only* criterion used to decide if a regulation is in society's best interest. There are often other, overriding considerations for promulgating regulation. Statutory instructions, political concerns, institutional and technical feasibility, enforceability and sustainability are all important considerations in environmental regulation. In some cases, a policy may be considered desirable even if the benefits to society do not outweigh its costs, particularly if there are ethical or equity concerns.[16] There are also practical limitations to BCA. Most importantly, this type of analysis requires assigning monetized values to non-market benefits and costs. In practice it can be very difficult or even impossible to quantify gains and losses in monetary terms (e.g., the loss of a species, intangible effects).[17] In general, however, economists believe that BCA provides a systematic framework for comparing the social costs and benefits of proposed regulations, and that it contributes useful information to the decision-making process about how scarce resources can be put to the best social use.

A.4 Measuring Economic Impacts

A.4.1 Elasticities

The net change in social welfare brought about by a new environmental regulation is the sum of the negative effects (i.e., loss of producer and consumer surplus) and the positive effects (or social benefits) of the improved environmental quality. This is shown graphically for a single market in

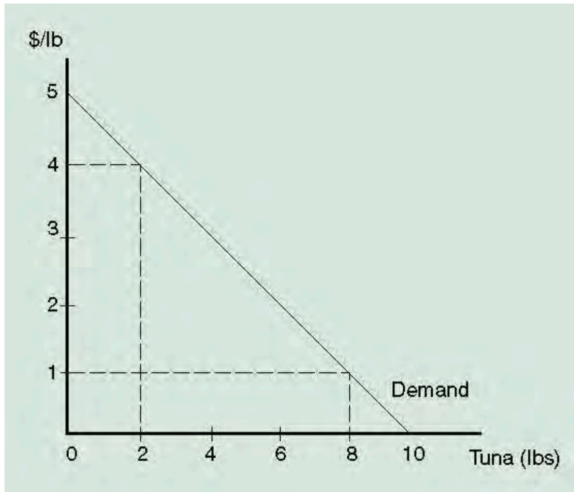
15 Chapter 7 discusses a variety of methods economists use to value environmental improvements.

16 Chapter 9 addresses equity assessment and describes the methods available for examining the distributional effects of a regulation.

17 Kelman (1981) argues that it is even unethical to try to assign quantitative values to non-marketed benefits.

Figure A.5 above. The use of demand and supply curves highlights the importance of assessing how individuals will respond to changes in market conditions. The net benefits of a policy will depend on how responsively producers and consumers react to a change in price. Economists measure this responsiveness by the supply and demand elasticities.

Figure A.6 - Demand Curve for Tuna



The term "elasticity" refers to the sensitivity of one variable to changes in another variable. The price elasticity of demand (or supply) for a good or service is equal to the percentage change in the quantity demanded (or supplied) that would result from a 1% 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. When calculating elasticities it is important realize where one is on the supply or demand curve, and the price or quantity should be stated when reporting an elasticity estimate.

Elasticities are important in measuring economic impacts because they determine how much of a price increase will be passed on to the consumer. For example, if a pollution control policy leads to an increase in the price of a good, multiplying the price increase by current quantity sold generally will not provide an accurate measure of impact of the policy. Some of the impact will take the form of higher prices for the consumer, but some of the impact will be a decrease in the quantity sold. The amount of the price increase that is passed on to consumers is determined by the elasticity of demand relative to supply (as well as existing price controls). "Elastic" demand (or supply) indicates that a small percentage increase in price results in a larger percentage decrease (increase) in quantity demanded (supplied).[18] All else equal, an industry facing a relatively elastic demand is less likely to pass on costs to the consumer because increasing prices will result in reduced revenues. In determining the economic impacts of a rule, supply characteristics in the industries affected by a regulation can be as important as demand characteristics. For highly elastic *supply* curves relative to the demand curves, it is likely that cost increases or decreases will be passed on to consumers.

The many variables that affect the elasticity of demand include:

- The cost and availability of close substitutes;
- The percentage of income a consumer spends on the good;
- How necessary the good is for the consumer;
- The amount of time available to the consumer to locate substitutes;
- The expected future price of the good; and
- The level of aggregation used in the study to estimate the elasticity.

The availability of close substitutes is one of the most important factors that determine demand elasticity. A product with close substitutes at similar prices tends to have an elastic demand, because consumers can readily switch to substitutes rather than paying a higher price. Therefore, a company is less likely to be able to pass through costs if there are many close substitutes for its product. Narrowly defined markets (e.g., salmon) will have more elastic demands than broadly defined markets (e.g., food) since there are more substitutes for narrow goods.

Another factor that affects demand elasticities is whether the affected product represents a substantial or necessary portion of customers' costs or budgets. Goods that account for a substantial portion of consumers' budgets or disposable income tend to be relatively price elastic. This is because consumers are more aware of small changes in the price of expensive goods compared to small changes in the price of inexpensive goods, and therefore may be more likely to seek alternatives. A similar issue concerns the type of final good involved. Reductions in demand may be more likely to occur when prices increase for "luxuries" or optional purchases. If the good is a necessity item, the quantity demanded is unlikely to change drastically for a given change in price. Demand will be relatively inelastic.

Elasticities tend to increase over time, as firms and customers have more time to respond to changes in prices. Although a company may face an inelastic demand curve in the short run, it could experience greater losses in sales from a price increase in the long run. Over time customers begin to find substitutes or new substitutes are developed. However, temporary price changes may affect consumers' decisions differently than permanent ones. The response of quantity demanded during a one-day sale, for example, will be much greater than the response of quantity demanded when prices are expected to decrease permanently. Finally, it is important to keep in mind that elasticities differ at the firm versus the industry level. It is not appropriate to use an industry-level elasticity to estimate the ability of only one firm to pass on compliance costs when its competitors are not subject to the same cost.

Characteristics of supply in the industries affected by a regulation can be as important as demand characteristics in determining the economic impacts of a rule. For relatively elastic supply curves, it is likely that cost increases or decreases will be passed on to consumers. The elasticity of supply

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18 Demand (or supply) is said to be "elastic" if the absolute value of the price elasticity of demand (supply) is greater than one and "inelastic" if the absolute value of the elasticity is less than one. If a percentage change in price leads to an equal percentage change in quantity demanded (supplied) (i.e., if the absolute value of elasticity equals one), demand (supply) is "unit elastic."

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

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

Similar to the determinants of demand elasticity, the factors influencing the price elasticity of supply all relate to a firm's degree of flexibility in adjusting production decisions in response to changing market conditions. The more easily a firm can adjust production levels, find input substitutes or adopt new production technologies, the more elastic is supply. Supply elasticities tend to increase over time as firms have more opportunities to renegotiate contracts and change production technologies. When production takes time, the quantity supplied may be more responsive to expected future price changes than to current price changes.

Demand and supply elasticities are available for the aggregate output of final goods in most industries. They are usually published in journal articles on research pertaining to a particular industry.[19] When such information is unavailable, as is often the case for intermediate goods, elasticities may be quantitatively or qualitatively assessed.[20] Econometric tools are frequently used to estimate supply and demand equations (thereby the elasticities) and the factors that influence them.

A.4.2 Measuring the Welfare Effect of a Change in Environmental Goods

As introduced in Section A.1, changes in consumer surplus are measured by the trapezoidal region below the ordinary, or Marshallian, demand curve as price changes. This region reflects the benefit a consumer receives by being able to consume more of a good at a lower price. If the price of a good decreases, some of the consumer's satisfaction comes from being able to consume more of a commodity when its price falls, but some of it comes from the fact that the lower price means that the consumer has more income to spend. However, the change in (Marshallian) consumer surplus only serves as a monetary measure of the welfare gain or loss experienced by the consumer under the strict assumption that the marginal utility of income is constant.[21] This assumption is almost never true in reality. Luckily, there are alternative, less demanding monetary measures of

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19 Another useful source of elasticity estimates is the recently developed EPA Elasticity Databank (U.S. EPA 2007d). In the absence of an encyclopedic, "Book of Elasticities," the Elasticity Databank, serves as a searchable database of elasticity parameters across a variety of types (i.e., demand and supply elasticities, substitution elasticities, income elasticities and trade elasticities) and economic sectors/product markets. The database is populated with EPA-generated estimates used in Environmental Impact Assessment studies conducted by the Agency since 1990, as well as estimates found in the economics literature. It can be accessed from the Technology Transfer Network Economics and Cost Analysis Support website: http://www.epa.gov/ttnecas1/Elasticity.htm.

20 Final goods are those that are available for direct use by consumers and are not utilized as inputs by firms in the process of production. Goods that contribute to the production of a final good are called intermediate goods. It is of course possible for a good to be final from one perspective and intermediate from another (Pearce 1992).

21 See Perman et al. (2003), Just et al. (2005) or any graduate level text for a more thorough exposition of this issue.

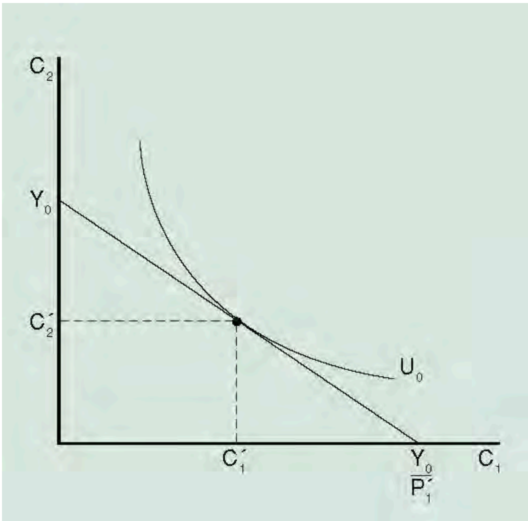
consumer welfare that prove useful in treatments of BCA. Intuitively, these measures determine the size of payment that would be necessary to compensate the consumer for the price change. In other words, they estimate the consumer's WTP for a price change.

As mentioned above, a price decline results in two effects on consumption. The change in relative prices will increase consumption of the cheaper good (the substitution effect), and consumption will be affected by the change in overall purchasing power (the income effect). A Marshallian demand curve reflects both substitution and income effects. Movements along it show how the quantity demanded changes as price changes (holding all other prices and income constant), so it reflects both the substitution and the income effects. The Hicksian (or "compensated") demand curve, on the other hand, shows the relationship between quantity demanded of a commodity and its price, holding all other prices and *utility* (rather than income) constant. This is the correct measure of a consumer's WTP for a price change. The Hicksian demand curve is constructed by adjusting income as the price changes so as to keep the consumer's utility the same at each point on the curve. In this way, the income effect of a price change is eliminated and the substitution effect can be considered alone. Movements along the Hicksian demand function can be used to determine the monetary change that would compensate the consumer for the price change.

Hicks (1941) developed two correct monetary measures of utility change associated with a price change: compensating variation and equivalent variation. *Compensating variation* (CV) assesses how much money must be taken away from consumers after a price decrease occurred to return them to the original utility level. It is equal to the amount of money that would "compensate" the consumer for the price decrease. *Equivalent variation* (EV) measures how much money would need to be given to the consumer to bring the consumer to the higher utility level instead of introducing the price change. In other words, it is the monetary change that would be 'equivalent' to the proposed price change.

Before examining the implications of these measures for valuing environmental changes, it is useful to understand CV and EV in the case of a reduction in the price of some normal, private good, C_1.[22] This is shown with indifference curves and a budget line, as seen in Figure A.7.

Figure A.7 - Indifference Curve

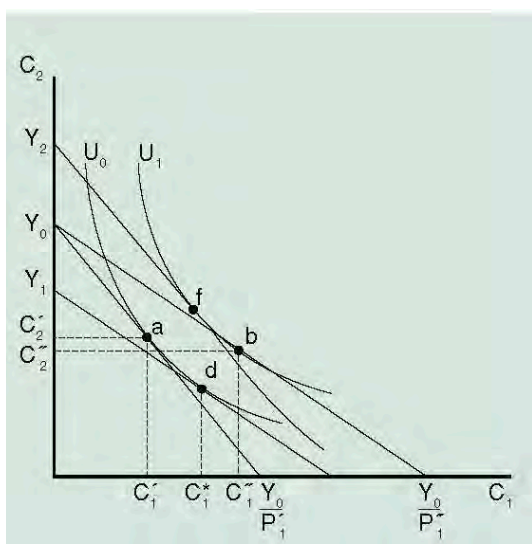


22 The notation and discussion in this section follow Chapter 12 of Perman et al. (2003).

Assume that the consumer is considering the trade-off between C_1 and all other goods, denoted by a composite good, C_2. The indifference curve, U_0, depicts the different combinations of the two goods that yield the same level of utility. Because of diminishing marginal utility, the curve is concave, where increasing amounts of C_1 must be offered for each unit of C_2 given up to keep the consumer indifferent. The budget line on the graph reflects what the consumer is able to purchase given her income, Y_0, and the prices of the two goods — P_1' and P_2', respectively.[23] A utility-maximizing consumer will choose quantities C_1' and C_2', the point where the indifference curve is tangent to the budget constraint.[24]

Figure A.8 shows the change in the optimal consumption bundle resulting from a reduction in the price of C_1. If the price of C_1 falls, the budget line shifts out on the C_1 axis because more C_1 can be purchased for a given amount of money. The consumer now chooses C_1'' and C_2'' at point *b* and moves to a new, higher utility curve, U_1. CV then measures how much money must be taken away at the new prices to return the consumer to the old utility level. That is, starting at point b and keeping the slope of the budget line fixed at the new level, by how much must it be shifted downward to make it tangent to the initial indifference curve, U_0? It is, therefore, the maximum amount the consumer would be willing to pay to have the price fall occur — i.e., the precise monetary measure of the welfare change.[25] In Figure A.8, CV is simply given by the amount $Y_0 - Y_1$. EV, on the other hand, measures how much income must be given to the individual at the old price set to maintain the same level of well-being as if the price change did occur. That is, keeping the slope of the budget line fixed at the old level, by how much must it be shifted upwards to make it tangent to U_1? EV is, then, the minimum amount of money the consumer would accept in lieu of the price fall. This too is a proper monetary measure of the utility change resulting from the price decrease. In Figure A.8 then EV is the amount $Y_2 - Y_0$, leaving the individual at point *f*.

Figure A.8 - Change in Optimal Consumption Bundle



23 In Figure A.7, C_2 is considered the numeraire good (i.e., prices are adjusted so that P_2' is equal to 1).

24 For a review of the utility maximizing behavior of consumers, see any general microeconomics textbook.

25 In Figure A.8, this would result in a shift from C_1'' to C_1^*. This is known as the income effect of the price change. The shift from C_1' to C_1^* is considered the substitution effect.

CV and EV are simply measures of the distance between the two indifference curves. However, the amount of money associated with CV, EV and Marshallian consumer surplus (MCS) is generally not the same. For a price fall, it can be shown that CV < MCS < EV, and for a price increase, CV > MCS > EV.[26] Notice that in the case of a price decrease, the CV measures the consumer's willingness to pay (WTP) to receive the price reduction and EV measures the consumer's willingness to accept (WTA) to forgo the lower price. If the price of C_1 were to increase, then the relationships between WTP/WTA and CV/EV would be reversed. CV would measure the consumer's WTA to suffer the price increase and EV would be the individual's WTP to avoid the increase in price.

In order to examine the implications of these measures for valuing changes in environmental conditions, one can think of C_1 in the above discussion as an environmental commodity, henceforth denoted by E. Then an improvement in environmental quality (or an increase in an environmental public good) resulting from some policy is reflected by an increase in the amount of E. Holding all else constant, such an increase is equivalent to a decrease in the price of E and can be depicted as a shifting outward of the budget line along the E axis.

Welfare changes due to an increase in E follow along the lines of the previous discussion. However, because E is generally non-exclusive and non-divisible, the consumer consumption level cannot be adjusted. Therefore, the associated monetary measures of the welfare change are not technically CV and EV but are referred to as *compensating surplus* (CS) and *equivalent surplus* (ES). In practice, however, the process is the same; a Hicksian demand curve is estimated for the unpriced environmental good. Analogous to the preceding discussion, if there is an environmental improvement, then CS measures the amount of money the consumer would be willing to pay for the improvement that would result in the pre-improvement level of utility. For the purposes of environmental valuation, this is the primary measure of concern when considering environmental improvements. ES measures how much society would have to pay the consumer to give the consumer the same utility as if the improvement had occurred. In other words, this is how much the consumer would be willing to accept to not experience the gain in environmental quality. If valuing an environmental degradation, then CS measures the WTA and ES measures WTP.

Whereas statements can be made about the relative size of CV, EV and MCS for price changes of normal goods, Bockstael and McConnell (1993) find that it is not possible to make similar statements about CS, ES and MCS for a change in environmental quality.[27] Given that environmental quality is generally an unpriced public good, ordinary Marshallian demand functions cannot be estimated, so it may seem irrelevant that one cannot say anything about how MCS approximates the proper measure. However, Bockstael and McConnell's results are important in relation to indirect methods for environmental valuation. However, most indirect valuation studies are based on Marshallian demand functions in practice, in the hope of keeping the associated error small.

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26 This can be seen by redrawing Figure A.8 using a graph of Marshallian and Hicksian demand curves. See Perman et al. (2003) for a detailed explanation.

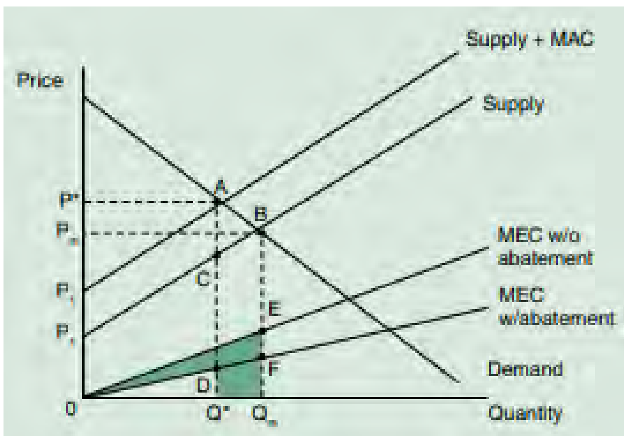
27 Willig (1976) shows that ordinary, or Marshallian, demand curves can provide an approximate measure of welfare changes resulting from a price change. In most cases the error associated with using MCS, with respect to CV or EV, will be less than 5% (see Perman et al. 2003).

A.4.3 Single Market, Multi-Market and General Equilibrium Analysis

Both supply and demand elasticities are affected by the availability of close complements and substitutes. This highlights the fact that regulating one industry can have an impact on other, non-regulated markets. However, this does not necessarily imply that all of these other markets must be modeled. Changes due to government regulation can be captured using only the equilibrium supply and demand curves for the affected market, assuming: (1) there are small, competitive adjustments in all other markets; and (2) there are no distortions in other markets. This is referred to as *partial equilibrium analysis*.

For example, suppose a new environmental regulation increases per unit production costs. The benefits and costs of abatement in a partial equilibrium setting are illustrated in Figure A.9 where the market produces the quantity Q_m in equilibrium without intervention. The external costs of production are shown by the marginal external costs (MEC) curve without 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.

Figure A.9 - Benefits and Costs of Abatement



With required abatement production, costs are the total of supply plus marginal abatement costs (MAC), shown as the new, higher supply curve in the figure. These higher costs result in a new market equilibrium quantity shown as Q^*. The social cost of the requirement is the resulting change in consumer and supplier surplus, shown here as the total observed abatement costs (parallelogram P_0P_1AC) plus the area of triangle ABC, which can be described as deadweight loss.

Abatement also produces benefits by shifting the MEC curve downward, reflecting the fact that each unit of production now results in less pollution and social costs. Additionally, the reduced quantity of the output good results in reduced external costs. The reduced external costs, i.e., the benefits, are given by the difference between triangle Q_mE0 and triangle Q^*D0, represented by the shaded area in the figure.

The net benefits of abatement are the benefits (the reduced external costs) minus the costs (the loss in consumer and producer surplus). In the figure, this would equal the shaded area (the benefits) minus total abatement costs and deadweight loss as described above.

While the single market analysis is theoretically possible, it is generally impractical for rulemaking. As mentioned in Section A.3, this is often because the gains occur outside of markets and cannot be linked directly to the output of the regulated market. Therefore, BCA is frequently done as two separate analyses: a benefits analysis and a cost analysis.

When a regulation is expected to have a large impact outside of the regulated market, then the analysis should be extended beyond that market. If the effects are significant but not anticipated to be widespread, one potential improvement is to use multi-market modeling in which vertically or horizontally integrated markets are incorporated into the analysis. The analysis begins with the relationship of input markets to output markets. A multi-market analysis extends the partial equilibrium analysis to measuring the losses in other related markets.[28]

In some cases, a regulation can have such a significant impact on the economy that a general equilibrium modeling framework is required.[29] This may be because regulation in one industry has broad indirect effects on other sectors, households may alter their consumption patterns when they encounter increases in the price of a regulated good, or there may be interaction effects between the new regulation and pre-existing distortions, such as taxes on labor. In these cases, partial equilibrium analyses are likely to result in an inaccurate estimation of total social costs. Using a general equilibrium framework accounts for linkages between all sectors of the economy and all feedback effects and can measure total costs comprehensively.[30]

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.

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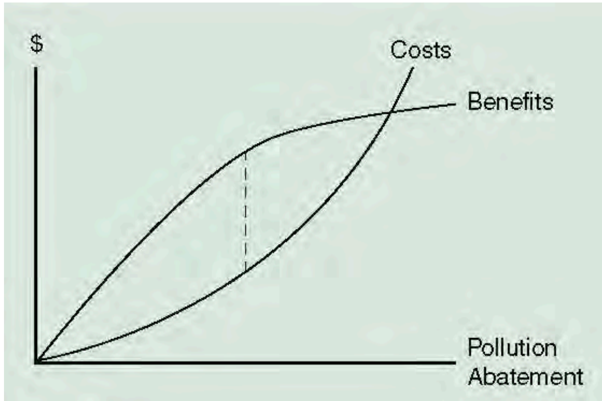
28 An example of the use of multi-market model for environmental policy analysis is contained in a report prepared for EPA on the regulatory impact of control on asbestos and asbestos products (U.S. EPA 1989).

29 General equilibrium analysis is built around the assumption that, for some discrete period of time, an economy can be characterized by a set of equilibrium conditions in which supply equals demand in all markets. When this equilibrium is "shocked" through a change in policy or a change in some exogenous variable, prices and quantities adjust until a new equilibrium is reached. The prices and quantities from the post-shock equilibrium can then be compared with their pre-shock values to determine the expected impacts of the policy or change in exogenous variables.

30 Chapter 8 provides a more detailed discussion of partial equilibrium, multi-market and general equilibrium analysis.

Figure A.10 - Maximized Net Benefits



Note that this is *not* necessarily the point at which:

- Benefits are maximized;
- Costs are minimized;
- Total benefits = total costs (i.e., benefit-cost ratio = 1);
- Benefit-cost ratio is the largest; or
- The policy is most cost-effective.

If the regulation were designed to maximize benefits, then any policy, no matter how expensive, would be justified if it produced any benefit, no matter how small. Similarly, minimizing costs would, in most cases, simply justify no action at all. A benefit-cost ratio equal to one is equivalent to saying that the benefits to society would be exactly offset by the cost of implementing the policy. This implies that society is indifferent between no regulation and being regulated; hence, there would be no net benefit from adopting the policy. Maximizing the benefit-cost ratio is not optimal either. Two policy options could yield equivalent benefit-cost ratios but have vastly different net benefits. For example, a policy that cost $100 million per year but produced $200 million in benefits has the same benefit-cost ratio as a policy that cost $100,000 but produced $200,000 in benefits, even though the first policy produces substantially more net benefit for society.[31] Finally, finding the most cost-effective policy has similar problems because the cost-effectiveness ratio can be seen as the inverse of the benefit-cost ratio. A policy is cost effective if it meets a given goal at least cost

31 Benefit-cost ratios are useful when choosing one or more policy options subject to a budget constraint. For example, consider a case where five options are available and the budget is $1,000. The first option will cost $1,000 and will deliver benefits of $2,000. Each of the other four will cost $250 and deliver benefits of $750. If options are selected according to the net benefits criterion, the first option will be selected, because its net benefits are $1,000 while the net benefits of each of the other options are $500. However, if options are selected by the benefit-cost ratio criterion, the other four options will be selected, as each of their benefit-cost ratios equal 3, versus a benefit-cost ratio of 2 for the first option. In this case, choosing options by the net benefits criterion will yield $1,000 in total net benefits, while choosing options by the benefit-cost ratio criterion will yield $500 in total net benefits. In most cases, choosing options in decreasing order of benefit-cost ratios will yield the largest possible net benefits given a fixed budget. This method will guarantee the optimal solution if the benefits and costs of each option are independent, and if each option can be infinitely subdivided, simply select the options in decreasing order of their benefit-cost ratios and once the budget is exceeded subdivide the last option selected such that the budget constraint is met exactly (see Dantzig 1957). Also note that this strategy does not require measuring benefits and costs in the same units, which means that it is directly useful for CEA (Hyman and Leibowitz 2000), while the net-benefit criterion is not.

— i.e., minimizes the cost per unit of benefit achieved. Cost-effectiveness analysis (CEA) can provide useful information to supplement existing BCA and may be appropriate to rank policy options when the benefits are fixed and cannot be monetized, but it provides no guidance in setting an environmental standard or goal.

Conceptually, net social benefits will be maximized if regulation is set such that emissions are reduced up to the point where the benefit of abating one more unit of pollution (i.e., marginal social benefit)[32] is equal to the cost of abating an additional unit (i.e., marginal abatement cost).[33] If the marginal benefits are greater than the marginal costs, then additional reductions in pollution will offer greater benefits than costs, and society will be better off. If the marginal benefits are less than marginal costs, then additional reductions in pollution will cost society more than they provide in benefits and will make society worse off. When the marginal cost of abatement is equal to society's marginal benefit, no gains can be made from changing the level of pollution reduction, and an efficient aggregate level of emissions is achieved. In other words, *a pollution reduction policy is at its optimal, most economically efficient point when the marginal benefits equal the marginal costs of the rule.*[34]

The condition that marginal benefits must equal marginal costs assumes that the initial pollution reduction produces the largest benefits for the lowest costs. As pollution reduction (i.e., regulatory stringency) is increased, the additional benefits decline and the additional costs rise. While not always true, a case can be made that the benefits of pollution reduction follow this behavior. The behavior of total abatement costs, however, will depend on how the pollution reduction is distributed among the polluters since firms may differ in their ability to reduce emissions. The aggregate marginal abatement cost function shows the least costly way of achieving reductions in emissions. It is equal to the horizontal sum of the marginal abatement cost curves for the individual polluters. Although each firm faces increasing costs of abatement, marginal cost functions still vary across sources. Some firms may abate pollution relatively cheaply, while others require great expense. To achieve economic efficiency, the lowest marginal cost of abatement must be achieved first, and then the next lowest. Pollution reduction is achieved at lowest cost only if firms are required to make equiproportionate

32 The benefits of pollution reduction are the reduced damages from being exposed to pollution. Therefore, the marginal social benefit of abatement is measured as the additional reduction in damages from abating one more unit of pollution.

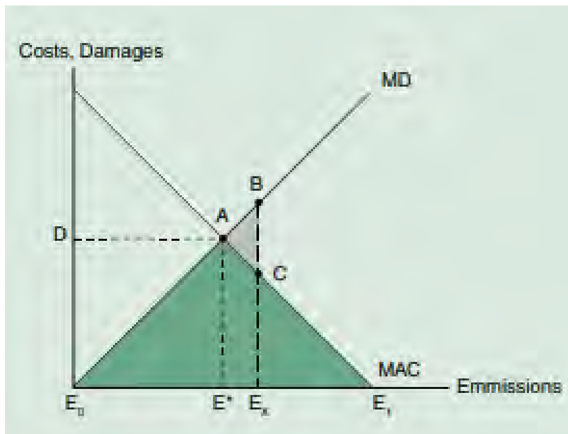
33 The idea that a given level of abatement is efficient — as opposed to abating until pollution is equal to zero — is based on the economic concept of diminishing returns. For each additional unit of abatement, marginal social benefits decrease while marginal social costs of that abatement increase. Thus, it only makes sense to continue to increase abatement until the point where marginal abatement benefits and marginal costs are just equal. Any abatement beyond that point will incur more additional costs than benefits. Alternatively, one can understand the efficient level of abatement as the amount of regulation that achieves the efficient level of pollution. If one considers a market for pollution, the socially-efficient outcome would be the point where the marginal WTP for pollution equals the marginal social cost of polluting.

34 It is important to reemphasize the word "marginal" in this statement. Marginal, in economic parlance, means the extra or next unit of the item being measured. If regulatory options could be ranked in order of regulatory stringency, then marginal benefits equal to marginal costs means that the additional benefits of increasing the regulation to the next degree of stringency is equal to the additional cost of that change.

cutbacks in emissions. At the optimal level of regulation, the cost of abating one more unit of pollution is equal across all polluters.[35]

Figure A.11 illustrates why the level of pollution that sets the marginal benefits and marginal costs of abatement equal to each other is efficient.[36] Emissions are drawn on the horizontal axis and increase from left to right. The damages from emissions are represented by the marginal damage (MD) curve. Damages may include the costs of worsened human health, reduced visibility, lower property values, and loss of crop yields or biodiversity. As emissions rise, the marginal damages increase. E_1 represents the amount of emissions in the absence of regulation on firms. The costs of controlling emissions are represented by the marginal abatement cost curve (MAC). As emissions are reduced below E_1, the marginal cost of abatement rises.

Figure A.11 - Efficient Level of Pollution



The total damages associated with emissions level E^* are represented by the area of the triangle AE_0E^*, while the total abatement costs are represented by area AE_1E^*. The total burden on society of this level is equal to the total abatement costs of reducing emissions from E_1 to E^* plus the total damages of the remaining emissions, E^*. That is, the total burden is the darkly shaded triangle, E_0AE_1.

Now assume that emissions are something other than E^*. For example, suppose emissions were E_X, which is greater than E^*. Total damages for this level of emissions are equal to the area of the triangle BE_0E_X, while total costs of abatement to this level is equal to the area CE_xE_1. The total burden on society of this level is the sum of the areas of the darkly shaded and the lightly shaded triangles. This means that the excess social cost of choosing emissions E_X rather than E^* is equal to the area of the lightly shaded triangle, *ABC*. A similar analysis could be done if emissions levels were below

35 Thus, a regulation that requires all firms to achieve the same level of reduction will probably result in different marginal costs for each firm and not be efficient (see Field and Field 2005 or any other environmental economics text for a detailed explanation and example).

36 Figure A.11 illustrates the simplest possible case, where the pollutant is a flow (i.e., it does not accumulate over time) and marginal damages are independent of location. When pollution levels and damages vary by location, then the efficient level of pollution is reached when marginal abatement costs adjusted by individual transfer coefficients are equal across all polluters. Temporal variability also implies an adjustment to this equilibrium condition. In the case of a stock pollutant, marginal abatement costs are equal across the discounted sum of damages from today's emissions in all future time periods. In the case of a flow pollutant, this condition should be adjusted to reflect seasonal or daily variations (see Sterner 2003).

level, *E**. Here, the additional abatement costs would be greater than the decrease in damages, resulting in excess social costs. The policy that sets the emissions level at *E** — at the point where marginal benefits of pollution reduction (represented by the MD curve) and the MAC curve intersect — is economically efficient because it imposes the least net cost on, and yields the highest net benefits for, society. That is, the triangle E_0AE_1 is the smallest shaded region that can be obtained.

This section has focused on first-best optimal regulation when there are no pre-existing market distortions. However, it is important to note that realizable policy outcomes will often be "second best" due to information constraints, political constraints, imperfect competition and market distortions created by tax and other government interventions. For example, many of the emissions-based policies emphasized in these *Guidelines* may be less feasible for addressing nonpoint source pollution, such as agriculture, which is less observable and more stochastic than emissions from point sources. Agriculture is also subject to multiple non-environmental policy distortions that must be considered in the measurement of the social benefits and costs of regulating agriculture.

A.6 Conclusion

The purpose of this appendix is to present a brief explanation of some of the fundamental economics relevant to Chapters 3 through 9. It is not intended to provide a comprehensive discussion of all microeconomic theory and its application to environmental issues. The interested reader can turn to undergraduate or graduate level textbooks for a more thorough exposition of the topics covered here. At the undergraduate level, Field and Field (2005) provide an introduction to the basic principles of environmental economics. Tietenberg's (2002) and Perman et al.'s (2003) presentations are more technical but still used primarily for undergraduate courses. Freeman (2003) is the standard text for graduate courses in environmental economics and deals with the methodology of non-market valuation. Supplemental texts that provide a good handle on environmental economics with less technical detail include Stavins (2000a), and Portney and Stavins (2000). Finally, general microeconomics textbooks (Mankiw 2004, and Varian 2005 at the undergraduate level; and Mas-Colell et al. 1995, Kreps 1990, and Varian 2005 at the graduate level), and applied welfare economics textbooks (Just et al. 2005) are useful references as well.

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Appendix B - Mortality Risk Valuation Estimates

Some U.S. Environmental Protection Agency (EPA) policies are designed to reduce the risk of contracting a potentially fatal health effect such as cancer. Reducing these risks of premature death provides welfare increases to those individuals affected by the policy. These policies generally provide marginal changes in relatively small risks. That is, these policies do not provide assurance that an individual will not die prematurely from environmental exposures; rather, they marginally reduce the probability of such an event. For BCA, analysts generally aggregate these small risks over the affected population to derive the number of statistical lives saved (or the number of statistical deaths avoided) and then use a "value of statistical life" (VSL) to express these benefits in monetary terms.

The risk reductions themselves can generally be classified according to the characteristics of the risk in question (e.g., voluntariness or controllability) and the characteristics of the affected population (e.g., age and health status). These dimensions may affect the *value* of reducing mortality risks. Ideally the VSL would account for all possible risk and demographic characteristics that matter. It would be derived from the preferences of the population affected by the policy, based on the type of risk that the policy is expected to reduce. For example, if a policy were designed to remove carcinogens at a suburban hazardous waste site, the ideal measure would represent the preferences for reduced cancer risks for the exposed population in the area and would reflect the changes in life expectancy that would result. Unfortunately, time and resource constraints make it difficult if not impossible to obtain such unique valuation estimates for each EPA policy. Instead, analysts need to draw from existing VSL estimates obtained using well-established methods (see Chapter 7).

This appendix describes the default VSL estimate currently used by the Agency and its derivation, as well as how analysts should characterize and assess benefit transfer issues that may arise in its application. Benefit transfer considerations that are common to all valuation applications, including the effect of most demographic characteristics of the study and policy populations, are described in Chapter 7 Section 7.3 and will not be repeated here.

B.1 Central Estimate of VSL

Table B.1 contains the VSL estimates that currently form the basis of the Agency's recommended central VSL estimate. Fitting a Weibull distribution to these estimates yields a central estimate (mean) of $10.7 million ($2022) with a standard deviation of $7.2 million.[1, 2] The EPA recommends that the central estimate, updated to the base year of the analysis, be used in all benefits analyses that seek to quantify mortality risk reduction benefits.

This approach was vetted and endorsed by the Agency when the 2000 *Guidelines for Preparing Economic Analyses* were drafted.[3] It remains the EPA's default guidance for valuing mortality risk changes although the Agency has considered and presented alternatives.[4]

B.2 Other VSL Information

For most of mortality risk reductions, the EPA uniformly applies the VSL estimate discussed above. For a period of time (2004-2008), the Office of Air and Radiation (OAR) valued mortality risk reductions using a VSL estimate derived from a limited analysis of some of the available studies. OAR arrived at a VSL using a range of $1 million to $10 million (2000$) consistent with two meta-analyses of the wage-risk literature. The $1 million value represented the lower end of the interquartile range from the Mrozek and Taylor (2002) meta-analysis of 33 studies. The $10 million value represented the upper end of the interquartile range from the Viscusi and Aldy (2003) meta-analysis of 43 studies. The mean estimate of $5.5 million (2000$) was also consistent with the mean VSL of $5.4 million estimated in the Kochi et al. (2006) meta-analysis. However, the Agency neither changed its official guidance on the use of VSL in rulemakings nor subjected the interim estimate to a scientific peer-review process through the Science Advisory Board (SAB) or other peer-review group.

During this time, the Agency continued work to update its guidance on valuing mortality risk reductions. The EPA commissioned a report from meta-analytic experts to evaluate methodological questions raised by the EPA and the SAB on combining estimates from the various data sources. In

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1 The VSL was updated from the $4.8 million ($1990) estimate referenced in the 2000 Guidelines by adjusting the individual study estimates for inflation using CPI-U and then fitting a Weibull distribution to the estimates. The updated Weibull parameters are: location = 0, scale = 11.91, shape = 1.51 (updated from location = 0; scale = 5.32; shape = 1.51). The Weibull distribution was determined to provide the best fit for this set of estimates. See U.S. EPA 1997a for more details.

2 This VSL estimate was produced using the Consumer Price Index (CPI). Some economists prefer using the GDP deflator inflation index in some applications. The key issue for EPA analysts is to ensure that the chosen index is used consistently throughout the analysis.

3 The studies listed in Table B.1 were published between 1974 and 1991, and most are hedonic wage estimates that may be subject to considerable measurement error (Black et al. 2003; Black and Kniesner 2003). Although these were the best available data at the time, they are sufficiently dated and may rely on obsolete preferences for risk and income. The Agency is currently considering more recent studies as it evaluates approaches to revise its guidance.

4 The EPA engaged the SAB-EEAC on several issues including the use of meta-analysis as a means of combining estimates and approaches for assessing mortality benefits when changes in longevity may vary widely (U.S. EPA 2006d; U.S. EPA 2016). see U.S. EPA 2017 for recent SAB recommendations.

addition, the Agency consulted several times with the SAB Environmental Economics Advisory Committee (SAB-EEAC) on the issue (e.g., U.S. EPA 2017).

Table B.1 - Value of Statistical Life Estimates (mean values in millions of 2022 dollars)

Study	Method	Value of Statistical Life
Kniesner and Leeth (1991 - US)	Labor Market	$1.34
Smith and Gilbert (1984)	Labor Market	$1.57
Dillingham (1985)	Labor Market	$2.02
Butler (1983)	Labor Market	$2.46
Miller and Guria (1991)	Contingent Valuation	$2.69
Moore and Viscusi (1988)	Labor Market	$5.60
Viscusi, Magat, and Huber (1991)	Contingent Valuation	$6.05
Marin and Psacharopoulos (1982)	Labor Market	$7.39
Gegax et al. (1985)	Contingent Valuation	$6.27
Kniesner and Leeth (1991 - Australia)	Labor Market	$7.39
Gerking, de Haan, and Schulze (1988)	Contingent Valuation	$7.61
Cousineau, Lecroix, and Girard (1988)	Labor Market	$8.06
Jones-Lee (1989)	Contingent Valuation	$8.51
Dillingham (1985)	Labor Market	$8.73
Viscusi (1978)	Labor Market	$9.18
R.S. Smith (1976)	Labor Market	$10.30
V.K. Smith (1983)	Labor Market	$10.52
Olson (1981)	Labor Market	$11.64
Viscusi (1981)	Labor Market	$14.55
R.S. Smith (1974)	Labor Market	$16.12
Moore and Viscusi (1988)	Labor Market	$16.35
Kniesner and Leeth (1991 - Japan)	Labor Market	$17.02
Herzog and Schlottman (1987)	Labor Market	$20.38
Leigh and Folsom (1984)	Labor Market	$21.72
Leigh (1987)	Labor Market	$23.29
Garen (1988)	Labor Market	$30.23

Derived from U.S. EPA (1997a) and Viscusi (1992). Updated to 2022$ with CPI-U.

Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently best reflects the SAB-EEAC advice received to date. Therefore, the VSL described above that was vetted and endorsed by the SAB should be applied in relevant analyses while the Agency continues its efforts to update its guidance on this issue.

B.3 Benefit Transfer Considerations

Policy analysts valuing mortality risk reductions should account for differences in risk and population characteristics between the policy and study scenarios and their potential effect on the overall results. The ultimate objective of the benefit transfer exercise is to account for all of the factors that significantly affect the value of mortality risk reduction in the context of the policy. Analysts should carefully consider the implications of correcting for some relevant factors, but not for others, recognizing that it may not be feasible to account for all factors.

B.4 Adjustments Associated with Risk Characteristics

Risk characteristics appear to affect the value that people place on risk reduction. A large body of work identifies eight dimensions of risk that affect human risk perception.[5]

1. Voluntary/involuntary
2. Ordinary/catastrophic
3. Delayed/immediate
4. Natural/man-made
5. Old/new
6. Controllable/uncontrollable
7. Necessary/unnecessary
8. Occasional/continuous

Transferring VSL estimates among these categories may introduce bias. There have been some recent efforts attempting to quantitatively assess these sources of bias.[6] These studies generally conclude that voluntariness, control and responsibility affect individual values for safety, although there is no consensus on the direction and magnitude of these effects.

In addition, environmental risks may differ from those that form the basis of VSL estimates in many of these dimensions. Occupational risks, for example, are generally considered to be more voluntary in nature than are environmental risks and may be more controllable. As part of the Agency's review of our mortality risk guidance we are evaluating the literature from which the studies are drawn.

Support for quantitative adjustments in the empirical literature is lacking for most of these factors. The SAB reviewed an Agency summary of the available empirical literature on the effects of risk and population characteristics on WTP for mortality risk reductions (U.S. EPA 2000d). The SAB review concludes that among the demographic and risk factors that might affect VSL estimates, the

5 A review of issues in risk perception is found in Lichtenstein and Slovic (2006). Other informative sources include Slovic (1987), Rowe (1977), Otway (1977), and Fischoff et al. (1978).

6 Examples include Hammitt and Liu (2004), Sunstein (1997), Mendeloff and Kaplan (1990), McDaniels et al. (1992), Savage (1993), Jones-Lee and Loomes (1994, 1995, 1996), and Covey et al. (1995).

current literature can only support empirical adjustments related to the timing of the risk. The review supports making the following adjustments to primary benefits estimates: (1) adjusting WTP estimates to account for higher future income levels, though not for cross-sectional differences in income; and (2) discounting risk reductions that are brought about in the future by current policy initiatives (that is, after a cessation lag), using the same rates used to discount other future benefits and costs. All other adjustments, if made, should be relegated to sensitivity analyses.

Increases in income over time. The economics literature shows that the income elasticity of WTP to reduce mortality risk is positive, based on cross-sectional data. As a result, benefits estimates of reduced mortality risk accruing in future years may be adjusted to reflect anticipated income growth, using the range of income elasticities (0.08, 0.40 and 1.0) employed in *The Benefits and Costs of the Clean Air Act, 1990-2010.*[7] Recent EPA analyses have assumed a triangular distribution from these values and used the results in a probabilistic assessment of benefits.[8] At the time of this writing, the EPA is engaged in a consultation with the SAB-EEAC on the appropriate range of income elasticities and will update this guidance as needed.

Timing of reduced exposure and reduced risk. Many environmental policies are targeted at reducing the risk of effects such as cancer, where there may be an extended period of time between the reduced exposure and the reduction in the risk of death from the disease.[9] This delay between the change in exposure and realization of the reduced risk may affect the value of that risk reduction. Most existing VSL estimates are based on risks of relatively immediate fatalities making them an imperfect fit for a benefits analysis of many environmental policies. Economic theory suggests that reducing the risk of a delayed health effect will be valued less than reducing the risk of a more immediate one, when controlling for other factors.

B.5 Effects on WTP Associated with Demographic Characteristics

Two population characteristics are particularly noteworthy for their potential effect on mortality risk valuation estimates: age and health status of the exposed population. In September 2006, the Agency requested an additional advisory from the SAB-EEAC on issues related to valuing changes in life expectancy for which age and baseline health status are close correlates.[10] Because the outcome of this review is not yet available, we focus here on previous advice received from the SAB on related questions.

Age. It has sometimes been posited that older individuals should have a lower WTP for changes in mortality risk given the fewer years of life expectancy remaining compared to younger individuals. This hypothesis may be confounded, however, by the finding that older persons reveal a greater

7 For details see Kleckner and Neuman (2000).

8 See, for example, pp. 6-84 of the Final Economic Analysis for the Stage 2 Disinfection Byproducts Rule (DBPR) (U.S. EPA 2005d).

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

10 U.S. EPA (2006d) summarizes much of the literature related to the effects of age and health status on WTP for changes in mortality risk and includes the charge questions put to the SAB-EEAC on these issues.

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.[11] In general, this literature has shown that the relationship between age and WTP for mortality risk changes is ambiguous, requiring strong assumptions to even sign the relationship.[12] Empirical evidence is also mixed. A number of empirical studies (mostly hedonic wage studies) suggest that the VSL follows a consistent "inverted-U" life-cycle, peaking in the region of mean age.[13] Others find no such statistically significant relationship and still others show WTP increasing with age.[14] Stated preference results are also mixed, with some studies showing declining WTP for older age groups and others finding no statistically significant relationship between age and WTP.[15]

In spite of the ambiguous relationship between age and WTP, two alternative adjustment techniques have been derived from this literature. The first technique, *value of statistical life-years (VSLY)*, is derived by dividing the estimated VSL by expected remaining life expectancy. This is by far the most common approach and presumes that: (1) the VSL equals the sum of discounted values for each life year; and (2) each life year has the same value. This method was applied as an alternative case in an effort to evaluate the sensitivity of the benefits estimates prepared for the EPA's retrospective and prospective studies of the costs and benefits of the Clean Air Act (U.S. EPA 1997a; U.S. EPA 1999).

A second technique is to apply a distinct value or suite of values for mortality risk reduction depending on the age of incidence. However, there is relatively little available literature upon which to base such adjustments.[16]

Neither approach enjoys general acceptance in the literature as they both require large assumptions to be made, some of which have been contradicted in empirical studies. Since published support is lacking, neither approach is recommended at this time.

Analysts are advised to note the age distribution of the affected population when possible, especially when children are found to be a significant portion of the affected population.[17] Although

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11 See, for example, Shepard and Zeckhauser (1982), Rosen (1988), Cropper and Sussman (1988, 1990), and Johansson (2002).

12 See Evans and Smith (2006) for a recent summary.

13 See Jones-Lee et al. (1985), Aldy and Viscusi (2008), Viscusi and Aldy (2007a, 2007b), and Kniesner et al. (2006).

14 Viscusi and Aldy (2003) review more than 60 studies of mortality risk estimates from 10 countries and discuss eight hedonic wage studies that explicitly examine the age-WTP relationship. Only five of the eight studies found a statistically significant, negative relationship between age and the return to risk. Smith et al. (2004) and Kniesner et al. (2006) find that WTP increases with age.

15 Krupnick et al. (2002) report that WTP for mortality risk reductions changes significantly with age after age 70. Alberini et al. (2004) find no difference in the WTP for younger age groups and find a 20% reduction for those aged 70 and older. However, this difference was not statistically significant.

16 This second approach was illustrated in one EPA study (U.S. EPA 2002) for valuation of air pollution mortality risks, drawing upon adjustments measured in Jones-Lee et al. (1985).

17 See U.S. EPA (2003a) for more information on the valuation of children's health risks. OMB's Circular A-4 advises agencies to use estimates of mortality risk valuation for children that are at least as large as those used for adult populations (OMB 2003).

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).[18]

Application of existing VSLY approaches implicitly assumes a linear relationship in which each discounted life year is valued equally. As Office of Management and Budget (OMB) (1996) notes "current research does not provide a definitive way of developing estimates of VSLY that are sensitive to such factors as current age, latency of effect, life years remaining, and social valuation of different risk reductions." The second alternative, applying a suite of values for these risks, lacks broad empirical support in the economics literature. However, the potential importance of this benefit transfer factor suggests that analysts consider sensitivity analysis when risk data — essentially risk estimates for specific age groups — are available. An emerging literature on the value of life expectancy extensions, based primarily on stated preference techniques, is beginning to help establish a basis for valuation in cases where the mortality risk reduction involves relatively short extensions of life.[19]

B.6 Conclusion

Due to current limitations in the existing economic literature, these *Guidelines* conclude that, for the present time, the appropriate default approach for valuing these benefits is provided by the central VSL estimate described earlier. However, analysts should carefully present the limitations of this estimate. Economic analyses should also fully characterize the nature of the risk and populations affected by the policy action and should confirm that these parameters are within the scope of the situations considered in these *Guidelines*. While a qualitative discussion of these issues is generally warranted in EPA economic analyses, analysts should also consider a variety of quantitative sensitivity analyses on a case-by-case basis as data allow. The analytical goal is to characterize the impact of key attributes that differ between the policy and study cases. These attributes, and the

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

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

degree to which they affect the value of risk reduction, may vary with each benefit transfer exercise, but analysts should consider the characteristics described above (e.g., age, health status, voluntariness of risk and latency) and values arising from altruism.

As the economic literature in this area evolves, WTP estimates for mortality risk reductions that more closely resemble those from environmental hazards may support more precise benefit transfers. Literature on the specific methods available to account for individual benefit-transfer considerations will also continue to develop. In addition, The EPA will continue to conduct periodic reviews of the risk valuation literature and will reconsider and revise the recommendations in these *Guidelines* accordingly. The EPA will seek advice from the SAB as guidance recommendations are revised.

Appendix B References

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Glossary

Abatement costs - Abatement costs are costs borne by firms when they are required to remove and/or reduce environmental byproducts created during production.

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

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

Benefit-cost analysis (BCA) - A BCA is an evaluation of the social benefits and social costs of a policy action. The social benefits of a policy are measured by society's willingness-to-pay for the policy outcome. The social costs are measured by the opportunity costs of adopting the policy. BCA addresses the question of whether the benefits for those who gain from the action are sufficient to, in principle, compensate those burdened by costs such that everyone would be at least as well off as before the policy. The calculation of net benefits (benefits minus costs) answers this question and helps ascertain the economic efficiency of the policy. Where all benefits and costs can be quantified and expressed in monetary units, BCA provides decision makers with a clear indication of the most economically efficient alternative, that is, the alternative that generates the largest net benefits to society (ignoring distributional effects).

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

Benefit transfer - Benefit transfer is the use of estimated values of environmental quality changes drawn from primary (usually published) studies for the evaluation of similar changes of interest to the analyst.

Cessation lag - Cessation lag is the time between a reduction in 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 is a prescriptive regulation that stipulates how much pollution an individual source or plant is allowed to emit and/or what types of control equipment it must use to reduce pollution.

Compliance cost - A compliance cost is the private cost that a regulated entity incurs to comply with a regulation — for instance, through the planning, design, installation, and operation of pollution abatement equipment.

Consumption rate of interest - Consumption rate of interest is the rate at which individuals are willing to exchange consumption in one period (usually year) for consumption in the next period. This rate reflects the individual's rate of time preference.

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

Distributional analysis - Distributional analysis assesses changes in social welfare by examining the effects of a regulation across different subpopulations and entities.

Distorted market - A distorted market is one that does not experience free and open competition due to government interventions and/or prevailing market conditions. Examples of distortions include externalities, regulations, pre-existing taxes, or imperfectly competitive markets.

Dollar year - The year to which the purchasing power of a dollar is indexed. For example, if costs and benefits are reported in 2016 dollars, it means that the purchasing power of those costs and benefits reflect what could have been bought in 2016.

Ecological production function - An ecological production function is a description of how ecosystems combine inputs to produce ecosystem services that consumers enjoy directly or are used in the production of goods or services that are enjoyed by consumers.

Economic efficiency - Economic efficiency can be defined as the maximization of social welfare. Under the efficient level of production, there is no way to rearrange production or reallocate goods such that someone is better off without making someone else worse off in the process.

Economic impact analysis (EIA) - Economic impact analyses (EIAs) examine how compliance costs, transfers, and other policy outcomes are distributed across groups. EIAs describe and often quantify outcomes such as changes in employment, plant closures or local government tax revenues that provide insight into the economic consequences of regulation.

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

Environmental justice - Environmental justice is the fair treatment and meaningful involvement of all people regardless of race, color, national origin or income with respect to the development, implementation and enforcement of environmental laws, regulations and policies. Fair treatment means that no group of people, including racial, ethnic or socioeconomic groups, should bear a disproportionate share of the negative environmental consequences resulting from industrial, government and commercial operations or policies. Meaningful involvement occurs when (1) potentially affected community members have an appropriate opportunity to participate in decisions about a proposed activity 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]

Expected value - Expected value is the probabilistically weighted outcome that defines a statistical mean and a measure of the central tendency of a set of data. For a variable with a discrete number of outcomes, the expected value is calculated by multiplying each of the possible outcomes by the likelihood that each outcome will occur and then summing all of those values.

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1 Definition taken from http://www.epa.gov/compliance/environmentaljustice/index.html (accessed December 22, 2020)

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.

Externality - An externality occurs when the actions of one individual (or firm) have a direct, unintentional and uncompensated effect on the well-being of other individuals or the profits of other firms.

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

General equilibrium - A general equilibrium modeling approach concurrently considers the effect of a regulation across all sectors in the economy.

Hedonic price - Hedonic price, recreational demand or locational choice models may be regarded as "reduced form" representations of ecological production from which the analyst can infer the values individuals ascribe to ecosystem services by observing the choices they make, provided that the analyst can adequately control for potentially confounding factors.

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

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

Latency - Latency is the time between the increase in exposure to a pollutant and the increase in health risk. See *cessation lag* for a definition of a related but distinct concept.

Marginal benefit - The marginal benefit is the benefit received from an incremental increase in the consumption of a good or service.

Marginal cost - The marginal cost is the change in total cost that results from a unit increase in output.

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

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

Market failure - A market failure occurs when the allocation of goods and services by the free market is not economically efficient. The most common causes of market failure are externalities, market power and inadequate or asymmetric information. Externalities are the most likely cause of market failure in an environmental context.

Market-based approaches - Market-based approaches to environmental policy include instruments such as taxes, fees, charges and subsidies. These approaches create a price incentive to reduce pollution and leave decisions about the level of emissions to each source. Another example is an allowance trading system, which sets the total quantity of emissions and then allows trading of permits among firms.

Meta-analysis - Meta-analysis is an umbrella term for a suite of techniques that synthesize the results of empirical research. This could include a simple ranking o results, a meta-analytic average or other central tendency estimate, or a multivariate regression.

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

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

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

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

Nudge - A nudge is a structural or design feature of an individual choice that alters people's behavior in a predictable way without precluding any options or significantly changing their economic incentives but that can still be easily avoided.

Operating costs - Operating costs are recurring expenditures associated with the operation and maintenance of equipment, including salaries and wages, energy inputs, materials and supplies, purchased services and maintenance or repair of equipment associated with pollution abatement or waste management.

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

Pareto efficiency or **Pareto optimality** - Pareto efficiency is an economic state in which it is impossible to reallocate resources to make one individual better off without making another worse off.

Partial equilibrium - A partial equilibrium modeling approach accounts for market changes in the regulated sector. Market responses to the regulation may include reduced industry output or higher prices as firms pass on some costs directly to consumers. The goal of a partial equilibrium approach is to measure the net change in consumer and producer surplus relative to the pre-regulatory equilibrium.

Performance-based standard - A performance-based standard is a pollution control standard that requires polluters to meet a source-level emission standard without mandating the specific method by which they must comply with the standard. A performance-based standard is defined in terms of an emission *level* or an emission *rate* (i.e., emissions per unit of output or input).

Prescriptive regulation - A prescriptive regulation is a policy that stipulates how much pollution an individual source or plant is allowed to emit and/or what types of control equipment or approaches it must use to reduce pollution. Prescriptive regulations are also known as "direct regulatory instruments" or "command-and-control" regulations.

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

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

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

Rebound effect - A rebound effect is the reduction in expected gains from improvements in the energy efficiency of a technology that results from changes in consumer behavior. For example, tighter vehicle fuel economy standards lead to rebound effects because these regulations make it cheaper to consume energy or fuel on a per-unit basis causing demand for these services and therefore emissions from them to increase.

Shadow price of capital - The shadow price of capital accounts for 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 benefits - Social benefits are the sum of all positive changes in societal well-being experienced as a result of the regulation or policy action. Economists define benefits by focusing on changes in individual well-being, referred to as welfare or utility. Willingness to pay (WTP) is the preferred measure of these changes as it theoretically provides a full accounting of individual preferences across trade-offs between income and the favorable effects.

Social cost - Social cost means the sum of all opportunity costs, or reductions in societal well-being, incurred as a result of the regulation or policy action. These opportunity costs consist of the value lost to society of all the goods and services that will not be produced and consumed as regulated entities reallocate resources in order to comply with the regulation. To be complete, an estimate of social cost should include both the opportunity costs of current consumption that will be foregone because of the regulation, and also the losses that may result if the regulation reduces capital investment and thus future consumption.

Social opportunity cost of capital - Social opportunity cost of capital is the rate at which consumption in the next period is reduced because private investment is displaced by required investments from policy. This is the rate at which society can trade consumption over time due to productive capital.

Social rate of time preference - Social rate of time preference is the discount rate at which society is willing to trade consumption in one period (usually year) for consumption in the next period.

Social welfare function - A social welfare function establishes criteria under which efficiency and equity outcomes are transformed into a single metric, making them directly comparable. A potential output of such a function is a ranking of policy outcomes that have different aggregate levels and distributions of net benefits. A social welfare function can provide empirical evidence that a policy alternative yielding higher net benefits, but a less equitable distribution of wealth, ranks better or worse than a less efficient alternative with more egalitarian distributional consequences.

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

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

Technology standard - A pollution control standard that mandates the use of a specific control technology or production process by individual polluters.

Transaction costs - Transactions costs are the costs incurred when buying or selling a good or service. They may include the costs of searching out a buyer or seller, bargaining and enforcing contracts.

Transfers - Transfers are shifts in money or resources from one part of the economy (e.g., a group of individuals, firms, or institutions) to another in a way that does not affect the total resources that are available to society.

Use value - Use value is the value that an individual may derive from consumption or use of a good or resource.

Value of statistical life (VSL) - The VSL is the marginal rate of substitution (MRS) between mortality risk and money, i.e., the willingness-to-pay (WTP) for small reduction in the risk of premature mortality.

Value of statistical life year (VSLY) - The VSLY is an estimated dollar value for a year of statistical life. In practice this metric is typically derived by dividing a VSL estimate by remaining life expectancy or discounted remaining life expectancy. This approach usually assumes that each year of life over the life cycle has the same value.

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

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