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

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.

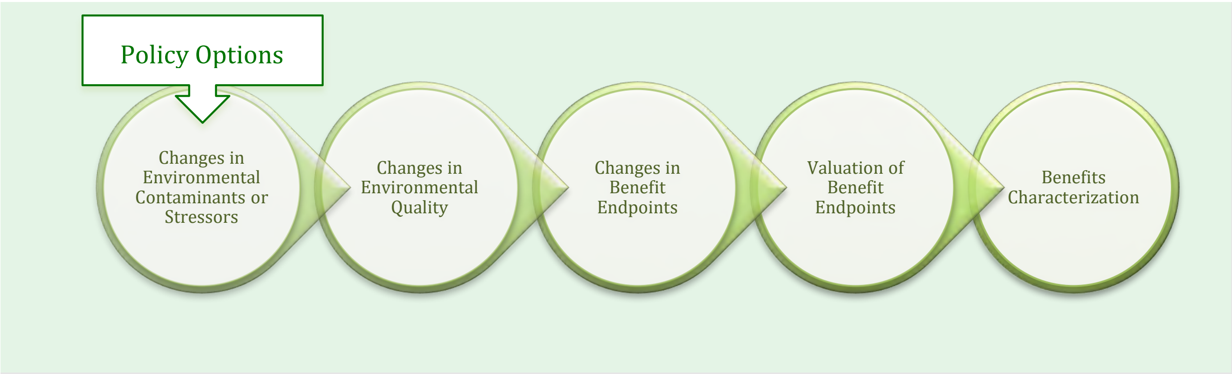
2 While this chapter focuses on the anticipated social benefits of regulation, the same approach also applies in a retrospective setting. See Chapter 5, Text Box 5.1, for more discussion of retrospective analysis.

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

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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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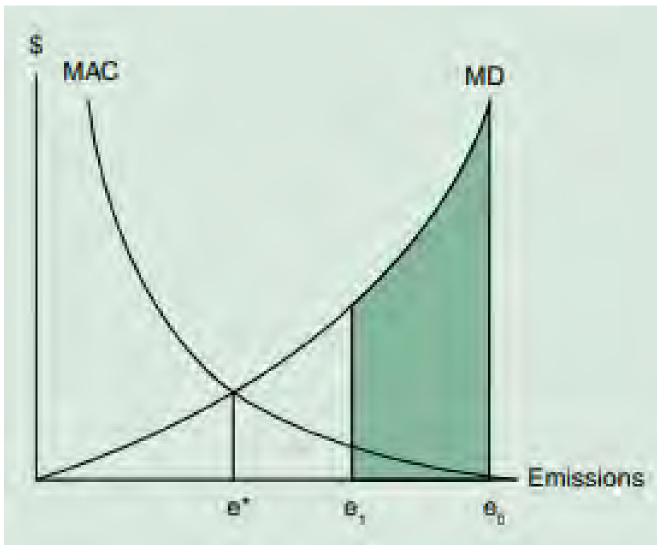
13 Because an individual's utility is unobservable, it cannot be measured in a cardinal sense. However, economists assume that consumers make choices based on whether one bundle of goods is preferred to another, but not necessarily by how much. In other words, consumers respond to changes in prices and 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

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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.

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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

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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).

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

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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.

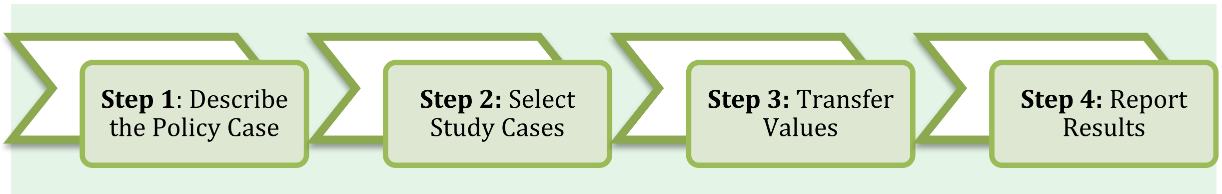
Benefit transfer is necessary when it is infeasible to conduct an original study focused directly on the policy case and is the most common approach for completing a BCA at the EPA. Given the time and analytical resource constraints under which most regulatory analysis activities are conducted, conducting 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

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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