METHODS FOR BENEFITS ANALYSES — CHAPTER 3

Regulations establishing Maximum Contaminant Levels (MCLs) or treatment requirements under the Safe Drinking Water Act (SDWA) may have several types of benefits. On a national level, the most significant benefits generally will be improvements in human health. As described in Chapter 1, other benefits may include aesthetic effects (such as improved taste or odor) and effects on materials (such as reduced pipe corrosion). Regulations that lead to greater source water protection may also have ecological benefits, such as increased protection of biodiversity.

As discussed in Chapter 2, analysis of these types of effects is necessary to meet the SDWA requirements for assessing the extent to which the benefits of achieving the lowest feasible MCL may be commensurate with the costs. Benefit-cost analysis is also necessary for all major rulemakings under government-wide and EPA requirements. These analyses also address the impact of regulations on certain groups of concern (including state and local governmental units, private entities, minorities, low income groups, and children), as required by SDWA and several statutes and executive orders.

The practice of benefits assessment is based on the discipline of welfare economics. In this chapter, we briefly introduce the theoretical foundation and economic methods for benefits analysis, then describe "best practices" for assessing particular types of benefits.[1] Although it is generally useful to express the value of benefits in dollar terms using the methods discussed below, analysts may often find that it is not possible to quantify or value all of the benefits of drinking water regulations. In such cases, nonquantified and nonmonetized benefits are carefully described in the analysis so that they can be taken into consideration by decision-makers.

This chapter is divided into three parts. First, we introduce the economic concepts that provide the foundation for benefits analyses. Next, we describe research methods commonly used to determine the dollar value of these benefits. Finally, we describe approaches for assessing specific types of benefits in more detail. Chapter 4 discusses the transfer of benefit estimates from existing studies to the analysis of drinking water regulations, while Chapter 5 provides information and examples related to implementing these methods.

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[1] For a more detailed technical discussion of the economic theory and methods for benefits assessment described in this chapter, see: U.S. Environmental Protection Agency, *Guidelines for Preparing Economic Analyses:* EPA 240-R-00-003, September 2000; and, Freeman, A. Myrick III, *The Measurement of Environmental and Resource Values: Theory and Methods,* Resources for the Future, Washington, D.C., 1993.

3.1 The Economist's Perspective

The simplest way to value benefits from a drinking water regulation would be to use market data or survey responses to determine directly the value of decreasing contaminant concentrations. For example, if a "decrease in arsenic contaminant concentrations from 50 μg/L to 10 μg/L" was a product available for purchase, we could use market data on the demand for this product to value benefits. Alternatively, we could survey consumers and ask how much they would be willing to pay to reduce contaminant concentrations by specific amounts.

Unfortunately, we often cannot determine the value of benefits in such a straightforward manner. Because reductions in contaminant concentrations are not goods that are directly bought and sold, there is little empirical information on the prices people would be willing to pay for these reductions. In addition, people who are not familiar with the effects of individual drinking water contaminants may have difficulty responding to a survey asking them what they would be willing to pay for reduced concentrations; conducting a survey that fully informs them about each contaminant can be quite expensive and time consuming.

Faced with these difficulties, benefit analysts usually begin by listing the possible effects reduced by the regulations, then focus on valuing each specific effect (such as the changes in the risks of contracting a particular disease). Values are derived for each effect, then aggregated (taking care to avoid double-counting) to determine the total benefits of the regulations. For example, rather than directly estimating the value of a specific reduction in the concentrations of a chemical (such as arsenic or benzene), analysts generally estimate the value of the risks averted (such as the risks of incurring certain nervous system disorders or kidney cancer) and other benefits (such as improved taste or odor), then aggregate the values of these effects to determine the total benefits of the rule.

To determine the monetary value of these benefits, economists focus on what people would be willing to pay for specific health improvements and other effects of the regulations. The basis for this focus on willingness to pay, and its advantages and limitations, are described below.

3.1.1 Willingness to Pay

In considering policies that affect social welfare, economists begin with the assumption that individuals derive utility (or a sense of satisfaction or well-being) from the goods and services they consume. Conversely, people may derive disutility from negative experiences, such as illness or harm to the environment. Individuals can maintain the same level of utility while trading off different bundles of goods and services (e.g., one may be equally happy going to the movies or a baseball game), and their willingness to make these trade-offs can be measured in dollar terms.

In theory, the dollar value of the benefits associated with a regulatory requirement is most appropriately measured by determining the change in income that has the same effect on utility (or the level of individual satisfaction) as the requirement. Because utility is impossible to measure directly, economists rely instead on estimates of willingness to pay or willingness to accept compensation to value the effects of regulations and other actions that lead to improvements in environmental quality. Willingness to pay is the maximum amount of money an individual would voluntarily exchange to obtain an improvement (e.g., in drinking water quality), given his or her budget constraints. Willingness to accept is the least amount of money an individual would accept to forego the improvement.

These two measures are not necessarily equal. One reason for the difference is that the two measures have different starting points. For environmental improvements, willingness to pay uses the level of utility *without* the improvement as a reference point, while willingness to accept uses as its reference point the level of utility *with* the improvement. Under conventional assumptions, economists expect that the difference between these measures will be small in many cases; e.g., as long as the amount involved is not a significant proportion of income.[2] In practice, benefits analysts usually rely on measures of willingness to pay because of concerns about the accuracy and reliability of the methods available for estimating willingness to accept compensation.[3] Willingness to pay is generally easier to measure and quantify.

While willingness to pay is constrained by income, it is a different concept than affordability. "Affordability" is a nontechnical term that is often used to refer to peoples' judgements about what is "reasonable" to pay for a particular good or service. In contrast, willingness to pay is the maximum amount an individual would

[2] In the case of environmental goods, additional considerations may lead to larger differences between willingness to pay and willingness to accept, as discussed in EPA's *Guidelines for Preparing Economic Analyses* and Hanemann, W.M., "Willingness to Pay and Willingness to Accept: How Much Can They Differ?," *American Economic Review*, Volume 81, Number 3, 1991, pp. 635- 647.

[3] Accuracy refers to whether the findings are correct; for example, to how well the study results mirror the value in the underlying population. Reliability refers to whether the findings can be replicated; for example, to whether applying a survey to a second sample would result in the same or similar estimates as those from the first sample.

actually pay for a good or service, given his or her income constraints and other desired expenditures.[4]

Willingness to pay is also a different concept than cost or price. "Cost" refers to the resources needed to produce a good or service; it does not measure the value of the good or service to members of society. "Price" is determined by the interactions of suppliers and consumers in the marketplace. For some individuals, the market price may exceed willingness to pay, in which case they will not purchase the good. For other individuals, willingness to pay may exceed the current price, in which case these individuals will benefit from the fact that the market price is less than he or she is willing to pay.

Economists refer to the aggregate amount that individuals are willing to spend on a good or service over and above that required by the market price as "consumer surplus." Changes in this surplus can be used to measure the benefits of various policy options. For example, if a government program reduces the price of a good or service, consumers are likely to purchase more of the product. For some consumers, the price drop will cause the difference between price and willingness to pay to rise. These impacts will increase consumer surplus, and the dollar amount of the increase can be used to measure the social welfare benefits of the policy.[5]

Measuring the value of benefits in dollar terms, based on estimates of willingness to pay, provides useful information for decision-makers. First, it is easier to compare costs and benefits and make related decisions when both are expressed in monetary terms. Second, valuation (accompanied by discussion of uncertainties in the estimates used) provides explicit, objective information on the amount of money members of society would be willing to exchange for the benefits of alternative drinking water standards or other policy choices.

[4] While it is reasonable to assume that individuals' donations to environmental causes or organizations reflect willingness to pay for protection and preservation of the environment, these contributions cannot be interpreted as a direct proxy for willingness to pay. Donations generally reflect only partial values; for example, some people will not make the donation if they believe that payments by others will lead to an adequate level of environmental protection.

[5] The concept of consumer surplus, and its relationship to the analysis of benefits, is described in more detail in: EPA's *Guidelines for Preparing Economic Analyses*; Freeman (1993); and Just, R.E., D.L. Hueth, and A. Schmitz, *Applied Welfare Economics and Public Policy*, Englewood Cliffs, NJ: Prentice Hall, 1982.

3.1.2 Equity Considerations

Some critics of the use of willingness to pay to value benefits are concerned about the effect of income on these values. If policy decisions were made solely on the basis of willingness to pay, critics argue, the results would not treat lower income individuals equitably. Economists deliberately attempt to separate these types of ethical judgements from the economic analysis of efficiency. They traditionally focus on how individuals value changes in their own well-being, aggregating the individual values to determine total benefits to society. If the group of individuals who benefit from a policy could compensate the group of individuals who are adversely affected, economists argue that net social welfare is maximized and the policy is considered economically efficient.

To address the limitations of this approach, economic analyses of EPA regulations are supplemented by analyses of effects on equity. As discussed in Chapter 2, analyses of environmental justice (risks and other effects on low income and minority groups) and risks to children are required for major EPA regulations. In addition, SDWA requires that EPA consider effects on sensitive subpopulations "such as infants, children, pregnant women, the elderly, and individuals with a history of serious illness, or other subpopulations likely to be at greater risk..." [SDWA, Section 1412(b)(3)(C)(i)]. SDWA also raises concerns about "affordability," particularly for small systems [SDWA, Section 1412(b)(4)(E)].[6] Requirements under other statutes mandate consideration of the costs the regulations impose on government units and private entities, as also discussed in Chapter 2.

The language of SDWA (e.g., on sensitive subpopulations and small systems) suggests that these types of equity effects should be considered when determining whether the costs of an MCL are justified by its benefits. In other words, SDWA appears to define "benefits" broadly to include both equity and traditional economic concerns about net social welfare. Because detailed information on conducting equity assessments is provided in the references cited in Chapter 2, we focus on the economic assessment of benefits in the remainder of this document.

The economic analyses described in this document can be designed to support the equity analyses. For example, in developing new studies, analysts may wish to ensure that these groups are adequately represented in the data collection strategy. When presenting the results of the analysis, analysts may decide to provide disaggregated estimates of the benefits for each subpopulation or group of concern, as well as national totals.

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[6] Information on EPA's definition of "affordability" for SDWA requirements is available in: U.S. Environmental Protection Agency, *Announcement of Small System Compliance Technology Lists for Existing National Primary Drinking Water Regulations and Findings Concerning Variance Technologies,* 63 *FR* 42032, August 6, 1998.

3.1.3 Nonquantified and Nonmonetized Benefits

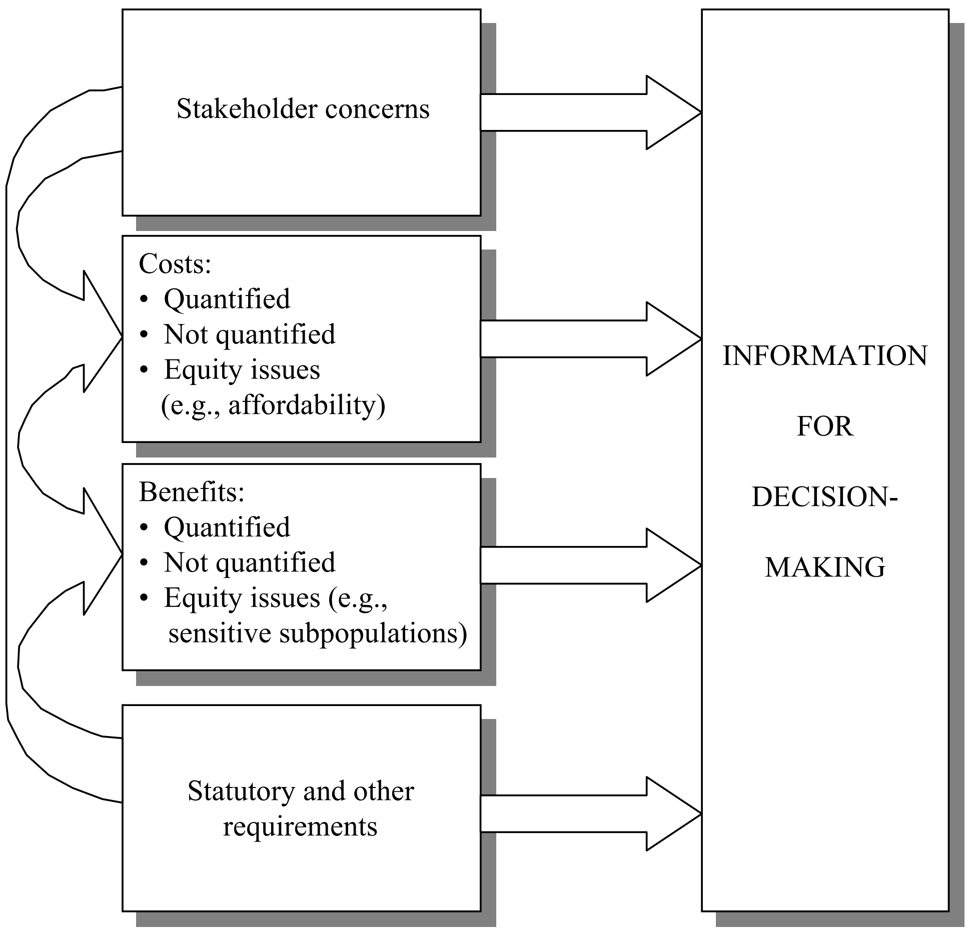
The economic framework for benefits analysis described in this document focuses on developing monetary measures for valuing benefits. Many benefits, however, can be difficult to quantify, or may be quantifiable but difficult to value in monetary terms. These types of benefits are generally described in the analysis and noted in any summary of the findings. SDWA specifically calls for consideration of such benefits, noting that "quantifiable and nonquantifiable" effects should be taken into account when establishing an MCL [SDWA, Section 1412(b)(3)(C)(i)].

For example, EPA may know that a drinking water contaminant causes adverse health effects, but lack data on how changes in exposure levels correspond to changes in the incidence or severity of the effects. Despite this uncertainty, EPA may consider these effects when establishing regulatory levels to ensure that human health is adequately protected. These nonquantified or nonmonetized benefits are often presented in the same tables or charts as the quantified results to ensure that they are taken into account by decision-makers, along with information on the uncertainties in the estimates.

When conducting a benefit-cost analysis, analysts may find that the quantified costs exceed the monetized benefits or vice-versa. The question then becomes determining whether it is reasonable to assume that the nonquantified or nonmonetized benefits (or costs) bridge the gap between the quantified costs and benefits. In some cases, the gap may be small enough that decision-makers will conclude that benefits may be equal to, or exceed, costs if nonquantified effects are considered. Analysts may also consider whether the nonquantified impacts could disproportionately impact the results across regulatory options. For example, if consideration of a particular health effect (e.g., a type of cancer not quantified in the analysis) is likely to increase the benefits estimates by a similar percentage across all regulatory options, its consideration may not change the relative rankings of the options. However, if the impacts are uneven (e.g., some regulatory options do not reduce exposure below the threshold level at which a health effect occurs), consideration of the nonquantified benefits may affect the relationship between costs and benefits for only some of the regulatory options.

The factors to be considered by decision-makers are summarized in Exhibit 3-1. As indicated by the exhibit, the analysis of costs and benefits includes quantified and nonquantified effects and addresses concerns about the distribution (or equity) of these impacts. Statutory requirements and stakeholder concerns help shape the contents of the analyses as well as the use of the analyses in regulatory decision-making.

Exhibit 3-1
Information for Decision-Making



3.2 Primary Valuation Methods

As discussed above, the preferred approach for valuing the benefits of environmental regulations generally is to determine individuals' willingness to pay (WTP) for the proposed improvements. When market data are not available, economists use a variety of other methods to estimate willingness to pay.[7] One of several approaches

[7] For information about market methods, see EPA's *Guidelines for Preparing Economic Analyses.*

for categorizing these methods is to divide them into two categories: *stated preference methods* and *revealed preference methods*.[8]

Stated preference methods typically employ survey techniques and ask respondents to "state" what they would pay for a good or service. These methods can be used to directly value the program of concern (e.g., "how much would you be willing to pay for a program that would reduce the concentrations of arsenic in drinking water from 10 µg/L to 5 µg/L?"), in which case they are designed to fully inform respondents about the effects of the reduction. Such studies are also used to assess specific effects (e.g., "how much would you be willing to pay for a program that would reduce the risks of incurring kidney disease from 10/100,000 to 5/100,000 annually?"). Stated preference methods are attractive in theory because they allow researchers to directly elicit values for particular effects. However, conducting a study that yields accurate and reliable results can be expensive, and relatively few have been completed that directly address the effects of concern for drinking water contaminants.

Revealed preference methods are based on observed behaviors that can "reveal" the values of nonmarket goods based on prices and preferences for related market goods or services. For example, if an individual would be charged $30 a month for tap water to drink, but instead pays $50 per month for bottled water that he or she believes to be cleaner and safer, then presumably this individual values the additional cleanliness and safety of the bottled drinking water at no less than $20 per month ($50 - $30 = $20). These methods use actual market data for related goods instead of relying on individuals' predictions of their own behavior. However, there is often an imperfect match between the commodities valued in these studies and individuals' willingness to pay for the effects associated with a particular rule.

Below, we introduce each of the primary research methods most likely to be used in valuing the effects of drinking water regulations; the footnotes provide references for more information on each method. We describe contingent valuation and conjoint analysis, wage-risk studies, cost-of-illness research, averting behavior studies, and avoided cost methods.

[8] More information on methods for valuing the benefits of drinking water regulations as well as examples of these studies is available in: Research Triangle Institute, *Valuing Water Quality: Theory, Methods, and Research Needs,* prepared for the U.S. Environmental Protection Agency, April 1998.

3.2.1 Contingent Valuation and Other Stated Preference Methods

Contingent valuation (CV) is a stated preference method that uses consumer surveys to directly elicit statements of willingness to pay for a commodity.[9] The values derived from the surveys are "contingent" on the realization of the scenarios described in the study. For example, a survey might ask individuals what they would be willing to pay for a specified reduction in the risk of developing kidney disease from long-term exposure to contaminants in drinking water. The researcher can define the scenario to address all the factors that may influence total willingness to pay, such as pain and suffering in the case of illness.

Contingent valuation surveys can be used to derive estimates for the full range of effects of environmental regulations, including changes in mortality and morbidity risks, improved aesthetic effects, reduced damages to materials, and changes in ecological risks. Contingent valuation is also the primary method used to assess the "nonuse" values of natural resources, such as the value of simply knowing that clean water exists.[10] Some examples of contingent valuation studies are provided in Exhibit 3-2.

[9] For more information on contingent valuation, see: Bjornstad, D.J. and J.R. Kahn, *The Contingent Valuation of Environmental Resources: Methodological Issues and Research Needs*, Edward Elgar: Brookfield, VT, 1996; Carson, R.T., "Contingent Valuation: A User's Guide," *Environmental Science and Technology*, Vol. 34, 2000, pp. 1413-1418; Hanemann, W. Michael, "Valuing the Environment through Contingent Valuation," *Journal of Economic Perspectives*, Fall 1994, Vol. 8, No. 4, pp. 19-43; Kopp, R., W.W. Pommerehne, and N.T. Schwarz, (eds.), *Determining the Value of Non-Marketed Goods*, Boston, MA: Kluwer Academic Publishers, 1997; and Mitchell, R. and R.T. Carson, *Using Surveys to Value Public Goods: The Contingent Valuation Method*, Resources for the Future, Washington, DC, 1989.

[10] For more information on nonuse values, see: Kopp, R.J., "Why Existence Value *Should* be Used in Cost-Benefit Analysis," *Journal of Policy Analysis and Management,* Vol. 11, No. 1, pp. 123-130, 1992; and Cummings, Ronald G. and Glenn W. Harrison, "The Measurement and Decomposition of Nonuse Values: A Critical Review," *Environmental and Resource Economics,* Vol. 4, pp. 225-247, 1995.

Exhibit 3-2

Examples of Contingent Valuation Studies[11]

Fatal Risks: Jones-Lee, et al. used contingent valuation to estimate individuals' willingness to pay to avoid the risk of death from auto accidents. The study also explored willingness to pay to reduce fatal risks for other people (e.g., passengers) and other types of fatalities (e.g., from heart disease and cancer). The researchers conducted face-to-face interviews with 1,103 persons in Great Britain and asked them to consider the value of avoiding fatalities expressed as "X" in 100,000 risks. Converted from British pounds (using the 1982 exchange rate) and inflated to 1997 dollars (using the GDP deflator), the average value of a statistical life resulting from this study is $4.6 million.

Minor Health Problems: Berger, et al. used contingent valuation to study willingness to pay to avoid an additional day of minor health problems such as headaches and itching eyes. Participants were asked to rank seven minor health problems, state values for symptom-free days, and summarize the values on a tally sheet. The researchers interviewed 119 respondents and determined, for example, that the average willingness to pay to avoid a day of headache is $109, and a day of itching eyes is $48 (1984 - 85 dollars).

Ground Water Protection: Powell studied individuals' willingness to pay for ground water protection using a contingent valuation survey conducted in 12 towns in the northeast. The survey was performed by mail and 1,041 people responded. The questionnaire presented information on contamination and asked respondents to indicate their willingness to pay for a water supply protection district funded through increased utility bills. Mean willingness to pay was $62 per household per year (1990 dollars).

Despite the widespread applicability and use of contingent valuation, the method has been heavily criticized in recent years. This criticism focuses largely on the measurement of nonuse values; the application of contingent valuation surveys to other types of effects tends to be less controversial. Contingent valuation studies need to be carefully implemented if they are to provide accurate and reliable estimates of willingness to pay, because individuals generally are not required to actually make the payments and may not fully understand the scenario presented in the survey.

Much of the debate has centered on the use of contingent valuation to assess damages to natural resources from oil spills and other contamination events as part of related litigation. The National Oceanic and Atmospheric Administration (NOAA) convened an expert panel in 1992 to develop guidelines for using

[11] Jones-Lee, Michael W., M. Hammerton and P.R. Phillips, "The Value of Safety: Results of a National Sample Survey," *Economic Journal,* Vol. 95, March 1985, pp. 49-72; Berger, M.C., et al., "Valuing Changes in Health Risks: A Comparison of Alternative Measures," *Southern Economic Journal,* Vol. 53, 1987, pp. 967-984; Powell, John R., David J. Allee, and Charles McClintock, "Groundwater Protection Benefits and Local Community Planning: Impact of Contingent Valuation Information," *American Journal of Agricultural Economics*, Vol. 76, December 1994, pp. 1068-1075.

contingent valuation to estimate nonuse values in such situations.[12] The panel made several recommendations for improving the reliability of these studies, such as encouraging in-person interviews (rather than mail or telephone surveys) and extensive pretesting of questionnaires and accompanying materials. Following the panel's recommendations can substantially increase the costs of contingent valuation research (e.g., to over $1 million per study) and very few existing studies fully comply with these guidelines. Several of the recommendations are controversial and may not be relevant to studies conducted for purposes other than assessment of nonuse values for litigation. As a result, EPA is currently developing its own guidelines to specifically address the use of contingent valuation for policy analysis.

Economists recently have been experimenting with other stated preference methods, particularly those referred to as conjoint analyses.[13] These methods are relatively complex and include presenting respondents with several scenarios involving various amenities and prices. Estimates of willingness to pay may be elicited based on the way in which respondents rank, rate, or construct equivalent sets of alternatives. For example, Adamowicz et al. asked respondents to make choices among several hypothetical fishing scenarios that differed along 13 attributes such as site terrain, average fish size, and water quality, and combined the results with data on actual site choices to value recreational opportunities.[14]

The "risk-risk trade-off" method is closely related to conjoint analysis, and has been used in research conducted by Viscusi and others to value changes in health risks. For example, Viscusi, et al. developed a computerized questionnaire that asked respondents to choose between places to live which varied with respect to the cost

[12] National Oceanic and Atmospheric Administration (NOAA), "Appendix I - Report of the NOAA Panel on Contingent Valuation," *Federal Register*, Vol. 58, No. 10, pp. 4602-4614, January 15, 1993. The official EPA response to these guidelines can be found in: U.S. Environmental Protection Agency, "Comments on Proposed Regulations on Natural Resource Damage Assessment," 1994.

[13] For more information on conjoint methods, see: Adamowicz, Wiktor, Jordan Louviere, and Joffre Swait, *Introduction to Attribute Based Stated Choice Methods*, prepared for the National Oceanic and Atmospheric Administration, January 1998; and Smith, V.K. "Pricing What is Priceless: A Status Report on NonMarket Valuation of Environmental Resources," *The International Yearbook of Environmental and Resource Economics, 1997/1998: A Survey of Current Issues,* (H. Folmer and T. Tietenberg, eds.), 1997.

[14] Adamowicz, Wiktor, et al., "Combining Revealed Preference and Stated Preference Methods for Valuing Environmental Amenities," *Journal of Environmental Economics and Management,* Vol. 26, No. 3, pp. 271- 292, 1994.

of living, the risks of chronic bronchitis, and/or the risks of automobile fatalities.[15] The results indicated that the median value of avoiding a case of chronic bronchitis is 32 percent of the value of avoiding an automobile fatality. When asked to trade-off changes in the cost of living for changes in risk, respondents indicated that the mean value of avoiding a case of chronic bronchitis was $457,000 (1988 dollars).

3.2.2 Wage-Risk Studies

A wage-risk (or hedonic wage) study is a revealed preference method that values changes in risk by examining the additional compensation workers demand for taking jobs with higher risks. Typically, these studies focus on small changes in the risks of accidental workplace fatalities. Researchers use statistical methods to separate changes in compensation associated with changes in risks from changes in compensation associated with other job and personal characteristics.[16] An example of a wage-risk study is provided below.

Exhibit 3-3
Example of a Wage-risk Study[17]

Fatal Risks: Moore and Viscusi used data from the Bureau of Labor Statistics (BLS) and the National Institute of Occupational Safety and Health (NIOSH), combined with information on worker attributes from the Panel Study of Income Dynamics, to estimate the value of a statistical life. The mean value of the risks studied was 5×10^{-5} for the BLS data and 8×10^{-5} for the NIOSH data. The researchers found that the value of statistical life estimates resulting from the NIOSH data ($6 million to $7 million) are significantly larger than values from the BLS data ($2 million), and argue that the NIOSH values are likely to be more accurate (1986 dollars).

The wage-risk approach has several advantages. For example, the data and methods for estimating risk reduction and associated wage differentials have been well-established through a number of studies. In addition, the approach directly measures changes in the risk of premature mortality. A number of factors, however, may complicate the use of wage-risk studies to value the benefits of drinking water regulations. For example, workplace risks usually involve some degree of voluntary acceptance, while environmental risks usually affect individuals involuntarily. In

[15] Viscusi, W.K., Magat, W.A., and Huber, J., "Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Tradeoffs," *Journal of Environmental Economics and Management,* 1991, Vol. 201, pp. 32-57.

[16] More information on wage-risk studies is provided in: Viscusi, W.P., "The Value of Risks to Life and Health," *Journal of Economic Literature*, Vol. 31, 1993, pp. 1912-1946.

[17] Moore, Michael J. and W. Kip Viscusi, "Doubling the Estimated Value of Life: Results Using New Occupational Fatality Data," *Journal of Policy Analysis and Management,* Vol. 7, No. 3, 1988, pp. 476-490.

addition, most wage-risk studies use data on middle-aged laborers (often male), who may not be representative of the members of the population most significantly affected by the risks associated with drinking water contaminants. Despite these limitations, these revealed preference studies may provide the most defensible estimates of the value of mortality risk reductions and are the source of many of the estimates used by EPA when valuing these risks, as discussed in more detail later in this chapter.

3.2.3 Cost-of-Illness Studies

Cost-of-illness (COI) studies are frequently used to value morbidity (i.e., nonfatal health effects). These studies examine the actual direct (e.g., medical expenses such as doctor visits, medication, and hospital stays) and indirect (e.g., lost wages) costs incurred by affected individuals.[18] While cost-of-illness is sometimes categorized as a revealed preference method, it does not directly measure willingness to pay. In general, the logic for using these studies to value benefits is as follows: if illness imposes the costs of medical expenditures and foregone earnings, then a regulation leading to a reduction in illness yields benefits equal at minimum to the costs saved.[19]

The cost-of-illness method has several advantages, including: (1) it is well-developed, widely applied, and easily explained; (2) many of the types of costs it includes are easily measured; and (3) existing studies provide estimates for a large number of illnesses. These studies can be designed to address all expenditures associated with an illness, regardless of whether they are paid by the patient or a third party (i.e., insurance). Lost productivity can be estimated by lost wages for those in the paid labor force; however, lost productivity for unpaid labor in the home and lost leisure time can be more difficult to measure. Examples of cost-of-illness estimates are provided in Exhibit 3-4.[20]

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[18] Sources of more information on cost-of-illness methods include: U.S. Environmental Protection Agency, *Cost of Illness Handbook*, February 2001; Hartunian, N.S., C.N. Smart, and M.S. Thompson, *The Incidence of Economic Costs of Major Health Impairments*, Lexington Books: Lexington, MA, 1981; Hu, T. and F.H. Sandifer, *Synthesis of Cost-of-Illness Methodology: Part I*, report to the National Center for Health Services Research, U.S. Department of Health and Human Services, 1981.

[19] Tolley, G., D.Kenkel, and R. Fabian (Eds.), *Valuing Health for Policy: An Economic Approach*, University of Chicago Press, 1994.

[20] Although cost of illness values have been developed for both fatal and nonfatal health effects, the value of statistical life is generally the preferred valuation measure for fatalities, as discussed later in this chapter. Cost-of-illness estimates are generally applied to those nonfatal health effects for which estimates of willingness to pay are unavailable.

Exhibit 3-4
Examples of Cost of Illness Studies[21]

Stomach Cancer: Based on research conducted by Baker et al, EPA examined both the medical costs and the lost time that result from stomach cancer. For survivors diagnosed at age 70, the direct medical costs for the 10 years following diagnosis were estimated at $85,700 per case (present value, 1996 dollars, 7 percent discount rate).

Low Birth Weight: Low birth weight in infants can lead to a variety of medical disorders, including heart failure and severe developmental disabilities. Infants with low birth weight incur high medical costs in their first year, but also tend to continue to incur elevated medical costs throughout their life. Based on research conducted by Lewitt et. al, EPA examined these costs as well as non-medical costs stemming from the need for special education and grade repetition. The present value (discounted at 7 percent) of the costs over a lifetime were estimated as $80,600 per case (1996 dollars).

Contaminated Water Supply: Harrington, et al. valued the losses incurred by households as a result of a water contamination episode in Luzerne County, Pennsylvania, during 1983 to 1984. As part of this analysis, they estimated the costs due to illness resulting from water contamination, including direct medical costs (doctor visits, hospital visits, emergency room visits, laboratory tests and medication), and time costs (including time spent obtaining medical care and related travel, lost work days, lost work productivity, and lost leisure time). The study relies on survey data (mail and phone) from affected households. Depending on the wage rate assumptions, the researchers found that cost-of-illness related losses averaged between approximately $900 and $1,300 per confirmed case of giardiasis (1984 dollars).

Although these studies are widely used for valuation, they generally do not provide estimates of willingness to pay. In many cases, cost-of-illness estimates may significantly underestimate individual willingness to pay, because they do not address the value of avoiding the pain and suffering associated with the illness, costs that an individual may have incurred in order to avoid the illness, and other factors. Cost-of-illness estimates may also occasionally overstate willingness to pay because the availability of insurance may lead people to agree to treatments that they would not willingly finance themselves.

In addition, cost-of-illness estimates do not reflect value associated with an individual's risk aversion, i.e., his or her willingness to pay to avoid future risks. Treatment also often does not return people to their original health state and hence does not address all of the benefits of avoiding the illness entirely.

3.2.4 Averting Behavior Studies

Averting behavior studies are a revealed preference method that use data on consumer behavior to estimate willingness to pay for risk reductions or other

[21] EPA's *Cost of Illness Handbook*; Harrington, Winston, Alan J. Krupnick and Walter O. Spofford, Jr., "The Economic Losses of a Waterborne Disease Outbreak," *Journal of Urban Economics*, Vol. 25, 1989, pp. 116-137.

effects.[22] For example, in the absence of regulation, individuals or households may avoid the health risks and aesthetic effects associated with drinking water contaminants by using bottled water, treating water at the tap, or using water softeners. Some of these studies also consider the medical treatments sought in response to particular types of contamination. If a regulation leads people to discontinue these behaviors, then the avoided costs may be one measure of the resulting benefits.

The averting actions considered in these studies often fall into three categories: (1) the purchase of a durable good (e.g., a water filter); (2) the purchase of a nondurable good (e.g., bottled water); and (3) a change in daily activities or behavior (e.g., boiling water before use or consuming less drinking water). Some averting actions allow an individual to completely eliminate exposure to the perceived contamination, while others allow the individual to mitigate the effects of potential exposure. The costs considered in such studies are sometimes referred to as defensive expenditures.

Use of these studies for benefits valuation can pose difficult problems related to separating out different motives for the behavior. For example, bottled water purchases may reflect the desire for convenience, or for better taste, as well as the desire to avoid the perceived risks of tap water ingestion. In addition, use of bottled water may reflect concerns about a wide variety of contaminants and health effects. It may be impossible to disentangle the various complex motives for engaging in these behaviors, and several of these motives may not be addressed by the regulations under consideration.

The extent to which such studies provide an estimate of willingness to pay is a subject of debate in the literature, and depends in part on the nature of the policy problem and the types of expenditures considered by the researcher. For example, bottled water expenditures may overstate the value of risk reductions if they also reflect convenience and taste. However, studies that consider only the money and time expended on boiling or purchasing water in response to drinking water contamination are likely to understate willingness to pay to avoid the contamination,

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[22] For more information on averting behavior studies see: Cropper, Maureen, and A. Myrick Freeman III, "Environmental Health Effects." *Measuring the Demand for Environmental Quality* (J.B. Branden and C.D. Kolstad, eds.), Elsevier Science Publishers: The Netherlands, 1991, pp. 165 - 213; Bartik, Timothy J., "Evaluating the Benefits of Nonmarginal Reductions in Pollution Using Information on Defensive Expenditures," *Journal of Environmental Economics and Management,* Vol. 15, 1988, pp. 111-127; Courant, Paul and Richard Porter, "Averting Expenditure and the Cost of Pollution," *Journal of Environmental Economics and Management,* Vol. 8, 1981, pp. 321-329; and Desvousges, W.H., F.R. Johnson, and H.S. Banzhaf, *Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method*, Northampton, MA: Edward Elgar, 1998.

since they leave out other responses to these incidents and do not address the value of averting the dread of such incidents.

In theory, researchers could combine data on averting behavior with other types of information (such as data on the associated changes in risk) to estimate willingness to pay for risk reductions. They could then apply statistical methods to separate the value of the risk reduction from the value of other effects. Because separating the value of the different effects of averting behavior is difficult (requiring a relatively large amount of data and the application of complex analytic techniques), such analysis is rarely, if ever, attempted.

In Exhibit 3-5, we provide examples of various types of averting behavior studies.

Exhibit 3-5
Examples of Averting Behavior Studies[23]

Lead Exposure: Agee and Crocker applied the averting behavior method to assess willingness to pay for reduced lead exposures. They assessed data for 256 Massachusetts children, considering the child's body burden of lead, parental decisions regarding treatment, and household characteristics such as the parent's educational level and income. They found that the mean value of a one percent reduction in child body lead burden ranged from $11 to $104 (1980 dollars).

Drinking Water Contamination -- Trichloroethylene: Abdalla, et al. researched the effect of a drinking water contamination incident in Perkasie, Pennsylvania, where trichloroethylene was detected in a well at levels far exceeding the MCL. They used a mail questionnaire to gather information about averting expenditures and behaviors in response to the contamination. They found that only 43 percent of the survey respondents knew of the contamination; of those, only 44 percent undertook averting actions such as purchasing bottled water or boiling water before use. The authors indicate that the total costs of these actions ($61,300 - $131,300 over 88 weeks; 1987-89 dollars) provide a conservative estimate of the benefits of avoiding the contamination.

Drinking Water Contamination -- Perchloroethylene: In another study, Abdalla quantified household level economic losses due to averting behavior in response to perchloroethylene groundwater contamination. Using a mail survey of residents in the affected Pennsylvania community, Abdalla determined the frequency and types of averting behaviors adopted in response to contamination, and estimated economic losses attributable to these behavior changes. He found that, on average, total household costs of averting behavior ranged from $252 to $383 (1987 dollars). Households incurred monthly costs of up to three times normal water bills as a result of behavioral changes such as home water treatment or hauling or purchasing of alternative water sources.

Drinking Water Contamination -- Giardia: The Harrington, et al. study mentioned earlier valued the costs of averting behaviors as well as the costs-of-illness resulting from a giardia contamination episode in Luzerne County, Pennsylvania. Losses due to averting behavior include water hauling or boiling, bottled water purchases, and other actions undertaken to avoid consumption of contaminated water. The study considers the time lost as well as direct expenditures, and is based on survey data (mail and phone) from the affected households. Depending on the costs included in the estimates, the researchers found that the averting behavior losses averaged between approximately $500 and $1,500 per household (1984 dollars).

[23] Agee, Mark D. and Thomas D. Crocker, "Parental Altruism and Child Lead Exposure: Inferences from the Demand for Chelation Therapy," *Journal of Human Resources*, Vol. 31, Summer 1996, pp. 677-691; Abdalla, Charles W., Brian A. Roach, Donald J. Epp., "Valuing Environmental Quality Changes Using Averting Expenditures: An Application to Groundwater Contamination," *Land Economics,* Vol. 68, No. 2, May 1992, pp. 163-169; Abdalla, Charles W., "Measuring Economic Losses From Ground Water Contamination: An Investigation of Household Avoidance Costs," *Water Resources Bulletin*, Vol. 26, 1990, pp. 451-463; Harrington, Winston, Alan J. Krupnick and Walter O. Spofford, Jr., (1989).

3.2.5 Avoided Cost Studies

Avoided cost studies are somewhat similar to the cost-of-illness and averting behavior studies discussed earlier, in that all three methods consider the expenditures averted (or displaced) by reduced exposure to contamination. The term "avoided cost" is generally used when the expenditures would be incurred by private sector or government organizations rather than individuals or households. Such studies often apply a relatively simple approach: they measure the expenditures likely to occur in the absence of the regulation, compare them to the likely expenditures once the regulation is promulgated, and use the difference to estimate benefits.[24] These methods are generally easy to apply and provide useful information for policy analysis. Whether they are a true measure of the value of related benefits depends on whether the researcher considers the effects of these costs on consumers.[25]

The avoided cost method is commonly used to assess material damages that are reduced, prevented, or mitigated by environmental regulations. Some examples include the following:

- If contaminants damage piping or other equipment, regulating the contaminant may reduce the costs of repairing the damages as well as the frequency with which the equipment needs to be replaced.

- If contaminants affect the use of water as a production input (because of the need for purity), regulating the contaminant may reduce industry's water treatment costs.

- If contaminants lead to soiling of items requiring cleansing, regulating the contaminants may reduce the costs and frequency of cleaning.

[24] For more information on avoided cost methods, see: Adams, Richard M. and Thomas D. Crocker, "Materials Damages," *Measuring the Demand for Environmental Quality* (J.B. Branden and C.D. Kolstad, eds.), Elsevier Science Publishers: The Netherlands, 1991, pp. 271-303.

[25] As discussed earlier in this chapter, consumers benefit if they are willing to pay more than current prices for a good or service. If, for example, industry costs decline because they no longer need to treat water received from the public water system, firms may pass some of these savings onto consumers in the form of lower prices. These lower prices will increase consumer surplus and may also affect producer surplus. A full treatment of avoided costs would account for these changes in consumer and producer surplus. A more detailed discussion of these issues is provided in: U.S. Environmental Protection Agency, *Handbook for Noncancer Health Effects Valuation (draft),* prepared by Industrial Economics, Incorporated, September 1999.

Examples of this type of study are provided in Exhibit 3-6. As is evident from the examples, such analysis includes only those costs that can be avoided by the regulations and that are not attributable to other causes.

Exhibit 3-6
Examples of Avoided Cost Studies[26]

Corrosion: Many of the treatment techniques used to reduce the concentration of lead in drinking water reduce the water's acidity, and thus reduce corrosion. To support development of EPA's drinking water standards for lead, Levin reviewed the literature on the costs of corrosive damages to distribution systems and residential users and determined the per capita value of these damages. She then calculated the proportion of these damages that could be reduced by water treatment, estimating that $8.50 in costs per capita could be avoided annually. She multiplied this cost by the population likely to be served by systems with corrosive waters, estimating the national value of avoided costs as $525.3 million annually (1985 dollars).

Ground Water Remediation: In the absence of clean-up of hazardous waste sites, contaminants released to ground water may eventually affect drinking water supplies in surrounding areas. Water systems and private well users will then incur costs for treating the water and/or for replacing it with uncontaminated supplies. To support development of EPA's corrective action regulations, researchers reviewed the water treatment and replacement costs that might be incurred at several sample facilities in the absence of site remediation. They assessed the likely impact of these costs on water prices and the resulting change in consumer surplus. They found that only about two percent of the loss in consumer surplus ($4.7 million) would be avoided by clean-up of the site because ground water remediation techniques may be only partially effective and can take several years to significantly reduce contaminant concentrations (1992 dollars).

3.3 Methods for Valuing Mortality, Morbidity, and Other Effects

As discussed earlier, the primary benefits of regulations establishing MCLs are effects on human health. This section provides more detailed descriptions of the particular steps needed to assess reduced mortality, morbidity, and other (non-health) effects, summarizing relevant information provided in EPA's *Guidelines for Preparing Economic Analyses* and other references cited in the footnotes.

3.3.1 Valuing Mortality Risk Reductions

The benefits of mortality risk reductions from environmental regulations are generally assessed using empirical estimates of the value of a statistical life (VSL). The value of statistical life does not refer to the value of an identifiable life, but instead to the value of small reductions in mortality risks in a population. A

[26] Levin, Ronnie, *Reducing Lead in Drinking Water: A Benefits Analysis,* prepared for the U.S. Environmental Protection Agency, 1986; Industrial Economics, Incorporated, "Chapter 9: Averted Water Use Costs," *Draft Regulatory Impact Analysis for the Final Rulemaking on Corrective Action for Solid Waste Management Units*, prepared for the U.S. Environmental Protection Agency, March 1993.

"statistical" life can be thought of as the sum of small individual risk reductions across an entire exposed population.

For example, if 100,000 people would each experience a reduction of 1/100,000 in their risk of premature death as the result of a regulation, the regulation can be said to "save" one statistical life (i.e., 100,000 * 1/100,000). The sum of the individual willingness to pay values for the given risk reduction across the population provide a value per statistical life. Continuing with the previous example, if each member of the population of 100,000 were willing to pay $50 for the risk reduction, the corresponding value of a statistical life would be $5 million (i.e., $50 * 100,000). Note that these estimates rely on studies of relatively small changes in risk; they are not values for saving a specific individual's life.

A variation on this approach involves accounting for the effect of risk reductions on the number of life years remaining.[27] The value of statistical life-year (VSLY) approach assigns a value to each year of life extended. In its simplest form, the VSLY approach translates the value of statistical life into annual values, implicitly assuming a linear relationship in which each discounted life year is valued equally. There is significant controversy over this approach, particularly because the value of remaining life years is likely to vary depending on the age of the individual and other factors.

Exhibit 3-7 presents the value of statistical life estimates applied in EPA's recent report to Congress, *The Benefits and Costs of the Clean Air Act, 1990 to 2010,* updated to 2000 values.[28] These estimates, derived from wage-risk and contingent valuation studies, range from $0.8 million to $17.8 million, with a mean of $6.3 million.[29]

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[27] An emerging literature takes a third approach to valuation, focusing on changes in life expectancy. However, this approach is not yet well enough developed for use in valuation of regulatory programs. See, for example: Johannesson, Magus and Per-Olov Johansson, "The Value of Life Extension and the Marginal Rate of Time Preference: A Pilot Study," *Applied Economic Letters,* Vol. 4, 1997, pp. 53-55.

[28] U.S. Environmental Protection Agency, *The Benefits and Costs of the Clean Air Act, 1990 to 2010*, EPA 410-R-99-001, November 1999.

[29] To allow probabilistic modeling of mortality risk reduction benefits, analysts reviewed common distributions and selected the Weibull distribution as a best fit for the mean values from these studies. Percentile values from this distribution can be used for sensitivity analysis in cases where the analyst is interested in estimating reasonable "high" and "low" values.

Exhibit 3-7
Value of Statistical Life Estimates (Mean Values in 2000 Dollars)

Study	Method	Value of Statistical Life
Kneisner and Leeth (1991 - US)	Wage-Risk	$0.8 million
Smith and Gilbert (1984)	Wage-Risk	$0.9 million
Dillingham (1985)	Wage-Risk	$1.2 million
Butler (1983)	Wage-Risk	$1.4 million
Miller and Guria (1991)	Contingent Valuation	$1.6 million
Moore and Viscusi (1988)	Wage-Risk	$3.3 million
Viscusi, Magat, and Huber (1991)	Contingent Valuation	$3.6 million
Marin and Psacharopoulos (1982)	Wage-Risk	$3.7 million
Gegax et al. (1985)	Contingent Valuation	$4.3 million
Kneisner and Leeth (1991 - Australia)	Wage-Risk	$4.3 million
Gerking, de Haan, and Schulze (1988)	Contingent Valuation	$4.5 million
Cousineau, Lecroix, and Girard (1988)	Wage-Risk	$4.7 million
Jones-Lee (1989)	Contingent Valuation	$5.0 million
Dillingham (1985)	Wage-Risk	$5.1 million
Viscusi (1978, 1979)	Wage-Risk	$5.4 million
R.S. Smith (1976)	Wage-Risk	$6.1 million
V.K. Smith (1976)	Wage-Risk	$6.2 million
Olson (1981)	Wage-Risk	$6.9 million
Viscusi (1981)	Wage-Risk	$8.6 million
R.S. Smith t(1974)	Wage-Risk	$9.5 million
Moore and Viscusi (1988)	Wage-Risk	$9.6 million
Kneisner and Leeth (1991 - Japan)	Wage-Risk	$10.0 million
Herzog and Schlottman (1987)	Wage-Risk	$12.0 million
Leigh and Folson (1984)	Wage-Risk	$12.8 million
Leigh (1987)	Wage-Risk	$13.7 million
Garen (1988)	Wage-Risk	$17.8 million

See Viscusi, W.K., *Fatal Tradeoffs* (Oxford University Press, 1992) or Viscusi, W.K., "The Value of Risks to Life and Health," *Journal of Economic Literature*, Vol. 31, pp. 1912-1946, 1993, for full references for these studies. Values are updated to 2000 dollars using the Consumer Price Index.

EPA analysts currently apply these values in most regulatory analyses due to the substantial research and peer review used to develop this range of estimates. However, EPA staff continue to explore options for refining this approach. An example of this approach is provided in Chapter 5.

Use of these estimates to value the mortality risks of environmental policies is an example of the use of benefit transfer techniques, since the subject of most of the studies (i.e., job-related risks) differs from the fatal risks reduced by environmental policies (usually associated with various forms of cancer). Benefit transfer is discussed in detail in Chapter 4 of this document. As is the case in any transfer, when applying this range of estimates to a particular rule, analysts consider differences between the scenarios considered in these studies and the risk reductions addressed by the regulations, as discussed below.

Reliable methods for adjusting these values to address potential biases have not yet been fully developed or adequately tested in most cases. More empirical research is needed to determine the appropriate adjustments, and here is substantial disagreement within the economics profession about many of these issues. In addition, several of the potential biases are counterbalancing and adjusting for only some sources of bias may lead to significant over- or underestimates of actual value. At minimum, the existing literature can be used to support a qualitative discussion of the direction and magnitude of these biases and their implications for decision-making.[30]

Sources of bias can be grouped into two general areas, including those related to the *risk characteristics* (risk perception, altruism, baseline risk, and delayed manifestation) and the *population characteristics* (income, age, and health status). Quantitative adjustment for these sources of bias is generally considered only for income and latency. As described briefly below, the other sources are usually discussed qualitatively given the status of related research.

Risk perception. The value that people place on risk reduction appears to depend in part on the nature of the risk. Individuals are likely to place different values on avoiding different types of fatal risks, even if the magnitude of the risks (e.g., a 1/100,000 change in the risk of death) is the same. These differences result, at least in part, from how individuals perceive, or feel about, risks with varying characteristics. A substantial body of literature suggests that there are nine major categories that influence individuals' perception and rankings of risks: (1) voluntariness; (2) controllability; (3) known to science; (4) known to those exposed; (5) familiarity; (6) dread; (7) certainty of being fatal; (8) catastrophic; and (9) immediately manifested.[31] Many of these characteristics are highly correlated with

[30] For a more detailed discussion of each of these sources of bias as well as references to the primary research on these topics see Industrial Economics, Incorporated, *The Use of Benefit-Cost Analysis: Valuing Fatal Risk Reductions,* prepared for U.S. Environmental Protection Agency, Office of Ground Water and Drinking Water (forthcoming), as well as EPA's *Guidelines for Conducting Economic Analyses*.

[31] See, for example, Slovic, P., B. Fischoff, and Sarah Lichtenstein, "Perceived Risk: Psychological Factors and Social Implications," *Proceedings of the Royal Society of London. Series A: Mathematical and Physical Sciences,* Vol. 430, No. 1878, 1981, pp.

each other, either directly or inversely. For example, risks with a high degree of dread, such as nuclear accidents, also have a low degree of controllability and voluntariness. As a result, differences in risk ranking can be explained by relatively few of these factors. Researchers have found that one of the most important determinants may be the degree of dread.

Altruism. Another factor to consider is the presence of altruism. The existing literature focuses on individual risk tradeoffs, but there is substantial evidence that people are willing to pay to reduce risks incurred by others (e.g., the current generation may choose to bear the costs of a program that will benefit future generations). However, many researchers advocate caution in attempting to increase value of life estimates to reflect altruism, primarily because of concerns over the potential for double-counting.[32]

Baseline Risk. Willingness to pay for fatal risk reduction may vary depending on the whether the affected individuals are already facing high or low levels of fatal risks. These risks can include both those that are relatively voluntary in nature (e.g., smoking, participating in extreme sports) as well as those that are less so (e.g., hereditary health conditions, other environmental hazards). Available evidence indicates that changes in willingness to pay are only significant when the level of baseline risk varies substantially; differences in baseline risk may have little effect in the case of the relatively modest risk reductions typical of many drinking water regulations.[33]

Delayed Manifestation *(latency and cessation lag)***.** Latency generally refers to the delay between exposure and mortality or manifestation of an adverse health effect. When there is a significant delay between manifestation of an adverse health effect and death (i.e., some cancers), this period may include illness and impaired function. Latent risks are likely to be valued differently from risks that are more immediate. Cessation lag is the time between the cessation or reduction of exposure and a reduction of risk. The existence of a cessation lag implies that the physiological damage caused by the contaminant can be completely or partially repaired over a period of time once exposure ceases, thus decreasing the risk for later disease or

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[32] For more information on altruism, see for example, Jones-Lee, M.W., "Paternalistic Altruism and Value of Statistical Life," *The Economic Journal,* Vol. 102, 1992, pp. 80-90.

[33] Hammitt, James K., "Valuing Mortality Risk: Theory and Practice," *Environmental Science and Technology,* Vol. 34, 2000, pp. 1396-1400; and Miller, Ted R., "The Plausible Range for the Value of Life - Red Herrings Among the Mackerel," *Journal of Forensic Science,* Vol. 3, No. 3, 1990, pp. 17-39.

death among populations that have already been exposed. The value placed on risks that decline quickly after cessation of exposure may be different from those that decline slowly or not at all.

Heath risks from latent illnesses, like cancer, introduce additional valuation issues. Current valuation estimates are based on risks of relatively immediate fatality. Reducing the risk of an immediate death is generally valued more highly than reducing the risk of a delayed one, assuming the risks are identical in all other respects. If cessation lag applies to a reduction in risk, the length of the lag will also affect valuation.

Income. The most robust estimates of the value of a statistical life tend to come from samples of middle-aged workers, and the income levels associated with these studies may differ from the mean for individuals affected by most drinking water regulations. In addition, national average income is increasing over time. Making adjustments for income across population subgroups may imply that public policies should favor protection of higher income individuals. This implication clearly raises difficult ethical and legal issues and, as a result, these types of quantitative adjustments are rarely implemented. Adjusting value of statistical life estimates for changes in income over time has also been discussed.[34]

Age. The studies cited in Exhibit 3-7 focus on risks incurred by the working age population, not by very young or very old individuals. Several authors have attempted to address potential differences in the value of statistical life due to differences in the average age of the affected population or the average age at which an effect is experienced. While it may seem intuitive to assume that the value of statistical life is greater for young people than older people, studies of people's willingness to engage in high risk behavior suggest a more complex relationship. For example, Jones-Lee et. al. find that the value of a statistical life for adults follows an "inverted-U" pattern, peaking around the age of 40.[35] Valuation of risks to children presents special problems.

Health Status. Individual health status (i.e., whether a person is currently in good health) also may affect the valuation of mortality risk reduction, particularly because individuals with impaired health are often the most vulnerable to death from environmental causes. Health status is distinct from age (a "quality versus quantity" distinction) but the two factors are clearly correlated and therefore are often addressed jointly.

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[34] EPA's Science Advisory Board has recommended adjusting for income changes over time.

[35] Jones-Lee, Michael W., *The Economics of Safety and Physical Risk,* Basil Blackwell: Oxford, 1989.

Extensive public health literature exists on "quality adjusted life years" (QALY). This approach provides a health state scale for quality of life based on expert medical opinions and/or survey research. It involves determining, for example, that a year with a particular condition is equivalent to a specific percentage of a full year in good health. This approach is designed for use in assessing the cost-effectiveness of alternative medical treatments, and often considers levels of activity rather than the values individuals place on changes in health status.

3.3.2 Valuing Morbidity Risk Reductions

Many regulations establishing MCLs will reduce the risks of incurring nonfatal cancers or other human health effects, including both acute (short-term) and chronic (long-term) illnesses and other effects. One method sometimes used for valuing morbidity risk reductions is the cost of illness (COI) method. However, as discussed earlier, this method has several limitations. Cost-of-illness studies often include medical expenses and lost work time, but may exclude lost leisure time or unpaid work time (e.g., for those who work in the home). Willingness to pay to avoid pain and suffering and reduce future risks are also not addressed by cost-of-illness estimates. As a result, cost-of-illness estimates are usually thought to understate willingness to pay, which is the theoretically correct measure of value and captures the effects not addressed by the cost-of-illness method.[36]

If available, analysts prefer to rely on estimates of willingness to pay rather than cost of illness estimates. However, analysts may at times wish to present both cost-of-illness and total willingness to pay estimates because of limitations in the available literature. Willingness to pay studies are available for only a few types of health risks and in some cases may have methodological problems (such as reliance on surveys using very small samples). Whether benefit transfer techniques (discussed in Chapter 4) can be used to address the limitations in the willingness to pay literature will depend on the effect of concern. Cost-of-illness studies provide estimates of avoided costs that generally can be interpreted as a lower bound on willingness to pay; the willingness to pay estimates may be less certain (depending on study quality and applicability) but more consistent with the theoretically correct definition of value.

Available research suggests that willingness to pay may be two to 79 times higher than cost-of-illness; this multiplier varies significantly for different illnesses.[37] For

[36] A more detailed discussion of estimating morbidity values and benefits transfer is available in EPA's *Guidelines for Preparing Economic Analyses*.

[37] U.S. Environmental Protection Agency, *Handbook for Noncancer Health Effects Valuation (Draft)*, prepared by Industrial Economics, Incorporated, September 1999.

example, a study of the health effects of ozone shows that the ratio of willingness to pay to cost-of-illness estimates may range from a factor of about two to four, while a study of minor health effects shows ratios as high as 79.[38]

Because of the variety of nonfatal health effects that may be addressed and the variation in the availability of suitable studies, whether and how to address potential biases and sources of uncertainty will depend on the characteristics of the particular analysis. At a minimum, analysts usually discuss qualitatively any significant differences between the effects of the regulations and the effects addressed by the valuation studies used. Where significant differences exist and quantitative adjustments or sensitivity analysis is possible, the effects of the differences may be quantified. EPA's *Handbook for Noncancer Health Effects Valuation* provides more information on these topics, along with several valuation case studies. An example of morbidity valuation is also provided in Chapter 5 of this document.

3.3.3 Valuing Other Benefits

In addition to effects on morbidity or mortality, drinking water regulations may affect the aesthetic qualities of public water supplies (taste, odor, color) or the damages they cause to materials (corrosion, soiling, build-up, impurities). The typical approach to assessing these types of effects generally involves using avoided cost methods (described above), which often are interpreted as providing a lower bound estimate of willingness to pay to avoid these effects. The actual approach will depend on the particular effect of concern, and will usually include comparing costs in the absence of the rule to the costs assuming alternative MCLs or treatment requirements are established. In some cases, studies of individual willingness to pay for these benefits (e.g., using contingent valuation) may also be available.

Some regulations establishing MCLs will provide benefits other than those specifically addressed in this document. For example, a regulation establishing an MCL or treatment requirements may improve consumer confidence in water quality, affect the health of livestock or pets, or enhance crop production. Alternatively, source water protection measures may lead to ecological benefits. These benefits are usually explored in the context of the individual rulemaking. In some cases, they may be too small to warrant quantitative assessment, and may be discussed qualitatively when presenting the results of the analysis. To quantify these types of impacts, analysts generally apply the same concepts and types of methods as discussed above, tailored to the effects of concern for the particular rulemaking.

[38] Dickie, Mark and Shelby Gerking, "Willingness to Pay for Ozone Control: Inferences from the Demand for Medical Care," *Journal of Environmental Economics and Management,* Vol. 21, 1991, pp. 1 - 16; Berger, M.C., et al., (1987).