Chapter 6: Analysis of Social Discounting

6.1 Introduction

The costs and benefits of many environmental policies are frequently paid and received at different points over the course of sometimes long time horizons. As a result, benefit-cost and related analyses that are key components of EPA's policy development and evaluation process must describe future effects in terms that help present day policy makers choose appropriate approaches for environmental protection.

One common method for doing so is called discounting, which is the process whereby the values of future effects are adjusted to render them comparable to the values placed on current consumption, costs, and benefits, reflecting the fact that a given amount of future consumption is worth less than the same amount of consumption today. Time discounting is accomplished by multiplying the future values of a policy's effects by discount factors that reflect both the amount of time between the present and the point at which these events occur and the degree to which current consumption is more highly valued than future consumption.

Despite the relative simplicity of the discounting concept, choosing a discount rate has been one of the most contentious and controversial aspects of EPA's economic analyses of environmental policies. While there are several plausible explanations for why discounting in environmental policy evaluation has been unsettled for many years, the most important is that the theoretical and applied economics literature on discounting from a social perspective is voluminous and technically complex. This makes it difficult to distill precise advice on appropriate discounting procedures for policy analysis. Moreover, in some cases the economics literature by itself does not yield simple and robust discounting rules for practical applications because making such important social decisions requires inputs from disciplines other than economics.

Nonetheless, it is important to consider the uncertainties surrounding social discounting in the broader context of applied economic analysis. Benefit-cost analysis is not a precise tool that yields firm numerical results, rather, it is a general framework for more carefully accounting for the potential and varied effects of government programs. Some of these effects can be quantified, whereas others can only be assessed qualitatively. Some may be relatively certain, whereas others may be quite speculative.

The imprecision connected with assessing benefits and costs suggests that the controversy surrounding the discount rate, in many circumstances, may have more theoretical than practical significance. For example, the effects on net benefits of alternative assumptions made for measuring and valuing uncertain effects of environmental policies can overwhelm the effects of changes in the discount rate. Additionally, for some government projects, benefits and costs may have similar time profiles, or benefits may so outweigh costs (or vice versa), that changes in the discount rate will not influence the policy implications of the analysis.

This review of the basics of social discounting begins in Section 6.2 with a discussion of some general considerations in social discounting. In Section 6.3, various discounting procedures for environmental policy assessment are presented and evaluated. This detailed discussion is divided into social discounting as applied in intra-generational contexts, where very long time horizons involving multiple generations do not apply, and discounting for inter-generational

circumstances involving long time horizons and unborn generations. EPA guidance for intra-generational social discounting is presented in Section 6.3.1.5. EPA guidance on inter-generational social discounting follows in Section 6.3.2.4. Finally, discounting and related procedures for situations in which some effects are not monetized are addressed in Section 6.4.

6.2 General Considerations in Social Discounting

This section reviews a few basic concepts and considerations central to understanding the role and importance of discounting in public policy evaluation. The focus is mainly on describing social discounting and on distinguishing discounting per se from other aspects of measuring and summarizing the costs, benefits, impacts, and other consequences of environmental policies. It also discusses the circumstances in which discounting has a large impact on the net social benefits of an environmental policy.

6.2.1 Social and Private Discounting

Discounting in public policy evaluation is normally referred to as *social discounting* or *discounting using the social rate of interest.* The process itself—applying discount factors to future flows of costs, benefits, and other consequences of environmental and other policies—is mechanically the same as the discounting process in private individuals' economic and financial calculations. What makes it "social" discounting is that it is being applied in the context of evaluating a policy's effects from an overall social perspective. Clearly, private and social perspectives can yield very different conclusions concerning, for example, the cost of engaging in an activity that also generates environmental harms.

Whether social discounting also departs significantly from private discounting, however, is less clear. Some approaches to social discounting suggest that the procedures and rates should be the same as those used in private sector discounting. Other perspectives, however, suggest that social discounting is a very different process than a single individual's discounting. In any event, at a minimum, the term "social discounting" refers to the broad society-as-a-whole point of view embodied in benefit-cost and other analyses of public policies. Whether it also connotes procedures and rates different from private discounting is a central question explored in this chapter.

6.2.2 Methods for Summarizing Present and Future Costs and Benefits

Most applications of social discounting in environmental policy evaluation involve translating future values into present ones. The conceptual foundation of discounting is based on the fact that present consumption is valued differently from future consumption. Discounting renders costs and benefits that occur in different time periods comparable by stating them all in present day terms. The resulting net present value is at least one measure of social value that might be used in evaluating environmental policies.

6.2.2.1 Net Present Value

In formal terms, the net present value of a projected stream of current and future benefits and costs is found by multiplying the benefits and costs in each year by a time-dependent weight, d_t, and adding all of the weighted values as follows:

$$NPV = NB_0 + d_1 NB_1 + d_2 NB_2 + \ldots + d_n NB_n$$

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

$$d_t = 1/(1+r)^t$$

where r is the discount rate and n is the final period in the future in which the policy's effects are felt.

To account for inflation, either real or nominal values may be used, as long as they are used consistently. In other words, nominal costs and benefits require nominal discount rates, and real costs and benefits require real discount rates. Moreover, consistent decision making requires that the same discount rate be used for both benefits and costs. Otherwise, any policy can be justified by choosing a sufficiently low discount rate for benefits, by choosing a sufficiently high discount rates for costs, or by choosing a sufficiently long time horizon.

It is important to be explicit about how time periods are designated and when, within each time period, costs and benefits accrue. Typically, time periods are years, but alternative time periods may prove desirable or necessary if costs or benefits accrue at irregular or non-annual intervals. The preceding formula assumes that $t=0$ designates the beginning of the first period. Therefore, C_0 represents startup costs such as capital costs that occur immediately upon implementation of the regulation. The formula further assumes that no additional costs are incurred until the end of the first year of regulatory compliance.[1] Benefits, if any, also accrue at the end of each time period. Therefore, the following diagram illustrates how net benefits (measured in dollars) are distributed over time.

TIME →

Year t	0	1	2	3	4	...	n
$	NB_0	NB_1	NB_2	NB_3	NB_4	...	NB_n

6.2.2.2 Annualized Values

In addition to net present value, there are other procedures for rendering costs and benefits that occur in more than one time period comparable. One method is to annualize the costs and benefits over the duration of the policy. For example, in the absence of discount rates ($r=0$), a regulation that costs $100,000 at the end of the first year, $200,000 at the end of the second year, and $300,000 at the end of the third year can be said to cost $200,000 a year in annualized costs over the three year period. Comparing annualized costs to annualized benefits is equivalent to comparing the present values of costs and benefits.

Costs and benefits each may be annualized separately by using a two-step procedure. To annualize the costs, for example, the present value of costs is calculated using the NPV formula in Section 6.2.2.1, except that the stream of costs alone, not the net benefits, is used in the calculation. This present value is then annualized (as in calculating mortgage payments) according to the following formula:

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

where,

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

PVC = present value of costs;

r = the discount rate per period; and

n = the duration of the policy.

Note that the annualized cost is the amount one would have to pay at the end of each period t to add up to the same cost in present value terms as the stream of costs being annualized. There is no initial cost at $t=0$ in this annualization. Such an initial cost can be incorporated into the annualization using the slightly different formula:

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

This approach is also useful when analyzing non-monetized benefits, such as reductions in emissions or reductions in health risks, *when benefits are constant over time*. The average cost-effectiveness of a policy can be calculated by dividing the annualized cost by the annual benefit to produce measures of program effectiveness, such as the cost per ton of emissions avoided.

6.2.2.3 Net Future Value

Finally, there is yet another way of rendering costs and benefits that occur in more than one time period comparable. Instead of discounting all future values back to the present, it is possible to accumulate them forward to some future time period—for example, to the end of the last year of the policy's effects, n. Here, the net benefit test is whether the accumulated net future value (NFV) is positive.

$$NFV = d_0NB_0 + d_1NB_1 + d_2NB_2 + ... + d_{n-1}NB_{n-1} + NB_n$$

[1] See EPA (1995) for an example in which operating and monitoring costs were assumed to be spread out evenly throughout each year of compliance. While the exponential function above is the most accurate way of modeling the relationship between present value and a continuous stream of benefits and costs, simple adjustments to the equations above can sometimes adapt them for use under alternative assumptions about how dollar flows are distributed over time.

NB_t is the net difference between benefits and costs (B_t-C_t) that accrue in year *t* and the accumulation weights, d_t, are given by

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

where *r* is the discount rate.

Each of these methods employs an interest rate to translate values through or across time, so the methods are not really different ways to determine the benefits and costs of a policy. Instead, they are different ways to express and compare costs and benefits that occur in multiple time periods on a consistent basis. Discounting places all costs and benefits in the present time period, annualization spreads them smoothly through time, and accumulation states them all in the future. But each procedure uses the same discount (interest) rate, so they are different ways to describe the same underlying phenomenon.

Depending on the circumstances, one method might have advantages over the others. For example, annualizing the costs of two machines with different service lifetimes might reveal that the one with the higher total cost actually has a lower annual cost because of its longer lifetime. Similarly, discounting to the present is likely to be the most informative procedure when analyzing a policy that requires an immediate investment and offers a stream of highly variable future benefits.

In general, however, these are alternative ways of translating costs and benefits through time using an interest rate. Therefore, the analysis, discussion, and conclusions presented in this chapter apply to all methods of translating costs, benefits, and effects through time, even though the focus is mostly on discounting.

6.2.3 Sensitivity of Present Value Estimates to the Discount Rate

The impact of discounting streams of costs and benefits in public policy evaluation is sometimes large and sometimes not, depending on the circumstances. When all effects occur in the same period, discounting may be unnecessary or superfluous: net benefits are positive or negative regardless of the discount rate used or the procedure for translating them through time. Similarly, when costs and benefits of a policy are largely constant over the relevant time frame, discounting costs and benefits will produce the same conclusion concerning the policy as would examination of a single year's costs and benefits. Of course, higher discount rates will reduce the present value of any future cost or benefit. But if costs and benefits of a policy occur simultaneously and if their relative values do not change over time, whether the net present value of such a policy is positive does not depend on the discount rate.

Discounting can substantially affect the present value net benefits estimates for public policies when there is a significant difference in the timing of costs and benefits. For example, if the costs of a policy are incurred today, they are not discounted at all. But if the benefits will occur 30 years from now, the present value of the benefits, and, hence, the net present value of the policy's effects, depends critically on the discount rate used.

Suppose the cost of some environmental policy that is incurred entirely in the present is $1 billion and that after 30 years a benefit results that is estimated to be worth $5 billion in the future. Without discounting, a policy that offers benefits five times its cost appears to be a very worthwhile social investment. Discounting the $5 billion future benefits, however, can radically alter the economic assessment of the net present value of the policy. Five billion dollars, 30 years in the future, discounted at one percent is $3.71 billion, at three percent it is worth $2.06 billion, at seven percent it is worth $657 million, and at 10 percent it is worth only $287 million. In this case, the range of discount rates generates over an order of magnitude difference in the present value of benefits. And, longer time horizons will produce even more dramatic effects on a policy's net present value. Hence, the choice of the discount rate largely determines whether this policy is considered, at least on economic efficiency grounds, to offer society positive or negative net benefits.

Thus, for government projects and policies that require large initial outlays or that have long delays before benefits are realized, the selection of the discount rate can be a major factor in determining whether the net present value is positive. Many of EPA's policies fit these profiles. Large investments by public or private parties are usually required early on, whereas the benefits of those investments either accrue for many years thereafter, such as improvements in health and environmental quality, or will not begin for many years, such as reductions in the contamination of environmental systems from hazardous waste, landfill facilities, and the protection of the earth's atmosphere and climate.

6.2.4 Distinguishing Discounting from Other Procedures

Discounting is only one of several components that are necessary in order to produce comparable estimates of a policy's costs and benefits that accrue over more than one time period. Discounting is a technique for translating values from one time period to another in order to express the values of a policy's consequences in consistent terms. It is not, however, a method for actually determining the future values of future costs and benefits. Two considerations related to determining these future values—projecting future values based on present ones and accounting for risk—are closely related to discounting.

6.2.4.1 Future Values of Costs and Benefits

The future value of one of an environmental policy's effects may hinge critically on the assumed rate of growth of wealth over time. There may also be a connection between increasing wealth and the discount rate for expressing future values in present day terms. Nevertheless, the process of determining the values of future costs and benefits and then translating them into present terms are two conceptually distinct procedures.

It is generally appropriate to conduct each of these tasks separately. And, it is prudent to avoid attempting to "correct" for errors in one procedure by "adjusting" the other. For example, it is technically possible to use a current valuation for a future benefit whose future value is expected to increase, but then reduce the discount rate to reflect that assumed rising valuation through time. Nevertheless, this is usually unwise because the values of other consequences of a policy might not follow the same rate of increase over time. Thus, these might be over- or under-corrected by the adjusted discount rate. The only way to avoid that result would be to use a different adjusted discount rate for each cost and benefit stream, which is generally inappropriate.

6.2.4.2 Risk and the Social Discount Rate

The relationship between risk and the rate of return on assets has been an important subject in modern finance. Risk considerations also have played a role in the controversy surrounding the selection of an appropriate discount rate for benefit-cost analysis. For example, one recommendation is that public projects with risky or uncertain future costs and benefits should be discounted at a higher rate reflecting those risks, just as it is in the private sector.

The concept of risk is often interpreted narrowly as being measured by the variability or range of possible outcomes of a project. Greater variation implies more risk according to this view. But the notion of risk should be conceptualized more broadly. Rather than being taken in isolation, the risk of a project is measured by its effect on the variability in outcomes of the entire portfolio of assets. In general, the degree of risk associated with an asset is measured in terms of the covariance of its returns with those of the portfolio of assets to which it is added.[2] When viewed from this broader perspective, most environmental projects are either riskless or reduce risk. This is because most environmental projects have benefits and costs that are widely dispersed and that are uncorrelated or negatively correlated with future measured income and other aspects of economic welfare.

Nevertheless, the costs and benefits of some environmental policies can be risky in this broader sense. In these cases, it is commonly argued that the discount rate should be adjusted upward by a risk premium to value future uncertain returns. However, this is generally not the correct procedure because it requires the discount rate to reflect both the risk of future returns as well as the length of time until they materialize. That is, if the goal is to reduce the present value of a project's returns to reflect their risk, the same decrease in present value will be

[2] An assumption underlying this analysis is that the asset being acquired is a very small fraction of the portfolio of assets already held. If this assumption is violated, the variability in returns of the asset can directly affect the variability of returns to the entire portfolio. The potential costs and benefits of environmental policies generally are spread among large numbers of people, however, which satisfies the condition that the asset acquired be a small portion of the portfolio of assets already held.

produced by a smaller increase in the discount rate the longer the delay until the returns are received.[3]

Economic theory suggests using two different "instruments" to accomplish the two different goals. One such procedure to account for risk is to value a project's uncertain returns using the *certain monetary equivalent* or *certainty equivalent*. As discussed in Chapter 5 of this document, this is the amount risk averse individuals would be willing to pay with certainty for the risky prospect. The certainty equivalent should then be discounted using the rate of interest individuals use to discount other perfectly certain flows.

Hence, to properly account for risk in benefit-cost analyses, the first step should be to evaluate whether a project is actually risky from the broader perspective of society's larger portfolio of assets. Many government policies are not risky at all, so that their expected values of costs and benefits can be discounted directly using a risk-free interest rate. For projects that offer truly risky prospects, however, certain monetary equivalents for these returns should be derived and then discounted to the present using a risk-free rate. The discount rate should not be "adjusted" to account for risky costs and benefits.

6.3 Approaches to Social Discounting

This review of the basics of social discounting and the new EPA guidance on the subject begins with discounting in conventional or *intra-generational* contexts, where very long time horizons involving multiple generations do not apply. Next, approaches for *inter-generational* social discounting, involving very long time horizons and unborn generations, are presented and evaluated.

The main purpose of this discussion is to provide a broad overview of the extensive literature on social discounting in order to distill from it practical guidelines for environmental policy evaluation. It is not, however, a detailed review of the literature on discounting in public project evaluation, which is vast in scope and volume. Excellent sources for summaries of the social discounting literature are Lind (1982a), Lind (1982b), Lind (1990), Lind (1994), Lyon (1990), Lyon (1994), Kolb and Scheraga (1990), Scheraga (1990), IPCC (1996), Pearce and Turner (1990), and Pearce and Ulph (1994).

6.3.1 Intra-Generational Social Discounting

This section explores social discounting in conventional or intra-generational contexts, specifically those in which very-long-time-horizon issues are not important features. Most of the traditional discounting literature focuses on these circumstances. Intra-generational contexts may well have decades-long time frames, but they do not explicitly confront the extremely long time horizons and impacts on unborn generations that are central to the extensions of social discounting research into climate change, nuclear waste disposal, and other such policy issues. The division of the problem into intra-generational and inter-generational social discounting helps to understand the substantially different contribution economic approaches can offer in each area.

The discussion begins with a brief review of the analytical foundations of conventional social discounting. It next outlines the major social discounting approaches suggested in the literature. The section concludes with a review of the concrete conclusions and advice offered by the traditional discounting literature and the new guidance developed by EPA for use in social discounting in intra-generational contexts.

6.3.1.1 Analytical Foundation of Intra-Generational Social Discounting

Conventional social discounting is rooted firmly in the view that the government is acting on behalf of its citizens in undertaking public projects and promulgating environmental and other policies. Therefore, benefit-cost analysis of these actions should seek to estimate the costs and benefits experienced by all of the affected parties, and in so

[3] Note that if discount rates are adjusted to incorporate risk, the adjustment is not always upward. Risky benefits are worth less than their expected value, so an upward adjustment of the discount rate will reduce the present value. But uncertain costs would require a downward adjustment of the discount rate to increase the present value to reflect the fact that risk averse individuals would pay more than the expected value of the costs to avoid bearing the uncertain prospect.

doing determine whether, in aggregate, the gainers under a policy would be able to compensate the losers.

This foundation for social discounting has an important implication for the choice of a social discounting method. Just as consumer sovereignty dictates that the government should incorporate the specific values that particular individuals place on outcomes that affect them in assessing its actions, the government should also discount future costs and benefits in the same way that the affected individuals do. Strict adherence to the principles of consumer sovereignty is necessary in order to determine how much each person would agree he or she is made better or worse off by a given policy in present value terms.

The analytical and ethical foundation of the intra-generational social discounting literature thus rests on the traditional test of a "potential" Pareto improvement in social welfare—whether gainers could compensate the losers. This framework fundamentally casts the consequences of government policies in terms of collections of individuals contemplating changes in their own consumption (broadly defined) over time. Thus, social discounting in this context should seek to mimic the discounting practices of the affected individuals.

The Paretian economics point of view, however, is not the only ethical perspective possible in this context. As discussed in the section on inter-generational discounting, another approach is to cast the problem in terms of maximizing a social welfare function that includes utilities of present and future individuals and is maximized according to an alternative set of objectives and constraints. While alternative social welfare functions could apply to intra-generational circumstances, it is generally confined to inter-generational contexts. Hence, although there is nothing inherent in a short-time-horizon policy that dictates that only the Paretian perspective is appropriate for intra-generational situations, this is the most commonly accepted point of departure in the social discounting literature for these circumstances. It is also worth considering the two very distinct foundations for social discounting separately because their implications for determining the social discount rate are quite different. The Paretian economic approach suggests that the social discount rate is to be found by examining the preferences of affected parties, while the discount rate under alternative social welfare functions is not necessarily based on the preferences of existing individuals.[4]

6.3.1.2 Fundamental Procedures for Intra-Generational Social Discounting

Given the reasonably precise and circumscribed objective of social discounting as described above, the volume of literature on the topic is surprisingly diverse and complex. This section briefly reviews the major approaches suggested in the literature and evaluates their implications for practical social discounting in environmental policy assessments. The section concludes with a summary of recommended practices for social discounting in intra-generational contexts.

Consumption Rate of Interest Approach for Social Discounting

The economic literature begins by pointing out that under a variety of restrictive assumptions—no taxes, no risk, perfect capital markets—the task of discounting effects experienced by individuals would be straightforward. Analysts should simply use the observable market rate of interest that underlies the intertemporal consumption allocation decisions of those same individuals. The rate at which individuals are willing to exchange consumption over time is normally referred to as the "consumption rate of interest."

The simplifying assumptions (especially the absence of taxes on investment returns) imply that the consumption rate of interest equals the market interest rate, which also equals the rate of return on private sector investments. In this case, individuals discount future consumption at the market rate of interest, which is also the rate at which consumption can be translated through time via private sector investments. Hence, if the government seeks to value costs and benefits in present day terms in the same way as the affected individuals, it also should discount using the market rate of interest.

One of the simplifying assumptions underlying this result—that the consumption rate of investment at which consumers discount future consumption equals the rate of

[4] This concept is also treated in Chapter 10 in the discussion of ways to jointly consider efficiency and equity within a single analytical framework.

return on private sector investment—probably does not hold in practice. Taxes on private sector investment returns can cause the *social rate* at which consumption can be traded through time (the pre-tax rate of return) to exceed the rate at which *individuals* can trade consumption over time (the post-tax consumption rate of interest).

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

Over several decades, a very large body of economic literature developed, analyzing the implications for social discounting of divergences between the consumption rate of interest and the social rate of return on private sector investment. The dominant approaches in this literature are briefly outlined here.

Consumption Rate of Interest-Shadow Price of Capital: The Traditional View

One approach that enjoys widespread support among economists recommends that social discounting in intragenerational contexts should use the consumption rate of interest to discount future costs and benefits that have been valued in terms of future consumption. Intuitively, this procedure makes sense because the government is assumed to be valuing future consequences of its policies just as the affected citizens would. If individuals discount future consumption (and the costs and benefits of a public policy) using the consumption rate of interest, then so should the government. So, the social rate of discount should equal the consumption rate of interest.

But, if the costs of financing a public project or the costs of regulatory compliance displace private investments, society loses the total pre-tax returns from those foregone investments. Private capital investments might be displaced if, for example, public projects are financed with government debt and the supply of investment capital is relatively fixed. This is the "closed economy" condition. In this case, discounting costs and benefits using the consumption rate of interest (the post-tax rate of interest) does not seem to capture the fact that society loses the higher, social (pre-tax) rate of return on foregone investments.

Under the consumption rate of interest-shadow price of capital approach for social discounting, the social value of displacing private capital investments is taken into account prior to discounting. Under this approach, when a public project displaces private sector investments, the correct method for measuring the social costs and benefits requires an adjustment of the estimated costs (and perhaps benefits as well) prior to discounting using the consumption rate of interest. This adjustment factor is referred to as the "shadow price of capital."[5]

The shadow (social) price (value) of private capital is intended to capture the fact that a unit of private capital produces a stream of social returns at a rate greater than the rate at which they are discounted by individuals. If the social rate of discount is the consumption rate of interest, then the social value of a $1 private sector investment will be greater than $1. The investment produces a rate of return for its owners equal to the post-tax consumption rate of interest, plus a stream of tax revenues (considered to be consumption) for the government.

To illustrate this simply, suppose that the consumption rate of interest is three percent, that the pre-tax rate of return on private investments is five percent, that the net-of-tax earnings from these investments are consumed in each period, and that the investment exists in perpetuity (amortization payments from the gross returns of the investment are devoted to preserving the value of the capital intact). A $1 private investment with these characteristics will produce a stream of private consumption of $0.03 per year and tax revenues of $0.02 per year. Discounting the private post-tax stream of consumption at the three percent consumption rate of interest yields a present value of $1. Discounting the stream of tax revenues at the same rate yields a present value of about $0.67. The social value

[5] Lind (1982a) remains the seminal source for this approach in the social discounting literature.

of this $1 private investment—the shadow price of capital—is thus $1.67, substantially greater than the $1 private value that individuals place on it.

Therefore, if financing a public project displaces private investments, this "consumption rate of interest-shadow price of capital" approach suggests adjusting the project's costs upward by the shadow price of capital and then discounting all costs and benefits using a social rate of discount equal to the consumption rate of interest. To apply this approach, the first step is to determine whether private investment flows will be altered by a policy. Typically, project costs are thought to displace private capital, at least in part, although project benefits could encourage additional private sector investments. Next, all of the altered private investment flows (positive and negative) are multiplied by the shadow price of capital to convert them into consumption-equivalent units. All flows of consumption and consumption-equivalents are then discounted using the consumption rate of interest.

A simple example of this method is as follows. Suppose the pre-tax rate of return from private investments is five percent and the post-tax rate is three percent, with the difference attributable to taxation of capital income. Assume as well that increases in government debt displace private investments dollar-for-dollar, and that increased taxes reduce individuals' current consumption also on a one-for-one basis.[6] Finally, assume that the $1 current cost of a public project is financed 75 percent with government debt and 25 percent with current taxes and that this project produces a benefit 40 years from now that is estimated to be worth $5 in the future.

Using the consumption rate of interest-shadow price of capital approach, first multiply 75 percent of the $1 current cost (which is the amount of displaced private investment) by the shadow price of capital (assume this is the $1.67 figure from above). This yields $1.2525, to which is added the $0.25 amount by which the project's costs displace current consumption. The total social cost is therefore $1.5025. This results in a net social present value of about $0.03, which is the present value of the future $5 benefit discounted at the three percent consumption rate of interest ($1.5328) minus the $1.5025 social cost.

Thus, under the consumption rate of interest-shadow price of capital approach, costs are adjusted upward to reflect the higher social costs of displacing private investments, but discounting for time itself is accomplished using the consumption rate of interest—consistent with how individuals trade and value consumption over time.

Variants of this approach exist. For example, the Kolb-Scheraga (1988) approach recommends annualizing capital expenditures using the pre-tax rate and then discounting all cost and benefits using the consumption rate of interest.

Other Social Discounting Approaches

Other approaches for social discounting in the literature have been recommended on and off over the years. These alternatives focus on different methods than the shadow price of capital approach for evaluating policies that displace private sector investments. However, the procedures these approaches use will not generally produce a correct estimate of the social present value of a policy's costs and benefits. Some of these other methods for social discounting are reviewed and evaluated below.

Weighted average of pre- and post-tax rates of return: A major alternative approach for addressing the divergence between the higher social rate of return on private investments and lower consumption rate of interest is to set the discount rate for public projects equal to a weighted average of the two. The weights would equal the proportions of project financing that displace private investment and consumption, respectively. Intuitively, this approach would set an overall project discount according to the amount lost by displacing consumption (using the lower consumption rate of interest) and the amount lost by displacing investments (using the higher social rate of return on private capital).

For example, suppose the social rate of return from private investments is five percent and the consumption rate of interest is three percent, as above. Suppose further that 75 percent of a public project's costs are financed using

[6] The assumption that additional government borrowing crowds out private investment dollar-for-dollar is not critical to the example. If crowding out is less than dollar-for-dollar, then the 75 percent of the project's cost that is financed by additional debt would be further divided into the proportion of that percentage of the cost that displaces private investment, which should then be adjusted using the shadow price of capital, and the remainder of the cost, which is drawn from consumption and therefore does not need to be adjusted.

government debt, with the remaining 25 percent of the costs raised through taxation. Finally, assume that government debt crowds out private investment on a dollar-for-dollar basis and that increased taxes reduce individuals' current consumption also on a one-for-one basis. The weighted average approach then suggests that the social rate of discount should be 75 percent of five percent plus 25 percent of three percent, or four and a half percent. If the proportions of the project's financing from each revenue source were reversed, however, the weighted average discount rate would instead be 25 percent of five percent plus 75 percent of three percent, or three and a half percent.

This approach has enjoyed considerable popularity over the years, and is probably acceptable for similarly timed cost and benefit flows.[7] As presented above, however, it is technically incorrect and can produce net present value results substantially different from the correct result (where "correct" is defined by the consumption rate of interest-shadow price of capital approach). The problem with the simple weighted average approach is that it seeks to accomplish two tasks using the social discount rate—pure time discounting and adjusting for the displacement of private investments that yield pre-tax social returns higher than the consumption rate of interest.

In general, the "synthesized" discount rate based on the social rate of return from private investments and the consumption rate of interest that accomplishes both objectives—and so arrives the correct present value—depends on the timing of the cost and benefits flows. A simple weighted average based only on project cost components will not in general produce the correct result.

To understand this, consider how the weighted average discount rate approach performs for the simple numerical example discussed above. Assume that the social rate of return on private investments is five percent, that the consumption rate of interest is three percent, that increases in government debt displace private investments dollar-for-dollar, that the $1 current cost of a public project is financed 75 percent with government debt and 25 percent with current taxes, and that the project produces a benefit 40 years from now that is estimated to be worth $5 in the future.

The weighted average social discount rate approach would suggest discounting the future benefit at a four and a half percent rate (0.75 times five percent plus 0.25 times three percent). This produces an estimated net social present value of -$0.14, which is the present value of the future $5 benefit discounted using a four and a half percent rate ($0.86) minus the current year $1 cost. In this case, the weighted average social discount rate approach suggests that the project's net social present value is negative. But earlier, the consumption rate of interest-shadow price of capital approach was applied to exactly this scenario, concluding that the net social present value is positive.

The problem with the weighted average approach is that its method for accounting for the higher social cost of displaced private investments is to "over discount" the benefits. But the amount of "over discounting" necessary in this example to adjust for the actual social costs of the project's costs depends on the time profile of the benefit stream—the farther in the future the benefits occur, the less "over discounting" is needed. The source of the project's financing is therefore insufficient to define a single rate of social discount that will produce correct net social present value results for any given policy.

Accordingly, to derive the weighted average discount rate that will produce the correct net present value requires that the consumption rate of interest-shadow price of capital method be used first to compute the net present value. The discount rate that produces this correct present value based on discounting costs and benefits, but not adjusting for the shadow price of capital, can then be calculated. There seems to be little purpose to this exercise because it requires the net present value of a policy to be computed using accurate procedures first before the adjusted discount rate can be derived.

Opportunity cost of capital: Another approach for social discounting argues that the government should not invest (or compel investment through its policies) in any project that offers a rate of return less than the social rate of return on private investments. Stated another way, because the citizens collectively enjoy the benefits of all

[7] Lind (1982b) provides a clear exposition of the weighted average approach for estimating the social discount rate. The large literature on this topic, spanning the 1960s through the early 1980s, has been summarized well by Lind and others.

public and private investments, welfare will be higher overall if the government invests in projects with the highest rates of return.[8]

Critics of this social investment rule argue that the government cannot realistically tax citizens and then invest in private sector projects. Therefore, the issue is not what "could" be done with the funds, but rather what "would" be done with them. Thus, if the government obtains funds for a project through taxation and this displaces only private consumption, then relative to consuming the resources today, welfare is increased as long as the project generates future benefits that exceed those costs when discounted at the consumption rate of interest. Of course, it remains true that welfare would be further increased if the funds were devoted to an even more valuable project.

A closely related opportunity cost-based observation is that the government faces a menu of projects and, for whatever reason, is not able to undertake all projects that have positive net social benefits when computed using a social rate of discount equal to the consumption rate of interest. In this event, the opportunity costs of funding one program are the benefits of other programs not funded. Proponents of this view typically conclude that the "hurdle" discount rate for a particular project should be equal to the rate of return offered by other projects foregone.

Regardless of the particular point of departure, the central point of the opportunity cost strand of the social discounting literature is valid. Social welfare will be improved if the government invests in projects that have higher values than if it invests in lower value ones. Hence, if the net present value of benefits of all courses of action are examined using the consumption rate of interest and the set with the highest net benefits are pursued, social welfare will be higher than otherwise.[9] So stated, this advice is correct.

However, it does not follow that rates of return offered by alternative private or public projects define the level of the social discount rate. An alternative project might produce large benefits over the future and thus offer a large "rate of return." But if individuals discount these future benefits using the consumption rate of interest, the correct way to describe this project is that it offers substantial present value net benefits. In general, the opportunity cost argument is not about the social discount rate *per se*, but about correctly and consistently examining the social values of all alternatives. As was the case for the shadow price of capital, an alternative project with a high rate of return will have a high social net present value. But this does not imply that its rate of return should become the social rate of discount to be used for pure time discounting for other projects.

Consumption Rate of Interest-Shadow Price of Capital: The New View

Over the years, the consumption rate of interest-shadow price of capital approach to social discounting has gained increasingly wide acceptance among economists. Recently, however, a key assumption in that analysis has been questioned—the assumption that the economy is "closed" to foreign capital flows—and an alternative hypothesis concerning government crowding out of private investment has been put forward.[10] According to this new view, earlier analyses implicitly assumed that capital flows into the nation were either nonexistent or very insensitive to interest rates, a "closed economy" assumption. Empirical evidence suggests, however, that international capital flows are quite large and very sensitive to interest rate changes. In this case, the supply of investment funds to the U.S. equity and debt markets is likely to be highly elastic (the "open economy" condition) and, thus, private capital displacement is much less important than it was previously thought to be.

Under this new view, it is inappropriate to assume that financing a public project through borrowing will result in dollar-for-dollar crowding out of private investment. If, instead, financing public projects results in no crowding out of private investment, then no adjustments using the shadow price of capital are necessary. Benefits and costs should be discounted using the consumption rate of interest alone. However, the literature to date does not adequately support the assumption of zero crowding out. It is more likely that there exists some degree of private capital

[8] Many authors cite high opportunity costs of public investments. Among these are Birdsall and Steer (1993), Schelling (1995), and Lyon (1994). On the technical issue of rates of return vs. net present values, see Lind (1990) and Cowen and Parfit (1992).

[9] Clearly, such an approach cannot be followed when a particular action is mandated.

[10] See Lind (1990) for this revision of the consumption rate of interest-shadow price of capital approach.

displacement within the spectrum between zero and dollar-for-dollar displacement. The degree of crowding out will depend on the magnitude of the policy or program being analyzed. Unfortunately, while the shadow price of capital adjustment requires an assessment of the proportion of costs that displace investment,[11] the literature provides little empirical evidence as to the relationship between project size and capital displacement.[12]

6.3.1.3 Applying the Consumption Rate of Interest Approach to Environmental Policies

The extension of the consumption rate of interest-shadow price of capital approach to the case of an economy "open" to substantial capital flows is relatively recent. And, as is true for most of the discounting literature, virtually all of the discussion focuses on public project financing, rather than on environmental policies that largely mandate that private parties undertake certain actions or expenditures in pursuit of social objectives. Finally, while it is intuitive to argue that private investments are not displaced by either additional government borrowing or mandatory private investments for environmental protection, it is often the gross gains and losses of the affected parties in the economy that are the focus of economic impact analyses. How the change in the assumption concerning the availability of investment funds to the economy translates into these gross gains and losses is critical for conducting accurate environmental policy assessments.

For all of these reasons, it is worth clarifying the capital displacement and adjustment issue for environmental policies that mandate capital investments in the context of both the "open" and "closed" economy assumptions regarding capital flows.

Environmentally-Mandated Private Investments in a "Closed" Economy

To focus closely and exclusively on the shadow price of capital adjustment issue, some simplifying assumptions are helpful. Assume that there is no risk and uncertainty, that all firms and the government borrow at the interest rate i, that taxes on investment income are levied on all sources of such income at a rate of t, and that the resulting post-tax interest rate, r $(=i\times(1\text{-}t))$, is the rate at which individuals discount future consumption.[13] Further, assume that the net-of-tax returns from all investments are consumed in each year (to assist in making this illustration as simple as possible). Assume, finally, that the supply of investment funds is perfectly inelastic with respect to their price, the interest rate.

Consider, first, a public project that costs \$1, is financed through taxes on labor and other factors of production (but not capital), and offers future environmental benefits. Assuming that increased current taxation only reduces consumption, the cost of the project is this amount of reduced current consumption. Future benefits, once valued in terms of future consumption, can be discounted to the present using a social rate of discount equal to the consumption rate of interest. For the remainder of this discussion, the benefits side of the calculations will be ignored to focus on the cost calculation considerations.

Now, consider exactly the same project, but assume that it is financed only through government borrowing, which crowds out an equal amount of private sector investment. To calculate the costs of this project financing, it is helpful to analyze the impacts on the different entities affected. First, the private sector investors who lend \$1 to the government instead of to private firms are indifferent. They receive the interest rate i from either source and, therefore, continue to receive a stream of returns net-of-tax equal to $\$1\times r$.

Next, consider the government, which can be thought of as representing the interests of citizens in future years. The foregone private investments would have generated a stream of tax revenues of $\$1\times t\times i$ each year, which is lost. But the increased public debt is taxable, so the government regains this $\$1\times t\times i$ each year and the streams of gained and lost tax revenues offset each other. Nevertheless, the government must service this new debt

[11] See footnote 5 for a description of this adjustment.

[12] See Lind (1990) for a summary of the empirical literature in this area.

[13] The relevant tax rate t is the effective marginal tax rate. It is difficult to determine this rate in the aggregate with any reasonable degree of accuracy.

by raising future taxes each year by the amount $\$1 \times i$ (assuming, for simplicity, that the debt is a perpetuity).

As a result, the cost of financing this public project through government debt in a closed economy context is a stream of decreased consumption experienced in the future of $\$1 \times i$ per year forever. The present value of this stream of foregone consumption computed using the consumption rate of interest, r, exceeds $1. This is the essence of the shadow price of capital adjustment rationale. The value of i exceeds r in this example because of the tax wedge between the social (pre-tax) rate of return on investments and the (post-tax) consumption rate of interest. This is the equivalent of observing that a taxable investment yields a private return of r per year to the investor and a "return" of $t \times i$ to the government in the form of tax revenues.

Assume now that the relevant investment is a private sector capital project that must be undertaken in order to comply with an environmental policy. To estimate the social costs of this requirement under the closed economy assumption, two polar cases are useful to examine: no cost shifting to consumers or other factors of production and full cost shifting to consumers through higher product prices.

In the case of no cost shifting to consumers, the owners of the firms required to make these investments either must obtain debt or equity funds or reduce their other investment and lending activities, to comply. Wherever the required funds originate, two facts are clear. One is that other taxable investments of $1 will not be undertaken. The second is that, because the price of the products or services into which this environmental investment flows does not rise, the mandated investment will produce no "return" for their owners or for the government in the form of future tax revenues. The result is that the owners of the affected firms lose a stream of investment income, $\$1 \times r$, and the government loses a stream of tax revenues of $\$1 \times t \times i$, because of the displaced private investment. But since $r = i \times (1-t)$, this adds up to a stream of costs of $\$1 \times i$ per year. Once again, this is essentially the shadow price of capital adjustment.

Now assume that the cost of the mandated environmental investment is shifted to consumers through higher prices, which rise by enough to provide the full social pre-tax return of i. In this case, the owners of the firms required to make these investments are indifferent. Similarly, the government is indifferent—it still receives a stream of tax revenues from the $1 investment. Here, however, it is consumers of the affected product or service who are not indifferent. In fact, the product price increases they face are precisely enough to provide the $\$1 \times i$ pre-tax social return on the mandated investment. Here again, this is essentially the shadow price of capital adjustment.

Environmentally-Mandated Private Investments in an "Open" Economy

The central difference between the closed and open economy contexts concerns the conditions of supply of investment funds. In the closed economy case, the amount of these funds is fixed, so the total available for all projects, private and public, is constant. Hence, the key to analyzing that case lies in tracing the implications of altering the composition of the investments undertaken with and without a new public project or a new environmental policy mandating private investments.

In the open economy context, however, what is fixed is not the supply of investment funds, but the price at which they may be obtained. In this case, all investments worth undertaking without a new public project or a new environmental policy requiring investments will still be worth undertaking with those new policies—so that there will be no impact on capital availability and the level of private sector investments. This suggests that measuring the costs of these policies in this open economy context may be slightly different than in the closed economy case.

Purely tax-financed public projects are not discussed here because the results for that case do not depend on the assumption concerning the supply of capital. For debt-financed public projects, however, the results under the closed and open economy assumptions are very different. In this open economy case, the government's increased $1 of borrowing does not change the level of U.S. private sector investment. Hence, the government must service the debt at a cost of $\$1 \times i$ per year, but also gains from that $\$1 \times i \times t$ of tax revenues from these new taxable interest payments.[14] The net cost is only $\$1 \times i \times (1\text{-}t)$, which is the stream of future reduced consumption citizens will experience as the net cost of the new public project. But because $i \times (1\text{-}t) = r$, this stream of reduced consumption is equal to $\$1 \times r$. Discounted at the consumption rate of interest, r, the present value cost is $1. This is the rationale for not using the shadow price of capital adjustment.

To analyze the implications of the open economy assumption for mandated private investments, the no- and full-cost pass-through polar cases continue to be helpful. In the case of no-cost pass-through, the results are very simple. The owners of the firms required to make the investments to comply with an environmental policy will obtain the necessary funds either from their own resources that would have been invested elsewhere, or from other sources, and undertake the required investments. Because the price of the services or products subject to the new policy do not rise to compensate for these costs, no return to these owners or to the government in the form of tax revenues will result. But, because the supply of investment funds to the economy is perfectly elastic, no other private sector investments will be foregone.

The result in this case is that the owners of the entities required to make these investments will lose a stream of private investment returns of $\$1 \times r$ (their net-of-tax return on production investments) if the mandatory investment causes them to reduce investment elsewhere. Alternatively, the owners of the affected firms may increase their demands for investment funds in the market and continue with their pre-policy investment plans. Nevertheless, because i is constant, all investment projects that were profitable before the policy is imposed will still be profitable and these investments will be undertaken as if the policy did not exist. Hence, the government loses no tax revenue as a result and no shadow price of capital adjustment is appropriate here.

Finally, if the costs of the mandated private investments are fully passed through to consumers, the owners of the affected firms are now indifferent. The government and the consumers of the relevant services or products, however, are not. First, the consumers of the affected sector's output face price increases equivalent to $\$1 \times i$, which is the amount necessary to fully recoup the full pre-tax social return on the invested capital. But the government gains a stream of tax revenues associated with this mandated investment, amounting to $\$1 \times i \times t$ per year. Again, all other investments are still undertaken because of the assumption regarding the supply of investment funds.

As a result, the net cost to society is the price increase borne by consumers, equal to $\$1 \times i$ per year, minus the increase government tax revenues—which represents future reduced taxation—of $\$1 \times i \times t$ per year, for a net cost of only $\$1 \times i(1\text{-}t) = \$1 \times r$. Thus, the shadow price of capital adjustment is not necessary here. But, note that the cost increase for the firm and its consumers is measured by the pre-tax amount per year, $\$1 \times i$, not the net social cost of $\$1 \times r$ per year. The former is the relevant measure for modeling private sector "economic impacts" and for assessing the gross gains and losses of a policy, while the latter represents the social perspective.

6.3.1.4 Summary of Advice from the Economics Literature

The vast majority of the traditional social discounting literature has focused on exploring the implications for public project evaluation of a few, probably very important, departures from the idealized no-other-distortions simplified economy for which unambiguous social discounting recommendations can be made. Yet, in the development of that literature, many matters have been addressed and are considered by many contributors to this literature to be somewhat settled, some of which are discussed above, and others not (largely because they are not directly social discounting issues). In particular, for intra-generational social discounting:

- There is reasonable agreement that the social rate of discount ought to reflect the private rates of consumption discount of the citizens affected.
- If social and private returns from private investments are different, then adjustments should be introduced

[14] The taxability of the interest payments on the increased amount of government debt in an open economy context is complex because of the international nature of capital markets. Generally speaking, taxes are owed on interest earnings from government obligations to the country that pays the interest, although there are exceptions to this rule. Hence, if U.S. citizens increase their lending to the U.S. government, the interest earnings would clearly be taxable. If foreign investors purchase the increased U.S. government debt, normally these interest payments are taxable as U.S. income.

to reflect this when and if policies alter private investment flows.

- Uncertainty and risk should largely be addressed through appropriate valuation of costs and benefits (e.g., certain monetary equivalents) rather than through modifications of the discount rate.
- Changes in the values of environmental goods and other such factors should likewise be reflected in direct cost and benefit measurements, not through adjustments to the social discount rate.
- Irreversibility of consequences is an option value concept and requires separate treatment in benefit-cost analyses, but it does not provide a reason to adjust the discount rate.
- Opportunity costs of other public and private uses for funds should be considered in evaluating the desirability of undertaking a particular public investment or policy. That a project offers a positive present value of net benefits when discounted using the consumption rate of interest does not by itself imply that the policy should be undertaken.

These conclusions demonstrate the significant progress made in the theoretical social discounting literature, especially regarding the implications of divergences between social and private rates of returns on investments. However, exactly what numerical rate of interest to use for social discounting in practical policy evaluations remains somewhat unsettled.

Moreover, some recent literature questions some of the most basic premises underlying the conventional social discounting analysis. For example, recent studies of individuals' financial and other decision making suggest that even a single person may appear to value and discount different actions, goods, and wealth components differently. This "mental accounts" or "self-control" approach suggests that individuals may well evaluate some aspects of the future quite differently from other consequences. The discount rate an individual might apply to a given future benefit or cost, as a result, may not be observable from market prices, interest rates, or other phenomena. This may be especially the case if the future consequences in question are not tradable commodities. Some recent evidence from experimental economics also indicates that discount rates appear to be lower the larger the magnitude of the underlying effect being valued, higher for gains than for losses, and tend to decline as the length of time to the event increases.[15]

Despite all of these limitations, practical economic analyses must use social discounting to assist in evaluating environmental policies. Hence, even limited guidance is helpful in developing recommendations for practical analyses. What is offered in the empirical literature for choosing a social discount rate focuses on estimating the consumption rate of interest at which individuals translate consumption through time with reasonable certainty.

For this, historical rates of return, post-tax and after inflation, on "safe" assets, such as U.S. Treasury securities, are normally used, typically resulting in rates in the range of one to three percent.[16] Some studies have expanded this portfolio to include other bonds, stocks, and even housing and this generally raises the range of rates slightly. It should be noted that these rates are *ex post* rates of return, not anticipated, and they are somewhat sensitive to the time periods selected and the classes of assets considered.[17] A recent study of the social discount rate for the United Kingdom places the consumption rate of interest at two to four percent, with the balance of the evidence pointing toward the lower end of the range.[18]

[15] Shefrin and Thaler (1988) and Thaler (1985) are central sources for the mental accounts idea and Lowenstein and Thaler (1989) report numerous examples of various inconsistencies and other aspects of individual intertemporal choices.

[16] Estimates of the consumption rate of interest (an individual's marginal rate of time preference) could be based on either after-tax lending or borrowing rates. Because individuals may be in different marginal tax brackets, have different levels of assets, and have different opportunities to borrow and invest, the type of interest rate that best reflects marginal time preference will differ among individuals. Additionally, individuals routinely are observed to have several different types of savings, each possibly yielding different returns, while simultaneously borrowing at different rates of interest. Thus, discerning an average marginal rate of time preference from observed interest rates is very difficult. However, the fact that, on net, individuals generally accumulate assets over their working lives suggests that the after-tax returns on savings instruments generally available to the public will provide a reasonable estimate of the consumption rate of interest.

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

[18] Lind (1982b) offers some empirical estimates of the consumption rate of interest. Pearce and Ulph (1994) provide the estimates of the consumption rate of interest for the United Kingdom. Lyon (1994) provides estimates of the shadow price of capital under a variety of assumptions.

Finally, for the shadow price of capital, even less concrete empirical guidance is available. This parameter depends on the consumption rate of interest, the gross-of-tax rate of return on private investment, and the rate of consumption out of net investment returns, among other factors. Depending on the magnitudes of these factors, shadow prices from close to one to three, 20, 100, and infinity can result. Lyon (1990) has an excellent review of how to calculate the shadow price of capital and possible settings for the various parameters that determine its magnitude. Moreover, the shadow price of capital adjustment will require an assessment of the proportion of project costs that displace private investment. Whether or not this adjustment is necessary appears to depend largely on whether the economy in question is assumed to be open or closed and on the magnitude of the intervention or program considered relative to the flow of investment capital from abroad.[19]

6.3.1.5 Guidance for Intra-Generational Social Discounting

For economic analyses of intra-generational policies analysts should apply the consumption rate of interest approach. There should be no adjustments using the shadow price of capital unless there are strong reasons to believe that a particular policy will affect the level of U.S. private sector investment.[20] Based on historical rates of return on relatively risk-free investments, adjusted for taxes and inflation, a consumption rate of interest measured at two to three percent is justified.

OMB's own guidance on discounting[21] currently recommends discounting using a rate of seven percent, an estimate of the average real pre-tax rate of return generated by private sector investments. EPA economic analyses therefore should provide estimates of the present values of costs and benefits using both a two to three percent rate and OMB's guidance on discounting. In some cases, a sensitivity analysis at discount rates within this range may provide useful information to decision makers.

In addition, all analyses should present the undiscounted streams of benefits and costs. This is not equivalent to calculating a present value using a discount rate of zero. In other words, the flow of benefits and costs should be displayed rather than a summation of values.

6.3.2 Inter-Generational Social Discounting

This section focuses on social discounting in the context of policies with very long time horizons involving multiple generations. Policies with potential inter-generational impacts include global climate change, radioactive waste disposal, groundwater pollution, and biodiversity. Because of potentially large or catastrophic impacts on unborn generations and because policies with very long time horizons often involve high costs imposed by current generations, there is less agreement in the literature on the appropriate approach to discounting over very long time horizons. This section attempts to present a balanced discussion of alternative points of view. The discussion first focuses on how the point of departure for inter-generational discounting differs in some very fundamental ways from that of intra-generational social discounting. Next, various approaches for deciding whether and how to discount when evaluating inter-generational policies are reviewed. Finally, the section concludes by summarizing the advice offered by the economics literature and EPA's new guidelines for inter-generational social discounting.

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[19] Studies suggesting that increased U.S. government borrowing does not crowd out U.S. private investment generally examine the impact of changes in the level of government borrowing on interest rates. The lack of a significant positive correlation of government borrowing and interest rates is the foundation of this conclusion. Because changes in yearly U.S. government borrowing during the past several decades have been in the many billions of dollars, it is reasonable to conclude that EPA programs and policies costing a fraction of these amounts are not likely to result in significant crowding out of U.S. private investments.

[20] As the estimation of the shadow price of capital can be a costly exercise, analysts should use a value-of-information approach to determine whether it is worthwhile to pursue a quantitative assessment of the effects of private capital displacement. Should a quantitative assessment be undertaken, the analysis should include a sensitivity analysis of alternative assumptions regarding the degree of crowding out.

[21] OMB Circular A-94, "Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs," October 29, 1992 (note: see updates to cost-effectiveness rates—most recent released in January 2000, http://www.whitehouse.gov/OMB/circulars/a094/a094.html (accessed 8/28/2000)) and U.S. Office of Management and Budget, *Guidelines to Standardize Measures of Costs and Benefits and the Format of Accounting Statements,* report M-00-08, March 22, 2000,.http://www.whitehouse.gov/media/pdf/m00-08.pdf (accessed 8/28/2000).

6.3.2.1 Analytical Foundation of Inter-Generational Social Discounting

One obvious problem with long-time-horizon policies is that many of the people affected are not alive. Hence, while the preferences of each affected individual are knowable (if probably unknown in practice) for intra-generational social discounting problems, they are essentially unknowable for those involving future generations not yet born. This is not always a severe problem for practical policymaking, especially when policies impose relatively modest costs and benefits, or when the benefits begin immediately, or in the not too distant future. And most of the time, it suffices to assume that future generations will have preferences much like those of present generations.

The more serious challenge posed by long-time-horizon situations arises primarily when costs and benefits of an action or inaction are very large and are distributed asymmetrically over vast expanses of time. Here, the crux of the problem is that future generations are not present to participate in making the relevant social choices. Instead, these decisions will be made only by existing generations. Social discounting in these cases can no longer be thought of as a process of consulting the preferences of all affected parties concerning their valuation today of effects they will experience in different time periods.

Moreover, compounding interest over very long time horizons can have profound impacts on the inter-generational distribution of welfare. An extremely large cost far enough in the future has essentially zero present value when discounted at even a low rate. But a modest sum invested today at the same low interest rate also can grow to a staggering amount given enough time. Therefore, mechanically discounting very large distant future effects of a policy without thinking carefully about the ethical implications is not advised.

6.3.2.2 Perspectives on Inter-Generational Social Discounting

The social discounting literature contains many different perspectives on social discounting in inter-generational contexts. This section briefly describes the major approaches and their theoretical motivations. The focus in this discussion is on the social discount rate itself, so such other issues as the shadow price of capital adjustments, while clearly still relevant under certain assumptions, are kept in the background.

Social Welfare Planner Approach

One popular recommendation is that social discounting for inter-generational policies should be based upon methods economists have used for many years in optimal growth analyses. In these models, the policy maker is understood to be maximizing the utilities of all present and future generations using a well-defined social welfare function.[22]

In optimal growth models, the social rate of discount generally equals the sum of two factors. One is a discount rate for pure time preference, which measures the degree to which the social planner favors the utility of current and near future members of society over that of individuals in the more distant future. The other is an adjustment reflecting the fact that the marginal utility of consumption will decline over time as consumption per capita increases (equal to the elasticity of marginal utility multiplied by the rate of increase of consumption over time).

If the world actually corresponded to the theoretical construct of an optimal growth model and there were no taxes or other distortions, the social discount rate as defined in these models would be equal to the market interest rate. And, the market rate of interest, in turn, would also be equal to the social rate of return on private investments and the consumption rate of interest. But because the world contains many distortions, and is not likely to conform to the conditions that characterize optimal growth models, the social rate of discount is not observable in the economy.

Recent practical applications of this approach to very-long-time-horizon analyses have therefore attempted to estimate the social discount rate by constructing it from its components. Most assume that the rate of pure time discount is zero, adhering to the ethical precept that the policy maker ought not to inherently favor present generations' consumption over that of future generations. For the other component of the social discount rate,

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

hypothetical, but perhaps plausible, estimates of the elasticity of marginal utility and the rate of growth of consumption over time are introduced. The product of these two factors is the implied social discount rate. This computational procedure in essence derives an implied social rate of discount under the assumption that future generations will be richer than current generations, so that the marginal utility of consumption is projected to fall over time. Rates developed using this technique generally range from one-half percent to three percent.[23]

Optimal growth modeling, however, is only one strand of the substantial body of research and writing on intertemporal social welfare maximization and optimal growth. This literature extends from the economics and ethics of inter-personal and inter-generational wealth distribution, to the more specific environment-growth issues raised in the "sustainability" literature, and even to the appropriate form of the social welfare function, e.g., utilitarianism or Rawls' maxi-min criterion.

Clearly, economics alone cannot provide definitive guidance for selecting the "correct" social welfare function or the social rate of time preference. Nevertheless, economics can offer a few insights concerning the implications and consequences of alternative choices and some advice on the appropriate and consistent use of the social welfare function approach as a policy evaluation tool.

Approaches Based on Existing Individuals' Preferences

The major alternative to the social welfare planner approach for inter-generational discounting is to rely on the preferences of current individuals for an appropriate discount rate. At its core, this perspective rejects the view that the problem is one of balancing the interests of all humans who will live now and in the future. Instead, according to this perspective, it is fundamentally about individuals alive today allocating their scarce resources to competing ends, one of which happens to be the welfare of future generations. Several specific approaches fall into this category.

- **Consumption rate of interest/infinitely-lived individuals:** Although not popular in theoretical terms, in practice it is common to adopt the approach of simply making no great distinction between inter-generational and intra-generational social discounting. Models of infinitely-lived individuals, for example, suggest the consumption rate of interest as the social discount rate. But the assumption that people live forever is contrary to the fact that individuals actually do not live long enough to experience distant future consequences of a policy and to report today the present values they place on those effects. As such, models of infinitely-lived individuals essentially ignore the fundamental problem posed in evaluations of policies that affect distant future generations.

- **Inter-generational discounting vs. time discounting:** Another suggestion for social discounting in inter-generational contexts is to examine possible differences between how current individuals evaluate the welfare of their descendants versus how they discount their own future consumption.[24] It is possible that the year-by-year exponential time discounting [25] that underlies an individual's allocation of his or her own consumption in the present and the future does not apply to this individual's valuation of his or her descendant's welfare. That is, a person might indeed value future generations' consumption less than his or her own current and future consumption, but not as low as would be implied by standard discounting techniques. An individual's present valuation of the consumption of successive future generations might decline gradually and approach some constant positive value, so that the value of a unit of consumption by a person 10 generations from the present might be considered to be the same as a unit of consumption by a person 11 generations from now.

  A related line of reasoning suggests that large-scale catastrophic consequences in the future are viewed differently than marginal changes in welfare, so that it matters little whether these possibilities are 100 years

[23] See IPCC (1996), pp. 131-132.

[24] Sources discussing this approach include Rothenberg (1993), Cropper et al. (1992), Shefrin and Thaler (1988), Thaler (1985), and Cowen and Parfit (1992).

[25] Exponential time discounting applies discounting factors to future values that increase as the time between the present and the future when those values will be experienced increases. As a result, the present value of a benefit to be enjoyed 50 years from now is much higher than the present value of the same benefit if it accrues 100 years from now.

or 1,000 years in the future. If so, applying exponential, or year-by-year, time discounting to such future consequences is inappropriate.

- **Revealed/stated preferences for altruism:** According to this view, environmental policies that affect distant future generations are considered to be altruistic acts.[26] As such, they should be valued by current generations exactly the same as other acts of altruism. Hence, the discount rate in question is not that applied to an individual's consumption, but instead that applicable for an individual's valuation of the consumption or welfare of someone else.

  At least some altruism is apparent from international aid programs, private charitable giving, and bequests within overlapping generations of families. But the evidence suggests that the importance of other people's welfare to an individual appears to grow weaker with temporal, cultural, geographic, and other measures of "distance." The implied discount rates that survey respondents appear to apply in trading off present and future lives are also relevant under this approach. One such survey (Cropper, et al., 1992) suggests that these rates are positive on average, consistent with the rates at which people discount monetary outcomes, and decline as the time horizon involved lengthens.

- **Opportunity cost of alternatives:** A variety of perspectives in the inter-generational discounting literature converge on the broad notion that devoting resources to long-time-horizon environmental projects—largely because low discount rates appear to make these attractive in present value terms—neglects numerous other social investment opportunities with higher values.[27]

  Advocates of this point of view point to numerous alternative social investments that would generate far larger benefits now and in the future, such as basic infrastructure, education, medical assistance, and other projects in developing nations.

  Depending on the context, this point of view is often expressed in two different ways: (1) many other investments would be more beneficial to society and so long-time-horizon environmental programs face very high opportunity costs, and (2) the rates of return offered by these alternative investments are high and these rates ought to be used as the social rate of discount.

  As noted earlier in the context of intra-generational social discounting, the first statement of this opportunity cost argument is the correct one, the second is somewhat problematic. The opportunity costs of alternative government or private investment programs are appropriately measured by calculating their present values using the social rate of discount. If these projects have higher rates of return than the social discount rate, their social present values will also be high. But this does not imply that the social rate of discount itself ought to be set equal to some alternative project's rate of return. For example, an alternative project might offer a very large rate of return for only one year, but this should not become the social rate of discount for very-long-time-horizon projects and policies.

- **Paretian compensation tests:** One final approach for social discounting in an inter-generational context returns to the theoretical motivation and ethical underpinnings of intra-generational social discounting. This approach views social discounting in inter-generational contexts as a question of whether the distribution of wealth among many different generations could be adjusted in order to compensate the losers under an environmental policy and still leave the gainers better off. Whether gainers could compensate losers hinges on the rate of interest at which society (the U.S. presumably or perhaps the entire world) can transfer wealth across hundreds of years. Some argue that in the U.S. context, a good candidate for this rate is the federal government's borrowing rate.

  What lies at the foundation of this approach is the goal of maintaining overall inter-generational equity. The implicit assumption is that society starts from a position at which the distribution of wealth across present and future generations is "acceptable." Then

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

[27] Many authors cite high opportunity costs of public investments. Among these are Birdsall and Steer (1993), Nordhaus (1993), Schelling (1995), and Lyon (1994).

it is discovered that some current environmental action or inaction will impose large burdens on future generations. To maintain inter-generational equity, some sort of accumulation fund is necessary to provide compensation for those harms.

While this approach offers solid advice for selecting a social discount rate for inter-generational policy evaluation, its resolution of the many difficult social choice problems posed by such policies rests on two critical assumptions. One is that the initial distribution of inter-generational wealth is socially acceptable. If this is not the case, it is not clear that attempting to maintain that distribution after discovering the long-term environmental problem is an appropriate goal.

Second, if the compensation fund is not accumulated, then the decision not to remedy this environmental problem is once again recast as an inter-generational equity problem, not a question purely of economic efficiency. There is considerable skepticism regarding the willingness of the current generation to provide these compensation funds and a significant concern that intervening generations might not continue the accumulation process. Thus, actually undertaking the process of locking away sufficient savings for distant future generations to compensate them for environmental harms is a very different matter than determining the rates of interest at which such a fund might grow.

6.3.2.3 Summary of Advice from the Economics Literature

There is little consensus in the economic literature on social discounting for inter-generational policies. In particular, the fundamental choice of what moral perspective should guide inter-generational social discounting—a social planner who weighs the utilities of present and future generations, the preferences of the current generations regarding future generations, or perhaps other approaches—cannot be made on economic grounds alone.

It is important, however, to view this result in the proper context. In fact, the practical effect of this lack of consensus concerning social discounting for inter-generational policies is not as profound as it at first appears. The major problems with discounting in long-time-horizon contexts occur in probably a few cases out of a vastly larger set, particularly where costs and benefits are inherently high and are substantially divorced in time. But the environmental policies that fit this description are uncommon because most environmental programs are relatively short in duration and reversible, with their time frames determined largely by capital investments.

6.3.2.4 Guidance for Inter-Generational Social Discounting

Based on the theoretical social discounting literature and other considerations, economic analyses of policies with inter-generational effects should generally include a "no discounting" scenario by displaying the streams of costs and benefits over time. This is not equivalent to calculating a present value using a discount rate of zero (i.e., the flow of benefits and costs should be displayed rather than a summation of values).

Economic analyses should present a sensitivity analysis of alternative discount rates, including discounting at two to three percent and seven percent as in the intra-generational case, as well as scenarios using rates in the interval one-half to three percent as prescribed by optimal growth models. The discussion of the sensitivity analysis should include appropriate caveats regarding the state of the literature with respect to discounting for very long time horizons.

6.4 Discounting and Non-Monetized Effects

Despite analysts' best efforts to assign monetary values to all of the consequences of an environmental policy, there are instances in which monetization is not feasible. This section briefly explores social discounting when some elements are not expressed in monetary terms.[28]

[28] Although this discussion focuses exclusively on non-monetized benefits, many cost categories are often not monetized as well. The time costs consumers experience as a result of some policies, the financial costs of business delays that result from others, and the quality and performance impairments caused by yet other policies often are not monetized in economic analyses. Discounting policy regarding these non-monetized effects should largely track discounting practices for monetized costs unless there are reasons for not doing so, similar to those described in this section, for leaving some non-monetized benefits undiscounted.

6.4.1 Perspectives on Discounting Non-Monetized Effects

One strategy for addressing future non-monetized effects is to discount them as though they had been monetized. Some argue, however, that environmental benefits that have not been monetized cannot—or should not—be discounted and summarized together with costs in a cost-effectiveness or benefit-cost summary. Two basic lines of reasoning are normally offered. One is that because discounting is essentially a financial process designed to evaluate investment decisions, it is only relevant to dollar-denominated streams, and so benefits that are in physical rather than dollar terms *cannot* be discounted. Discounting some types of benefits, such as avoided damages to human lives or natural resources, treats these tangible risk-related benefits as monetary outcomes, when they are not in fact financial consequences.

The other line of reasoning for not discounting non-monetized benefits is that it is ethically unacceptable to discount physical units. If, for example, cancer cases that occur in the future are discounted to the present, this effectively asserts that a future cancer case is not really a cancer case, but rather is only 80 percent, 20 percent, or some other fraction of a "full" current cancer case. Discounting therefore somehow cheapens the future effect's value or reduces its importance and is unfair to future individuals or generations whose lives or natural resources are at stake. This argument is often applied not only to human health and environmental effects that are simply enumerated, but also to those that are monetized.

Evaluating these arguments requires a clear understanding of the various reasons why benefits might not be monetized in any given analysis. In some cases, benefits are not monetized because the environmental and health impacts may be unknown, so that only changes in emissions, production, exposure, or other imperfect proxies for benefits, damages, or harms, are available. Sometimes there may be an estimated time stream of human health and environmental impacts, but the needed valuation tools and information on how to monetize the benefits are not—or are only partially—available. Finally, in still other cases, physical effects have been estimated and could be monetized, but this last step—converting measured physical effects into dollar values of benefits—has simply not been taken.

6.4.2 When Discounting Non-Monetized Effects Is Appropriate

In many cases, quantitative information on the time streams of physical effects is available and these effects are measured in terms of human health consequences and ecosystem damages that correspond to endpoints that are normally monetized. If so, then these non-monetized benefits ought to be discounted if monetized costs and benefits are discounted. Discounting non-monetary effects in these cases is not inherently different from discounting these units after attaching a unit value in dollar terms. What is being conveyed is the notion that effects felt farther in the future are worth less in today's terms than those that occured earlier in time. Thus, if two policies have identical current costs and the same amount of benefits in the future except that one produces these benefits earlier in time, the policy that offers earlier benefits will have a higher social value.

Choosing not to discount non-monetized benefits can have perverse consequences. First, to the extent that the act of discounting and the choice of discount rate embody a rational investment criterion, failing to discount non-monetized benefits may produce results that appear to be irrational or intrinsically unappealing. Suppose, for example, there is a policy that is estimated to save five lives in the year it is implemented. This policy can either be implemented today (Option A) or 20 years from now (Option B), and the undiscounted costs in current dollars are the same for both options. If the discounted costs are compared with undiscounted benefits, a cost-effectiveness evaluation will clearly favor Option B. Thus, failing to discount benefits can produce a situation in which society has little motive to pursue current environmental benefits because by investing instead, larger net environmental benefits can be gained in the more distant future.

Finally, surveys that examine individuals' attitudes toward public policies with non-monetized benefits suggest that people do appear to apply a positive discount rate to these future effects. For example, contingent valuation studies (Cropper et al., 1992; Carson et al., 1987; Horowitz and Carson, 1990) that look at individuals' preferences for saving lives find that individuals prefer projects that save lives in the near term over equivalent cost projects that save lives in the future.

6.4.3 When Discounting Non-Monetized Effects Might Not Be Appropriate

While there are many cases in which non-monetized benefits can and should be discounted along with all of the other costs and benefits of environmental policies, there are others in which benefits are not monetized for reasons that pose more significant problems for discounting. Specifically, sometimes the available measures of benefits are very poor proxies for ultimate damages, making it difficult to discount them correctly.

When an analysis stops far short of the physical effects that are good proxies for damages, the relationship between harms and emissions—or other relevant physical measures—might be poorly understood. In the case of the greenhouse effect, for example, the ultimate impact of a ton of greenhouse gas emitted in a given year depends on the subsequent change in the time paths of temperature, sea level, and other variables, and on the physical effects and economic impacts accompanying these changes. Changes in temperature depend, in turn, on the magnitude of emissions of all greenhouse gases over time and their radiative forcing. Further, the impacts of climate change may depend not only on the absolute levels of these effects, but on the rate at which they occur. Because linking quantified physical harms to a unit of emissions is a difficult task, discounting greenhouse gas emissions would be a premature and problematic step in determining the cost-effectiveness of two alternative emission reduction strategies.

Similarly, even when benefit estimates are based on linkages from emissions to other physical and biological endpoints, often these benefit measures are still not close enough to the endpoints of ultimate concern to allow discounting. For example, although pollution damages can be measured in terms of species diversity, ecosystem health, and forest productivity, the further detailed linkages from those damages to current and future recreation, production, non-use, or other values identified by economists and ecologists often do not exist.

Discounting non-monetized effects is also not warranted when doing so actually conceals information of value to policy makers. For example, suppose a policy reduces current and future effluent discharges to a river. Suppose further that this river has a complex chemistry, so that interactions between the effluent reduced by this environmental policy and other natural and human inputs to the river are unknown and/or the relationship between effluent discharges and damages is nonlinear (e.g., the river is subject to degradation only after passing some threshold). Here, the same quantities of effluent reduction in different time periods are not necessarily identical in their effects, so not only is there a time element to contend with, but also possible differences in ultimate environmental benefits. In this case it might be far more useful to display the stream of effluent reduction and probabilities of exceeding thresholds each year, rather than to discount all of the future effluent reduction.

In all of these examples, the problem is that analysts have an incomplete understanding of the relationship between emissions—and production or other physical units that are potentially subject to control—and the actual harm to human health or the environment that result. However, a general preference for earlier benefits over later ones still applies. The problem is that discounting in these cases masks important information by implicitly assuming that a unit of benefits in one period has an identical effect on the ultimate benefit consequences of concern as a unit of the same benefit in another period. When non-monetized benefits measures are far from the human health and other benefit categories of true concern, this assumption often is contrary to reality.

When it is not appropriate to discount certain non-monetized benefits, comparisons of costs and benefits can still be made without directly discounting the benefits. For example, if costs and benefits occur in each time period over the course of a policy, and these do not change significantly over time, net social benefits can be explored without discounting by examining a representative year's costs and benefits. Similarly, if the benefits are relatively constant through time, but the costs are not, the costs can be annualized and compared to the annual benefits using cost effectiveness analysis. Another approach is to cumulate costs forward with interest to compare this future value to the benefits, a method that is particularly suitable when the benefits occur in only one future year. If none of these methods applies, simply presenting the streams of costs and monetized and non-monetized benefits to policy makers is often sufficient.

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Chapter 7: Analyzing Benefits

7.1 Introduction to Analyzing Benefits

At its roots, benefits analysis develops monetary values to inform the policy making process. These values are important because they allow decision makers to directly compare costs and benefits using the same measure (i.e., dollars). A complete benefits analysis is also useful because it makes explicit the assumptions about the value of benefits embedded in different policy choices. This chapter focuses on those benefits that can be expressed in terms of dollars. Chapter 10 discusses the presentation of non-monetized benefits, those that cannot be expressed in dollar terms.

This chapter presents information on the theory and practice of benefits assessment for environmental policies. The discussion focuses on the benefits possible from a "typical" EPA policy or regulation that reduces emissions of contaminants into the environment. However, the principles discussed here apply to other types of EPA policies, such as those that provide information or regulatory relief.

Most EPA benefits analyses face two serious challenges. First, a given policy may produce many different benefits, but it is seldom possible to obtain a single, comprehensive value estimate for the collection of effects. This will often leave analysts with no alternative but to address these effects individually, aggregating values to generate an estimate of the total benefits of a policy alternative. Although there are exceptions to this "effect by effect" process for benefits analysis, much of the discussion in this chapter assumes that analysts will be forced to adopt this approach.

The second major challenge faced by analysts is the difficulty of conducting original valuation research in support of specific policy actions. Because it is often too expensive or time consuming to perform original research, analysts will need to draw upon existing valuation estimates for use in benefits analysis. The process of applying these estimates to value the consequences of policy actions is called *benefits transfer*. Although the benefit transfer method is detailed in only one section, this chapter is generally written with benefit transfer in mind. For example, the descriptions of valuation methods in Section 7.5 include recommendations for assessing the quality of published studies. This is done to help analysts determine which studies deserve consideration for use in benefit transfers.

While analysts should always seek precision, they must make assumptions and exercise professional judgment to face the challenges noted above, as well as numerous others that arise in a benefits analysis. Existing value estimates, for example, are often subject to large uncertainty bounds due to measurement error, model uncertainty, and the inherent variability of individual preferences. When drawing from these studies—and when using quantitative estimates of any kind—analysts should carefully assess the quality of the data and should clearly state the reasons for their analytical choices. As with any analytical exercise, the maxim "garbage in, garbage out" always applies.

The next section briefly summarizes the conceptual economic framework for benefits analysis. Section 7.3 outlines the effect-by-effect process for benefits analysis, including some general implementation principles. The fourth section defines and describes the types of benefits associated with environmental policies, followed by a review of available economic valuation methods in Section 7.5. This chapter

concludes with specific recommendations for valuing types of benefits that are common to many EPA policies.

7.2 A Conceptual Framework for Benefits Analysis

This section describes the theoretical economic foundation for valuing benefits. The theoretical discussion here serves as a conceptual starting point for benefits estimation—it is not a full and comprehensive treatment of welfare economics. The section includes a discussion of willingness to pay, consumer surplus, and analytical problems arising from the lack of markets for environmental improvements. References are provided for further reading on the specific topics introduced in this section, but useful texts for general reference include Just et al. (1982), Braden and Kolstad (1991), and Freeman (1993). Boardman et al. (1996), Brent (1995) and Hanley and Spash (1993) are useful, general references for benefit-cost analysis.

7.2.1 Welfare Measures: Willingness to Pay and Willingness to Accept Compensation

Economists define benefits by focusing on measures of individual satisfaction or well-being, referred to as measures of welfare or utility. Economic theory assumes that individuals can maintain the same level of utility while trading-off different "bundles" of goods, services, and money. For example, one may be equally satisfied by going fishing or viewing a movie. The tradeoffs individuals make reveal information about the value they place on these goods and services.

The willingness to trade off compensation for goods or services can be measured either as *willingness to pay* (WTP) or as *willingness to accept compensation* (WTA). Economists generally express WTP and WTA in monetary terms. In the case of an environmental policy, willingness to pay is the maximum amount of money an individual would voluntarily exchange to obtain an improvement (or avoid a decrement) in the environmental effects of concern. Conversely, willingness to accept compensation is the least amount of money an individual would accept to forego the improvement (or endure the decrement).[1]

- **WTP and WTA are not necessarily equal.** The amount an individual would be willing to pay to obtain an environmental improvement is not necessarily identical to the amount he or she would be willing to accept to forego the improvement. One reason for this difference is that the starting points of the two measures differ. For environmental improvements, WTP uses the level of utility without the improvement as a reference point. WTA, on the other hand, uses as its reference point the level of utility *with* the improvement. Although these two measures are distinct and sometimes differ in practice, under conventional assumptions economists expect that the difference between them will be small in most cases. This result generally holds as long as the amounts in question are a relatively small proportion of the individual's income. Nonetheless, in the case of environmental goods, some additional considerations modify this general result. Hanemann (1991) shows that while this result holds for price changes, it does not strictly hold for changes in quantity or quality. Also, if a good has no close substitutes, differences in WTP and WTA may be large even if the effect on income is small.

- **WTP and WTA can also be identified with what they imply about property rights**—whether entities have a right to pollute, so the public must pay them not to, or whether the public has a right to a clean environment and must be compensated for pollution. For example, in the case of a policy that would reduce existing pollution levels, the use of WTP measures to value benefits implicitly assumes that the property right rests with the polluting firm.

[1] In the case of environmental improvements, WTP is identified as the *compensating variation* measure of welfare change, while WTA in this case is identified as the *equivalent variation measure*. For environmental decrements, these associations are reversed. For a more detailed treatment of welfare measures that includes these issues see Just et al. (1982), Freeman (1993), and Hanley and Spash (1993).

In practice, WTP is generally used to value benefits because it is often easier to measure and estimate. To simplify the presentation, we use the term "willingness to pay" (or WTP) throughout this chapter to refer to the underlying economic principles behind both WTA and WTP.

Aggregating Individual Willingness to Pay Measures

The benefits of a policy are the sum total of each affected individual's WTP for the policy. Because benefit-cost analysis assesses only the efficiency of policy choices, each individual's WTP must be given the same weight in the summation. This means that no individual or group of individuals is given preferential treatment in assessing the efficiency of the program except to the degree that they are willing to pay for it. As described in Chapter 9, equity assessments and impact analyses can be used to describe the effects of policies on populations of concern.

Altruism

While benefits are generally calculated by summing each individual's WTP for his or her own welfare, there are conditions under which it is appropriate to include altruistic values, or individuals' WTP for the welfare of others. Economic theory concludes that if one cares about a neighbor but respects the neighbor's preferences, and if the neighbor would have to pay for the policy action being analyzed, then altruistic benefits should not be counted in a benefit-cost analysis. The intuition behind this result is that, if one respects the neighbor's preferences, one cares about both the benefits and the costs the neighbor faces. It is therefore inappropriate to add the value one attaches to the neighbor's benefits without considering the cost implications of doing so. Comparing individual benefits and costs in this case is the appropriate decision rule.

Altruistic benefits may be counted either when altruism toward one's neighbor is paternalistic or when one will in fact bear the costs of the project but the neighbor will not. In the first case (paternalistic altruism), one cares about the benefits the neighbor will enjoy, e.g., from a health or safety project, but not about the costs the project will impose on him. An example of the second case would be a project whose costs are borne entirely by the current generation; i.e., the project imposes no costs on future generations. In this case, altruism toward future generations by the current generation could legitimately be counted as a benefit.

7.2.2 Market Goods: Using Consumer Surplus and Demand Curves

Willingness to pay is closely related to the concept of *consumer surplus*, which is both an individual and an aggregate concept. An individual demand curve indicates the maximum amount an individual would be willing to pay to acquire an additional unit of good. These *individual* demand curves can then be aggregated into a *market* demand curve that provides the cumulative WTP for additional units. Consumer surplus is derived from market estimates of how much of the good is demanded in the aggregate at each price and can be easier to estimate than individual WTP.

A market demand curve for a given good or service traces out the amounts that consumers will purchase at different price levels; i.e., their collective WTP for the good or service. Consumer surplus is the excess amount that purchasers are willing to spend on a good or service over and above that required by the market price (i.e., the area under the demand curve but above the price line). This surplus serves as a measure of the social benefits of producing the good. Policies that affect market conditions in ways that decrease prices will generally increase consumer surplus. This increase can be used to measure the benefits of the policy.[2]

The use of demand curves and consumer surplus highlights the importance of assessing how individuals will respond to changes in market conditions. For example, if a policy affects the price or availability of a commodity traded in a market (e.g., if it leads to increases in the commercial fish harvest), multiplying the increased quantity by current prices generally will not provide an accurate

[2] Technically, consumer surplus serves as a precise measure of benefits only if the demand curve represents a *compensated* or *Hicksian*, demand function. However, Willig (1976) shows that *ordinary*, or *Marshallian*, demand curves can often be used to derive an approximate measure of welfare. The difference in these two types of demand curves is that the former holds utility constant, while the latter holds income constant. More background on the theoretic basis for welfare measures can be found in several texts including Freeman (1993), Johansson (1993), Just et al. (1982), and Varian (1992).

measure of benefits. Depending on the elasticity of the demand curve, a one percent price increase may lead to more (or less) than a one percent increase in the quantity demanded, affecting the change in consumer surplus.[3] While not detailed here, supply curves also vary in elasticity and have an analogous effect on producer surplus. Information on the elasticity of the supply and demand curves is needed to estimate benefits in the form of increases in consumer (and/or producer) surplus.[4]

7.2.3 Non-Market Goods

One challenge facing analysts of environmental policies is the lack of a market for most environmental improvements. Because "cleaner air" or "cleaner water" is not normally bought or sold, market data are generally not available for benefit valuation. Economists have therefore developed other methods for eliciting values for these types of effects. These methods rely either on information from the markets for related goods (revealed preference methods) or on direct information on people's preferences (stated preference methods). Individual WTP values estimated in these studies can be aggregated (or an average value multiplied by the total number of affected individuals) to produce an estimate of the total benefit for a good or policy. Section 7.5 provides more information on the economic foundations of specific methods, and Section 7.6 details how these methods have been—or can be—applied in benefits analysis.

7.3 The Benefits Analysis Process

From the perspective of economic theory, an appropriate measure of a policy's benefits is the sum of individual WTP estimates for that policy. While it may be possible in some circumstances to obtain individual WTP estimates for the entirety of a policy decision, in practice, analysts must often use an "effect-by-effect" approach for benefit valuation. This section discusses this approach to benefits analysis, concluding with some general principles to keep in mind when implementing this approach.

7.3.1 A General "Effect-by-Effect" Approach

The most widely used approach for estimating the benefits of a policy option is to evaluate separately the major effects of a given policy and then sum these individual measures to arrive at total benefits. This general approach usually involves describing the physical effects of the pollutants (e.g., various types of damages to human health and ecological systems) and assessing each type of effect separately. In some cases, it may be desirable and feasible to diverge from this approach. For example, contingent valuation or other methods could be used to develop estimates of WTP for the combined effects of the policy change, reducing the need to identify, quantify, and value each effect separately. A comprehensive value estimate for the entire set of effects from a policy change can also be useful as an indication of the upper bound expected from the sum of values developed with the effect-by-effect approach.[5] However, because it is difficult to develop estimates of the total value of the pollution reduction and decision makers are often interested in information on individual benefit categories, an effect-by-effect valuation approach is most often used by EPA in economic analyses of regulations.

The general effect-by-effect approach for assessing the benefits of environmental policies includes three components:

- **Identify potentially affected benefit categories** by developing an inventory of the physical effects that may be averted by the policies.

[3] Elasticity is a measure of relative change. For a given demand curve, price elasticity is defined as the percentage change in quantity demanded divided by the percentage change in price. Where this value is less than one in absolute value, demand is considered to be "inelastic." Elasticity values greater than one (in absolute value) indicate that demand is "elastic."

[4] It is important to keep in mind that elasticity is a local concept. Generally, one can expect the elasticity of supply and demand curves to vary along their respective lengths. This means that elasticities measured at a particular point on these curves may not be appropriate for estimating large changes or changes elsewhere on the curve. In these cases, it may be necessary to characterize the demand and supply functions in the relevant range of prices and quantities.

[5] Randall (1991) presents a framework for comparing total value and "independent valuation and summation" and reviews many issues associated with estimating total values.

- **Quantify significant physical effects** to the extent possible working with managers, risk assessors, ecologists, physical scientists, and other experts.
- **Estimate the values of these effects** using studies that focus on the effects of concern or transferring estimates from studies of similar impacts.

These steps may be implemented using an iterative process. For example, analysts can begin by conducting screening analyses using available data and relatively simple assumptions, then collect additional data and refine the analysis as needed to better inform decision-making.

Each step in this approach is discussed in more detail below, focusing on the actions that are generally undertaken when conducting benefits analyses for typical EPA policies. However, this guidance is intended to be flexible. Analysts will need to determine on a case-by-case basis whether this framework is appropriate for assessing a specific policy, given the effects particular to that policy and the information needed for related decision-making.

Step1: Identify Potentially Affected Benefit Categories

The first step in the benefits assessment is to determine the types of benefits most likely to be associated with the particular policy. Section 4 of this chapter contains a detailed presentation of the categories of benefits typically associated with environmental policies and regulations. To identify benefit categories, analysts should, to the extent feasible, do several things:

- **Develop an initial understanding of policy options of interest** by working with cost analysts and policy makers. Information should also be collected on the likely range of emissions levels associated with the baseline and with implementation of each of the policy options. At the outset of the analysis, the range of options and associated emissions levels considered may be very broad because emissions levels and preferred policy options can change significantly in the course of the policy making process.
- **Research the physical effects of the pollutants** on human health, welfare, and the environment. This can be done by reviewing the literature and, if necessary, meeting with other experts. This step requires considering the transport of the pollutant through the environment along a variety of pathways, including movement through the air, surface water and groundwater, deposition in soils, and ingestion or uptake by plants and animals (including humans). Along these pathways, the pollutant may have detrimental effects on natural resources (e.g., affecting oxygen availability in surface water or reducing crop yields) as well as direct or indirect effects on human health (e.g., affecting cancer incidence through direct inhalation or through ingestion of contaminated food).
- **Consider the potential change in these effects** as a result of possible policy options. If policy options differ only in their level of stringency, then each option may have an impact on all identified physical effects. In other cases, however, some effects may be reduced while others remain unchanged under a specific policy option. Evaluating how physical effects change under each policy option requires evaluation of how the pathways differ in the "post-policy" world.
- **Evaluate which effects are likely to be significant** in the overall benefit analysis according to at least three criteria:
  - whether there are likely to be observable changes in the benefits category when comparing the policy options to each other and to the baseline;
  - whether the benefits category is likely to account for a major proportion of the total benefits of the policy; and
  - whether stakeholders or decision makers are likely to need information on the benefits category, even if its magnitude is relatively small.[6]

The outcome of this initial step in the benefits analysis can be summarized in a list or matrix that describes the physical effects of the pollutant, identifies the benefits categories associated with these effects, and an initial ranking of which effects may be significant enough to warrant further investigation.

Initially, the list of benefit categories may be lengthy and include all effects that reasonably can be associated with

[6] This criteria relates to equity considerations detailed in Chapter 9.

the policy options under consideration. Analysts should preserve and refine this list of benefit categories as the analysis proceeds, and the effects that are not assessed in detail should be discussed qualitatively when presenting analytic results. In some cases, it may not be feasible to assess some of the more significant impacts, either because of insufficient scientific data (e.g., data are lacking on the effects of changes in pollution levels on the benefit category of concern) or because the time or resources needed to assess the effect are high compared to the significance of the benefits category in the decision-making process. These issues should be discussed when presenting the results of the benefits analysis. The discussion should address (1) the criteria used to exclude selected benefit categories from detailed quantitative analysis, (2) the likely magnitude of the non-quantified benefits, and (3) the extent to which these effects are or are not important considerations for the decision-making process.

Step 2: Quantify Significant Physical Effects

The second step is to quantify the physical impacts related to each category. Data are usually needed on the extent, timing, age distribution of the affected population, and severity of the effects. The focus should be on the changes attributable to each policy option in comparison to the baseline. For example, if the risk of lung cancer is one of the effects of concern, data may be needed on the changes in risk associated with each option, the timing of the risk reductions, the age distribution of those experiencing the risk reductions, and the percentage of cases likely to be fatal. If visibility is a concern, data may be needed on the geographical areas affected and the change in visibility levels attributable to each policy option.

- **Work closely with analysts in other fields.** Estimating these impacts is largely, but not completely, the domain of other scientists, including risk assessors, ecologists, and other experts. These experts are generally responsible for evaluating the likely transport of the pollutant through the environment and its potential effects on humans, ecological systems, and manufactured materials under the baseline and each policy option. The principal role of the economist is to communicate with these experts in order to ensure that the information provided is adequate to support the benefits analysis, including information on the uncertainty associated with the estimates of physical impacts. However, economists may also be able to provide insights, information, and analysis on behavioral changes that can affect the results of the risk assessment.

- **Try to match the risk assessment and economic endpoints.** A key consideration in this interaction is that the endpoints quantified and described in the risk assessment match well the effects for which economic valuation is feasible. Effects that are described too broadly or that cannot be associated with economic welfare will limit the ability of the analysis to capture the full range of benefits associated with policy options. It is difficult, for example, to produce an economic measure of the benefits associated with a reduction in the number of persons exposed to a contaminant at a particular level. If, however, the risk assessment can produce an estimate of the reduction in the number and type of adverse health effects from exposure, then the economic valuation exercise is much more feasible. This means that the analyst must be aware of the available economic data and tools when working with risk assessors and other scientists.

- **Describe qualitatively effects that cannot be quantified.** It will not be possible to quantify all of the significant physical impacts for all policies. For example, animal studies may suggest that a contaminant causes severe illnesses in humans, but the data available may not be adequate to determine the number of expected cases associated with different human exposure levels. Likewise, it is often not possible to quantify all the ways in which an ecosystem may change as a result of an environmental policy. In these situations, the effect should be described qualitatively when presenting the results of the benefits analysis. Analysts should also assess the implications of not being able to include this effect in quantitative benefits estimates.

EPA has developed extensive guidance on the assessment of human health and ecological risks and analysts should refer to those documents and the offices responsible for their production and implementation for further guidance.[7] No specific guidance exists for assessing changes in materials damages or amenity effects. Analysts should consult relevant experts and existing literature to determine the "best practices" appropriate for this type of analysis.

Step 3: Estimate the Values of the Effects

Once information on the physical effects of the pollutant is available, the next step is to assess the value of related benefits based on estimates of individual WTP. As discussed earlier, no market exists for many of the types of benefits anticipated from environmental regulation. In most cases, analysts will need to rely upon the results of other methods for estimating economic values. Details on these methods and examples of how they may be applied can be found in Sections 7.5 and 7.6, respectively.

- **Consider using more than one method to estimate benefits.** Different methods often address different subsets of total benefits and the use of multiple methods allows for comparison of alternative measure of value. Double-counting is a significant concern when applying more than one method, however, and any overlap should be noted in presenting the results. In addition, some components of the total value of benefits may not be amenable to valuation and will need to be described in other terms when presenting the analytic results. The discussion of benefit transfer in Section 7.5 describes many of the issues involved in applying values from one study to another situation.

- **Describe the source of estimates and confidence in those sources.** Valuation estimates always contain a degree of uncertainty. Using them in a context other than the one in which they were initially estimated can only increase that uncertainty. If many high-quality studies of the same effect have produced comparable values, analysts can have more confidence in using these estimates in their benefits calculations. Some specific benefit transfer methods described in Section 7.5 provide a systematic manner of combining multiple estimates. In other cases, analysts may have only a single study—or even no directly comparable study—to draw from. In all cases, the presentation of the benefits analysis should clearly describe the sources of any values used, along with some assessment of the confidence associated with those sources.

7.3.2 Implementation Principles

When applying this framework to assess the benefits of specific policies, analysts should keep in mind the following general principles:

- **Focus on key issues.** Resources should be focused on benefit categories that are likely to influence policy decisions. To use time and resources effectively, analysts must weigh the costs of conducting additional analysis against the usefulness of the additional information provided for decision-making. The analysis should devote significant time and resources to carefully assessing those benefits categories that are likely to influence the selection among policy options. In some cases, relatively simple screening analyses may provide adequate information on these benefits. Additional data collection may not be warranted because it is unlikely to lead to significant changes in the conclusions of the analysis. For example, screening using a broad range of values for selected effects may indicate that a policy is clearly worth pursuing and analysts may conclude that any possible refinements to the analysis are likely to simply reinforce this conclusion. In this case, the analyst should discuss the approach taken and note that the benefits estimate may represent a lower bound. Likewise, some categories of benefits may not be assessed either because they are expected to be small or because the costs or time needed to quantify them far exceed the time or resource levels appropriate for analysis of the particular policy.

  Applying this approach to benefits assessment involves first conducting scoping analyses to collect available information on the potential benefits of the policies and using this information to develop

[7] In September 1986, EPA published final risk assessment guidelines for a number of health effects, including *Guidelines for Carcinogen Risk Assessment,* which are currently under revision. Many other risk-related guidelines have been published, revised, and updated since 1986. Recent additions include *Guidelines for Exposure Assessment* (EPA, 1992) and *Guidelines for Reproductive Toxicity Risk Assessment* (EPA, 1996). More information on these and other guidelines, as well as electronic copies of the documents themselves, can be found on the home page of EPA's National Center for Environmental Assessment at http://www.epa.gov/nceawww1/raf/rafguid.htm (accessed 8/28/2000).

preliminary estimates (see, for example, Morgan and Henrion,1990). The results from this initial screening analysis can then be used to inform the early stages of the policy development process and to focus future research on those areas most in need of further assessment. In many cases, it may be useful to use benefits transfer techniques in the initial stages of the analysis, as discussed later in this chapter.

- **Coordinate frequently with others involved in developing the policies.** Ongoing coordination with the analysts responsible for assessing costs and economic impacts, and with the work group considering policy options is crucial to ensure consistency as the policy options and analyses evolve. This coordination should begin in the planning stages of the analysis, and should continue throughout the development process. Successful efforts often involve informal conversations among lead analysts several times each week, supplemented by larger and more formal periodic meetings to report on progress and discuss next steps.

  Coordination will help ensure that the cost and benefit results are comparable and based on consistent baseline and policy assumptions. In addition, information from the cost analysis is often needed for the analysis of benefits and vice versa. For example, if a policy requires firms to install new pollution controls, benefits analysis requires information on the number of facilities likely to install each type of control and the associated reduction in emissions. On the other hand, where a performance standard is being considered, the cost analysis may need data from the risk models in considering which controls are likely to meet the standard.

- **Consider changes in behavior.** The use of an effect-by-effect approach does not necessarily mean that one should simply value benefits by estimating the physical changes attributable to changes in pollution emission levels (e.g., increases in the fish population) then assigning a unit value to these changes (e.g., the price of the fish). Such a limited analysis will be inappropriate in many cases because it leaves out the effects of changes in behavior attributable to changes in environmental quality. For example, increased fish populations may cause commercial prices to drop, in which case consumers may increase their purchases. Commercial fisheries may also respond to changes in pollution levels by altering their production processes. While it may not be possible in practice to capture all of these types of responses in the analysis, those that are likely to be significant should be addressed.

- **Guard against *double-counting* benefits.** If there is significant overlap across the values used for estimating the benefits of different effects, summing values across these effects could substantially overstate expected benefits. For example, property value studies may estimate people's WTP for all perceived effects. This would overlap with values estimated separately for any one of these effects, such as reduced risk, so simply adding these two values to estimate benefits would be inappropriate. Analysts should also take care to ensure that important effects of the policy have not been omitted in the benefits analysis, as this will lead to significant underestimates of total benefits.

- **Explicitly address uncertainty and non-monetized effects.** Benefits assessments for environmental policies often involve significant uncertainty. Sometimes this uncertainty cannot be reduced (or better characterized) given the need to regulate in a timely manner and the resources available for the analysis. These uncertainties should be clearly communicated when presenting the results of the analysis, focusing on the implications for decision-making. For example, if benefits may be significantly overstated due to the conservatism inherent in the risk estimates, then the materials summarizing the analysis should state this explicitly. Guiding principles for addressing and presenting uncertainty are presented in Chapter 5 of this guidance. The relative significance of benefits categories that are not quantified, or quantified but not monetized, should also be described, as discussed in Chapter 10.

7.4 Types of Benefits Associated with Environmental Policies

This section describes the types of benefits that are typically associated with environmental policies. These descriptions are provided with an understanding that it is desirable to quantify and monetize these benefits. Available valuation techniques are described in Section 7.5.

Benefits from environmental policies can be broadly classified into those that directly affect humans and human welfare and those that affect human welfare through systems or processes. The former category includes human health improvements such as reduced mortality rates, decreased incidence of nonfatal cancers, chronic conditions and other illnesses, and reduced adverse reproductive or developmental effects. Improved amenities are another type of benefit experienced directly by humans. Improved taste and odor of tap water resulting from treatment requirements are an example of direct amenity benefits.

Benefits that affect human welfare through systems or processes include reduced materials damages and numerous other effects collectively termed ecological benefits. EPA policies may result in ecological impacts that affect the human use of natural resources (e.g., improving commercial fishing, increasing agricultural yields, enhancing recreational opportunities.) Ecological effects may also provide passive use (or "non-use") benefits that arise from a variety of motives including, for example, one's own utility in knowing that clean resources exist or the desire to preserve clean resources for future generations. In some cases, environmental policies also reduce damages to manufactured materials or improve a resource's aesthetic qualities. Reducing air pollution may decrease damages to building exteriors or improve visibility. Exhibit 7-1 illustrates this categorization scheme

Exhibit 7-1 Examples of Benefit Categories, Service Flows, and Commonly-Used Valuation Methods

Benefit Category	Examples of Service Flows	Commonly-Used Valuation Methods
Human Health		
Mortality Risks	Reduced risk of • Cancer fatality • Acute fatality	• Averting behaviors • Hedonics • Stated preference
Morbidity Risks	Reduced risk of • Cancer • Asthma • Nausea	• Averting behaviors • Cost of illness • Hedonics • Stated preference
Amenities	• Taste • Odor • Visibility	• Averting behaviors • Hedonics • Stated preference
Ecological Benefits		
Market: products	Provision of • Food • Fuel • Fiber • Market • Timber • Fur, eather	• Market
Non-market: recreation and aesthetics	Provision of • Recreational opportunities, e.g., viewing, fishing, boating, swimming, hiking • Scenic vistas	• Production function • Averting behaviors • Hedonics • Recreation demand • Stated preference
Indirect: 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 • Pest control	• Production function • Averting behaviors • Stated preference
Non-use: existence and bequest values	No associated services	• Stated preference
Materials Damage	--	• Averting behaviors • Market

and suggests commonly-used techniques for estimating their values, although the list is not exhaustive.[8] A detailed discussion of valuation techniques is presented in the next section of this chapter. The remainder of this section describes each of these categories briefly and notes issues associated with quantification.

7.4.1 Human Health: Mortality Risks

Some EPA policies are designed to decrease the risks of contracting potentially fatal health effects, such as some cancers. Reducing these risks of premature fatality provides welfare increases to those individuals affected by the policy. It is important to keep in mind that policies generally provide marginal changes in relatively small risks. That is, most policies do not provide assurance that one will not prematurely die of environmental causes, they only marginally reduce the probability of such an event.

- **Reduced mortality risks are often measured in terms of "statistical lives."** This measure is the aggregation of many small risks over an exposed population. Suppose, for example, that a policy affects 100,000 people and reduces the risk of premature mortality by one in 10,000 for each individual. Summing these individual risk reductions across the entire affected population results in the policy saving 10 statistical lives. It is unknown who these ten people might be—everyone faces some risk of being affected—but the policy can be expected to prevent premature fatality for 10 individuals in the population.

- **Alternative measurements may include "statistical life years."** A somewhat more refined approach to measuring reduced mortality risks includes the degree of life extension in the estimate. This is usually done by looking not just at the reduced probability of a premature fatality, but also at the expected life span of those enjoying the risk reduction. A risk reduction of one in 10,000 experienced by a population of 100,000 people with an expected remaining life span of 50 years each, for example, would save 10 "statistical lives" or 500 "statistical life years." Measuring mortality risk reduction in terms of statistical life years provides more information about the expected benefits of a policy, but requires risk estimates for specific age groups.[9] Often these risk estimates are not available.

7.4.2 Human Health: Morbidity Effects

This benefits category consists of reductions in the risk of non-fatal health effects ranging from mild illnesses such as headache and nausea to very serious illnesses such as cancer. A complete list of morbidity effects is beyond the scope of this document, but the presumption for all of these effects is that the illness will not generally result in premature fatality.

- **Morbidity effects can generally be characterized by their duration and severity.** For duration of illness, the primary distinction is between acute effects and chronic effects. Acute effects are discrete episodes usually lasting only a few days, while chronic effects last much longer and are generally associated with long-term illness. Severity defines the degree of impairment associated with the illness and may be measured in terms of "restricted activity days," "bed disability days," or "lost work days."[10] Severity may also be described in terms of health state indices that may combine multiple dimensions of health into a single quantity, or index. The difference in the index

[8] This classification scheme is offered here to facilitate discussion in this document. It is similar in many respects to one offered in Freeman (1993), but other researchers have offered alternatives. Freeman (1993) describes some general characteristics of these alternatives. The list of techniques for each benefit category is not intended to be comprehensive or exclusive

[9] Additional refinements to account for quality of life or health status are often employed in the public health and health economics. Existing measures include "quality adjusted life years" (QALYs) and "disability adjusted life years." These measures have not been fully integrated with the literature on benefits analysis for environmental policies. More information on QALYs can be found in Gold et al. (1996) and additional information on DALYs can be found in Murray (1994).

[10] 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 requires 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.

value reflects the relative difference in disutility associated with symptoms or illnesses.[11] Morbidity effects can be further characterized by the set of symptoms associated with an illness.

- **Morbidity effects are usually quantified in terms of the number of expected cases of a particular illness.** Given the risks faced by each individual and the number of persons exposed to this risk, an estimate of "statistical cases" can be defined analogously to "statistical lives" described above. Alternatively, morbidity effects may be described according to the expected number and duration of particular symptoms associated with the illness. These estimates of "symptom days" may be used in benefits analysis when appropriate estimates of economic value are available.

7.4.3 Amenities

Direct amenities include improvements in aesthetic attributes associated with environmental commodities. This includes improvements in taste, odor, appearance, or visibility. In short, these benefits are determined by how the senses are affected and how individual's welfare is changed as a result. This class of benefits is unique in that the focus is on the sensory experience and not on a physical or material effect.

Despite this conceptual distinction, aesthetic benefits are often intertwined with other benefit categories such as health and recreation. A policy that improves air quality, for example, might simultaneously improve visibility and reduce mortality risks associated with airborne contaminants. New treatments for drinking water might reduce health risks as well as alter the taste and odor of tap water. These relationships may make it extremely difficult to separately quantify and value improvements in aesthetic qualities.

Many types of policies can be expected to have some impact on these kinds of amenities and they may be the focus of a given policy. Amenity improvements may be major component of total expected benefits. Improved visibility from better air quality is one example that has been the subject of several empirical studies.[12]

7.4.4 Ecological Benefits

Ecosystems provide services that benefit humans. For example, a freshwater lake may provide recreational and boating sites; a wetland provides a service by being a breeding ground for fish and fowl. Although ecosystems have a profound impact upon human well-being, the quantitative assessment of ecological benefits presents a formidable challenge for several reasons. First, natural systems are inherently complex. The many services they provide and how they provide them may be poorly understood by even the scientific community. Second, ecological risks vary widely in terms of persistence (e.g., eutrophication versus species extinction), geographic extent (e.g., toxic contamination versus global climate change), and the degree to which the overall threat can be predicted (e.g., effects of ozone on crops versus developmental and behavioral effects of chemicals on wildlife populations). Third, many of the less tangible benefits are not readily amenable to monetary valuation.

Section 7.3 discussed generally the three steps involved in assessing the benefits of environmental policies. However, some issues associated with identifying and quantifying ecological benefits are particularly complex and warrant more detailed treatment.

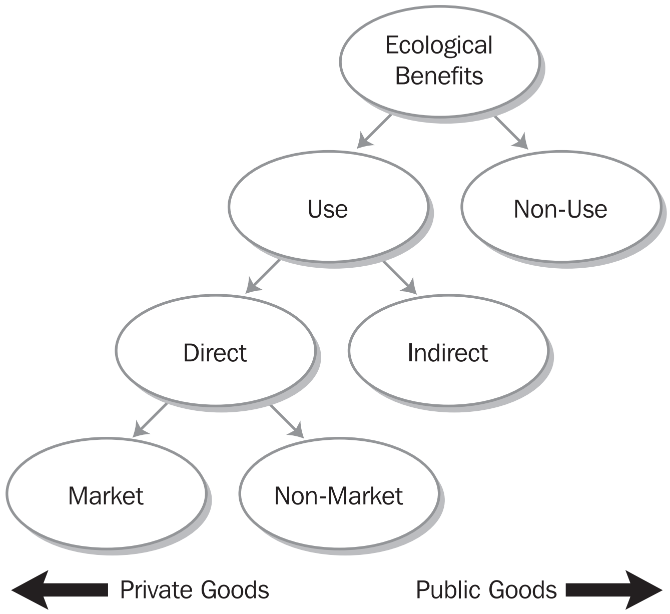
Identifying Ecological Benefits

The first step in assessing ecological benefits is to identify those relevant to policy options under consideration, focusing on service flows that are likely to change as a consequence of guidance or regulatory action. In general, these ecological benefits may be thought of as flows of services from the natural asset in question. These can be categorized by how directly they are experienced and where they fall along a private good/public good continuum. Exhibit 7-2 illustrates how the categories relate to one another. Not only is it useful as a conceptual tool, this

[11] These indices may be constructed in a number of ways, but consistency with welfare economics requires affected individuals to define these relative tradeoffs for themselves rather than having them determined by health experts. Several economic analyses have employed some form of health state index. Recent examples include Desvousges et al. (1998) and Magat et al. (1996).

[12] Examples of these studies include Rae (1983), Johnson et. al. (1983), Schulze et al. (1983), Chestnut and Rowe (1990), Crocker and Shogren (1991), and McClelland et al. (1993).

Exhibit 7-2 Ecological Benefits Classification Scheme



categorization helps direct analysts to suitable valuation methods.[13]

- **Market benefits:** Direct market benefits are some of the most readily identified service flows provided by ecosystems. These typically relate to primary products that can be bought and sold competitively as factors of production or final consumption products. Although they may be managed to a high degree, agricultural systems are nevertheless predicated on ecological processes. As a consequence, increased productivity of farmland and rangeland may provide significant market benefits. Other products include commercial fish species and timber. When access is controlled and appropriate user charges levied, recreational opportunities may also be considered direct, market benefits.

- **Non-market benefits:** Recreational opportunities and aesthetic qualities provided by ecosystems are also experienced directly by individuals, albeit in a non-market setting. Non-market benefits include both consumptive uses (e.g., recreational fishing and hunting) and non-consumptive uses (e.g., scenic vistas, wildlife viewing, hiking, and boating). These services are typically provided by natural assets held in common (e.g., public lands). They have public goods characteristics—since access is not or cannot be controlled, consumption is not exclusive. On the other hand, like private goods, they are rival in consumption because excessive use by others (i.e., congestion) tends to diminish one's own enjoyment of these services.

- **Indirect benefits:** Ecosystem services that do not directly provide some good or opportunity to individuals may be valued because they support off-site ecological resources or maintain the biological and biochemical processes required for life support. These indirect benefits tend to be purely public in nature—access to or use of the service is not exclusive and a virtually unlimited number of individuals can share in the benefits without reducing the average benefit accruing to each. Each type of ecosystem provides various indirect benefits. Wetlands recharge groundwater, mitigate flooding, and trap sediments. Forests sequester carbon, anchor soil, and maintain microclimates. Estuaries protect adolescent fish. Terrestrial ecosystems provide habitat for natural pollinators. All of these systems support biodiversity.

- **Non-use benefits:** Some benefits are not associated with any direct use by either individuals or mankind.

[13] A more detailed discussion of these concepts is also found in EPA's Conceptual Framework for Assessing Ecological Costs or Benefits (EPA, 1999b). A draft is available at http://intranet.epa.gov/oerrinet/ecoweb/index2.htm (accessed 8/29/2000).

Rather, they result because individuals might value an ecological resource without using or even intending to use it. Non-use values, also referred to as passive use values, are those associated with the knowledge the resource exists in an improved state, bequest values for future generations and altruistic values for others' enjoyment of the resource. An individual's commitment to environmental stewardship may also be the source of existence value. The commitment of some groups to particular animals or ecosystems provides an example of this.[14]

Quantifying Ecological Risk

The second step in the analysis of ecological benefits is to estimate the physical effects of each policy option, comparing the flow of services with and without the policy. It falls upon ecologists and environmental toxicologists to conduct the ecological risk assessments to estimate the expected adverse ecological effect of a particular stressor.[15] Ecological risk assessments can be either narrow in scope, with inquiry limited to a single species or population (e.g., the effect of chemical exposure on an endangered bird species) or focus broadly on an entire ecosystem. Further information on ecological risk assessment can be found in *Ecological Risk Assessment Guidelines* (EPA, 1998).

The results of an ecological risk assessment generally include the effect's magnitude (expressed in such metrics as hazard quotients or percent change in population), duration, spatial distribution, and time period of recovery. The analysis of ecological risks may be highly uncertain. Limited availability of data and models, and imperfect understanding of key issues, hampers our ability to describe ecological effects.

7.4.5 Reduced Materials Damages

The materials damages benefit category includes welfare impacts that arise from changes in the provision of service flows from the "material" environment. The "material" environment is distinguished from the natural environment discussed in the ecological benefits section and includes constructed or highly-managed physical systems. Changes in the stock and quality of these material environmental resources are assessed in a similar fashion to their natural environment counterparts. Analytically, benefits assessment for materials improvements parallels that for managed ecosystems such as agriculture or forestry, with most benefits arising from direct, market effects or use values. For example, effects from changes in air quality on the provision of the service flows from physical resources such as buildings, bridges, or roads are handled in a similar fashion to the effects from changes in air quality on crops or commercial timber stocks. The most common empirical applications involve air pollution damages and the soiling of structures and other property.

7.5 Methods for Benefits Valuation

Economists have developed a number of methodologies to measure the benefits of environmental improvements.

- **Market methods** can be used when direct markets for environmental goods and services exist. The benefits of a change in quantity of a good are estimated using data on these market transactions. By knowing how the good was bought and sold, economists can infer directly how people appear to value that good.

  Unfortunately, direct markets for environmental goods and services do not often exist. In the absence of these markets, environmental and natural resource economists must rely upon alternative methodologies to measure the benefits of environmental improvements.

- **Revealed preference methods** (or *indirect approaches*) allow economists to infer the value placed on environmental goods using data on actual choices made by individuals in related markets. Revealed preference methods include recreational demand models, hedonic wage and hedonic property models, and averting behavior models.

[14] Even though it does not involve use, non-use value still falls under the rubric of welfare economics. It emanates from human interest, alone, and does not encompass any rights or ethics-based justification for preservation (see Kopp, 1992 and Mazzotta and Kline, 1995).

[15] Other types of frameworks for ecological assessment include injury assessments undertaken as part of natural resource damage assessments (IEC, 1995 and Huguenin et al., 1996) and environmental assessments undertaken to meet the requirements of NEPA.

- **Stated preference methods** (or *direct approaches*) allow economists to estimate the value placed on environmental goods using data on hypothetical choices made by individuals responding to a survey. Stated preference methods include contingent valuation, conjoint analysis, and contingent ranking.

Specific approaches that fall under these two broad categories are presented below. This presentation includes an overview of each method, a description of its general application to environmental benefits assessment, and a discussion of issues involved in interpreting and understanding studies using the method. This information is primarily designed to help analysts evaluate existing studies being considered for benefit transfer, but it can also assist analysts in assessing the feasibility of employing these methods. The discussion below concludes with a separate overview of benefit transfer methodology in general. It is important to keep in mind that research on all of these methods is ongoing, sometimes at a rapid pace. The limitations and qualifications described here are meant to characterize the state of the science at the time these guidelines are published. Analysts should consult additional resources as they become available.

7.5.1 Market Methods

Economic Foundation of Market Methods

Market methods are used to value environmental goods and services that are directly traded as market commodities. Market methods are used, for example, to examine the effects of air quality improvements on agriculture and commercial timber industries and the effects of water quality improvements on commercial fisheries.

Market methods apply when environmental goods are factor inputs. Changes in the quality or stock of an environmental good can affect production costs, which can then alter the price and quantity of output and the returns to other factor inputs. In turn, these market responses affect the decisions and welfare of consumers and producers. Changes in the prices of marketed goods consumers face and changes in the income of the owners of the factor inputs reveal information about the welfare of consumers and producers. For example, the benefits of an environmental improvement are often realized as increases in consumer and producer surplus that arise from lower costs and prices and increases in the quantity of the marketed good. For more detailed discussion of the economic foundation of market methods, see Just et al.(1982) or Freeman (1993).

General Application to Benefits Assessment

When applying market methods to assess the benefits of environmental improvements, two types of market responses are important: the impacts of the environmental change on the relevant marketed good (e.g., factor) and the response of producers and consumers to this change. When examining these responses, it is important to consider the range of market responses available to producers and consumers. Overlooking market adjustments can bias benefits assessment. For instance, the damage function approach, which derives benefits by applying a unit price to a physical measure of damage or loss, ignores consumer responses to market adjustments.[16] Measures of price-elasticities, cross-price elasticities, and substitution possibilities indicate the extent to which market adjustments are likely to occur.

In practice, characterizing the market response to a change in environmental quality can be difficult. Two techniques that rely on observations of direct market behavior, cost and production function approaches, facilitate the measurement of consumer and producer surplus changes, but one must assume optimizing behavior on the part of producers and consumers. A different approach for benefits assessment is to use optimization models that simulate behavior. All three of these approaches require considerable information and data on the relevant market participants.

Benefits estimation using market methods varies with the types of markets affected by the environmental improvement. The nature of firms affected on the producer side (e.g., single-product firms or multi-product firms), the market structure (e.g., vertically linked markets), and the presence of market distortions (e.g., monopoly power, price supports) influence the complexity of benefits assessment. Freeman (1993) singles out two cases where

[16] Although the damage function approach does not account for market adjustments, it may be a useful screening tool when time and resources are limited.

benefits assessment is relatively straightforward. The first case is one in which the environmental good or quality is a perfect substitute for another input. Here, the benefits of an environmental improvement can be calculated by estimating the reduction in input costs caused by substituting away from the other input, as long as the change in total costs does not affect marginal costs or output. The second case is where observable market data (e.g., cost, demand, and market structure) imply that benefits from an environmental improvement will accrue to owners of fixed factors. Here, benefits can take the form of increased productivity and are realized as profit or quasi-rents. One such case would be where the producer affected by the environmental improvement is small relative to the market, and variable prices for factors and products are not affected by the environmental improvement.

Empirical applications of market methods are diverse. Among other topics, the empirical literature has addressed the effects of air quality changes on agriculture and commercial timber industries. It has also assessed the effects of water quality changes on water supply treatment costs and on the production costs of industry processors, irrigation operations, and commercial fisheries. Refer to Adams et al. (1986), Kopp and Krupnick (1987), Taylor (1993), and EPA (1997) for empirical examples.

Considerations in evaluating and understanding market studies

Issues to consider when interpreting the results of market studies include:

- **Data requirements and implications:** Employing market methods requires information on the effect of the environmental resource on production costs, supply conditions for output, demand curve for final good, and factor supplies.

- **The model for estimation:** Data availability plays a large role in the selection of a modeling approach and the structure of the model. Production function, cost function, and simulation optimization models are all options for understanding the market response to environmental improvements.

7.5.2 Revealed Preference Methods

In the absence of market data on the value of environmental improvements, WTP may be estimated by looking at related goods that are traded in markets. Methods that employ this general approach are referred to as "revealed preference" methods because people's behavior in associated markets reveals the value they place on the environmental improvements. For example, if pollution levels affect the use of a lake for recreational fishing, individual WTP to travel to a substitute site can be used to estimate the value of averting the damages to the lake of concern. Four distinct revealed preference methods have been widely used by economists: recreation demand models (including travel cost and discrete choice models), hedonic pricing models, averting behavior models, and cost-of-illness studies.

7.5.2.1 Recreation Demand Models

Improvements in environmental quality may enhance recreation opportunities at one or more sites in a region. For example, policies that control the level of toxics in surface water bodies might result in a reduction in the number of lakes and streams subject to fish consumption advisories, thereby enhancing recreational angling opportunities. Recreation improvements constitute a potentially large class of environmental benefits, but measurement of these values is complicated by the fact that access to recreation activities are only partially regulated by observable market mechanisms. Recreation demand models, including the travel cost model, the random utility model (RUM), and other approaches, may be used to assess nonmarket benefits associated with recreation activities.

Economic Foundation of Recreation Demand Models

Recreation demand models focus on the choice of trips or visits to sites for recreational purposes. The basic trade off to be considered is between the satisfaction gained from participating in an activity at a site and the value of money and time given up. The fundamental assumption is that people may weigh the money and time costs of travel to a site in the same way as an admission fee. Thus, by examining the patterns of travel to particular sites, one may infer how individuals value the site or particular aspects of the site such as environmental quality.

As with other economic studies, recreation demand models rely on individual perceptions. While it is possible to value changes in environmental quality that have an obvious effect on popular recreation activities, recreation demand methods may not be appropriate for valuing changes in environmental quality that are difficult for people to observe or only indirectly affect well-known species.

- **Travel cost models:** The simplest recreation demand model involves trips to a single site. User surveys provide data on visitors and trip origins and the data is organized by distance to the site. Generally, an inverse relation between distance traveled and the number of visits emerges. The distance variable may be converted to cost by including factors for the dollar per mile cost of vehicle travel as well as the cost of travel times and, the relationship among the variables may be interpreted as a demand function, with the number of trips from a particular area as a function of the travel costs of reaching the site.

  The single-site travel cost model may be extended to multiple sites, usually by estimating a system of demand equations, with the number of trips to a given site taken to be a function of the cost of visiting that site as well as the costs of visiting other available sites. A number of extensions to the simple travel cost model are described in Freeman (1993).

  Travel cost models are most appropriate for estimating changes in the number of trips over a given period of time, also known as participation. They are limited, however, in their ability to model the recreationist's choice among competing sites. A separate but related body of literature has developed around models that directly address the decision of "where to go" and estimate welfare changes associated with this alternative theoretical framework.

- **Discrete choice models:** For analyses focusing on the role of environmental quality variables, changes in social welfare may best be estimated through discrete choice models (also referred to as RUMs). Discrete choice models focus on the decision to recreate at a specific site as compared to alternative substitute sites. The model considers travel cost and environmental quality variables associated with all competing sites. Detailed treatments of the discrete choice model include Bockstael et al. (1986) and Bockstael et al. (1991).

  Although well suited for analyzing welfare effects of changes in site quality per visit, the discrete choice model is less useful for predicting the number of trips over a period and measuring seasonal welfare changes. Most recreation demand studies use either variations on the discrete choice model or combinations of travel cost and discrete choice approaches to estimate changes in social welfare.

Considerations in Evaluating and Understanding Recreation Demand Studies

There are several issues that must be confronted in a recreation demand model:

- **Definition of a site:** Ideally, one could estimate a recreation demand model in which sites are defined as specific points, such as launch ramps, campsites, etc., but the data requirements of detailed models are large. Similarly, for a given site, the range of alternative sites may vary by individual. Ultimately, every recreation demand study strikes a compromise in defining sites, balancing data needs and availability, costs, and time.

- **Opportunity cost of time:** Part of the cost of taking a recreation trip is the value of recreation time, which varies with respondent's income and work schedules. Recreation demand models typically use some fraction of the wage rate in calculating travel costs, but the tradeoffs between work hours and leisure time involve complex theoretical and methodological issues. Furthermore, it is presupposed that travel time detracts from the overall satisfaction of a recreation trip, but this assumption may not always hold. Other time-related issues include the treatment of on-site time, which varies from case to case but is often ignored altogether.

- **Multiple site or multipurpose trips:** Recreation demand models assume that the particular recreation activity being studied is the sole purpose for a given trip. Visits to multiple sites or multipurpose trips confound attempts to measure social welfare changes.

7.5.2.2 Hedonic Wage Studies (Wage-Risk studies)

Hedonic wage studies draw on the framework of hedonic pricing methods. This section describes the hedonic wage method, but first provides some general background on hedonic pricing methods in general. Property value studies, another large area of research based upon this framework, is described in Section 7.5.2.3.

Background on Hedonic Pricing Methods in General

Hedonic pricing methods apply to heterogeneous goods and services. Heterogeneous goods and services consist of "bundles" of attributes and are differentiated from each other by the quantity and quality of these attributes. Job opportunities, housing units, computers, and cars are common examples of heterogeneous goods. Hedonic pricing methods explain variations in price using information on attributes. For example, determinants of wages are expected to include worker characteristics (e.g., level of education, tenure, age) and job characteristics (e.g., risk of fatal injury). Determinants of housing prices may include structural attributes (e.g., number of bedrooms and age of house), neighborhood attributes (e.g., population demographics, crime, and school quality), and environmental attributes (e.g., air quality and proximity to hazardous waste sites).

The economic theory underlying hedonic pricing methods extends from a model of market equilibrium, where suppliers and demanders of heterogeneous goods interact under conditions of perfect information and zero transactions costs. Consumers derive utility from the attributes of the heterogeneous goods and adjust purchases in response to differences in these attributes. Producers or sellers of goods and services incur costs that vary with the range of attributes offered. An equilibrium price schedule develops from the market interactions of consumers and suppliers. The foundation of the hedonic pricing method as it relates to job opportunities is analogous, with workers and employers interacting in the labor market. The equilibrium price schedule is termed the hedonic price function and forms the basis for benefits assessment using hedonic pricing methods. Rosen (1974) is the seminal article on the economic theory of hedonic methods.

Empirical hedonic pricing research typically concentrates on the hedonic price function and the decisions of consumers or workers. The hedonic price function is approximated by regressing price on measures of attributes and the estimated coefficients represent the marginal WTP for the associated attribute. Applications of hedonic methods to labor wages and property values have been used to characterize the benefits of environmental improvements.[17] These are known as hedonic wage studies and hedonic property value studies, respectively. Each is considered separately below.

Economic Foundation of Hedonic Wage Studies (Wage-Risk Studies)

Hedonic wage studies, sometimes known as wage-risk or compensating wage studies, are based on the premise that individuals make tradeoffs between higher wages and increased occupational risks of death or injury. Essentially, higher risk jobs are expected to pay higher wages, all else held constant. Hedonic wage studies use statistical regression and data from labor markets to isolate the increment in wages associated with higher job risks. The outcome of these models is an estimated value of small changes in mortality risks. Some models also attempt to estimate the value of small changes in morbidity, or non-fatal risks.

The key to an effective hedonic wage study lies in separating the portion of compensation associated with occupational health risks from other job characteristics, including supervisory responsibility, job security, and similar factors. The wage rate is also affected by the industry in which the individual is employed, characteristics of the location and the personal characteristics of the workers (e.g., age, education, experience). All of these data are needed to disentangle the effects of worker characteristics from those of job attributes in determining wages paid.

In hedonic wage studies, workers' perceptions of risk levels across jobs are assumed to match actual risk levels. If perceived risks do not match actual risks faced by the workers, then the resulting estimates of compensation required to accept additional risk will be biased. Most

[17] Palmquist (1991) and Freeman (1993) contain current discussions of the use of hedonic methods for characterizing the demand for environmental quality and benefits assessment. See Palmquist (1982) for a discussion of a benefits assessment method related to hedonic property value studies known as repeat sales analysis.

analysts believe that this potential bias is small, but others argue that workers generally underestimate on-the-job risks. If the latter is true, hedonic wage studies will understate the additional compensation required for bearing risks. Some studies attempt to account for workers' perceived risks, but the results of these studies are not markedly different from those that do not.

Another assumption employed in hedonic wage studies is the existence of perfect labor markets in which workers are freely mobile and there is perfect information about jobs and job risks. Hedonic wage studies will not produce accurate estimates of the wage-risk tradeoff in imperfect markets where workers are unable to move freely between jobs or in which only union members have sufficient information and market power to receive higher wages for higher risk jobs. Since in reality labor markets are somewhat imperfect, many studies attempt to control for union membership and similar factors that might influence wage rates.

Hedonic wage models are limited to estimating values for relatively small risk changes. The observed wage and the estimated increment, or "premium," to accept higher risks represents the market equilibrium price for the entire set of workers in the study. This estimate is not necessarily the value that any particular worker would require to accept a risk increase, but for small changes in risk, it is very close.

A thorough treatment of the hedonic wage model that includes all of these considerations can be found in Viscusi (1992, 1993).

General Application of Hedonic Wage Studies to Benefits Assessment

Because they are narrowly focused on labor market tradeoffs, hedonic wage studies are not generally well-suited to measure the benefits of environmental regulation directly. That is, it is not usually feasible to perform a hedonic wage study to estimate the benefits that would accrue from a specific environmental policy action. Nonetheless, these studies have yielded consistent estimates of how groups of workers appear to value small risk changes. Environmental benefits assessments can draw upon these studies to estimate the value of reductions in environmental mortality risks.[18] Such an application is essentially an exercise in benefits transfer, which is described in greater detail later in this chapter.

Analysts should be aware that, although hedonic wage studies currently provide the most reliable and consistent estimates of the value of mortality risks, there are important differences in the types of risks captured in an hedonic wage study and the types of risks that are affected by environmental regulation. For instance, hedonic wage studies tend to focus on accidental deaths occurring among prime-aged males while deaths associated with environmental risk often occur among the elderly and may involve an extended latency period. Furthermore, elevated risks in hedonic wage studies are voluntarily accepted while environmental risks are often involuntarily borne. The nature and importance of these and other differences are detailed in Section 6 of this chapter.

Estimates of the value of changes in fatal risks are generally more relevant for environmental benefits assessment than are those for job-related non-fatal injuries. This is because these injuries are usually quite different from the non-fatal health risks associated with environmental policy actions.

Hedonic wage models have also used wage differentials across geographic areas to estimate values for environmental quality differences.[19] Theoretically, jobs in areas with poor environmental quality should pay less than identical jobs in areas with high environmental quality, again holding all else equal. There are a number of difficulties with employing hedonic wage models in this manner, including integrating wage and housing choices, and the need to assess intra-city variation in amenities. The majority of hedonic wage studies relevant for most EPA policies have focused on estimating values for health risks.

Considerations in Evaluating and Understanding Hedonic Wage Studies

- **Data requirements and implications:** Hedonic wage studies require large sets of data on labor

[18] Values of mortality risk have also been estimated using hedonic studies of automobile prices. While these studies produce values of life in a similar range, most environmental risk assessments rely on hedonic wage studies. For information on the automobile price literature, see Dreyfus and Viscusi (1996) and Viscusi (1992).

[19] This should not be confused with attempts to control for wage differentials across broad regions found in most existing wage-risk studies.

market behavior. Data on worker and job characteristics are generally collected using survey techniques. Risk information, however, is frequently retrieved from published sources reported at the occupation or industry level. These risk measures are then matched to the worker in the sample using information provided by the respondent on his or her job. The risk data used in most studies, however, are not complete. For example, although accidental, on-the-job (and almost immediate) deaths are generally reported, occupational diseases such as cancer are not accurately captured in most data.

The estimated wage-risk tradeoff can vary considerably across data sets and across methodologies. In particular, studies that draw upon data from high-risk jobs will generally provide valuations of risk that are lower than those that rely upon data from lower-risk jobs. This is due to a sample selection problem. Study results reflect the value to the sample population. High risk jobs tend to attract those who are less averse to taking risks and therefore require less compensation to face them.

- **Controlling for other risks:** If the study seeks to estimate values for fatal risks, it is important that the study control adequately for non-fatal risks in order to obtain an unbiased wage-risk estimate for mortality. Conversely, when values for non-fatal risk are being estimated, mortality risks should be considered in the wage-risk equation.
- **The scope of the risk measures:** Some labor market studies use actuarial data to determine the risk levels faced by workers. However, these data are not limited to occupational risks. They include all types of fatality risks faced by the individual both on and off the job. The degree to which these risks are correlated with job risks is unclear, but they would not be reflected in job-related compensation. These studies generally should be excluded from use in policy analysis since this problem will cause the tradeoffs they estimate to be biased downwards by an unknown amount.
- **The model for estimation:** Some labor market studies attempt to determine the value of a life year and the implicit discount rate workers apply to this value. While attractive in theory, the complexity of the structural models used in these studies leads to less robust estimates of the value of risk reduction than studies using conventional wage-risk estimation procedures. Such studies should be viewed as less reliable for use in valuing lifesaving programs.

7.5.2.3 Hedonic Property Value Studies

Hedonic property value studies are applications of the hedonic pricing method. The introduction to Section 7.5.2.2 (Hedonic Wage Studies) provides background on hedonic pricing methods in general.

Economic Foundation of Property Value Studies

Hedonic property value studies assert that individuals perceive housing units as bundles of attributes and derive different levels of utility from different combinations of these attributes. When transaction decisions are made, individuals make tradeoffs between money and attributes. These tradeoffs reveal the marginal values of these attributes and are central to hedonic property value studies. Hedonic property value studies use statistical regression methods and data from real estate markets to examine the increments in property values associated with different attributes.[20]

Structural attributes (e.g., number of bedrooms and age of house), neighborhood attributes (e.g., population demographics, crime, and school quality), and environmental attributes (e.g., air quality and proximity to hazardous waste sites) may influence property values. When assessing an environmental improvement, it is essential to separate the effect of the relevant environmental attribute on the price of a housing unit from the effects of other attributes. While deriving measures of marginal WTP using hedonic methods is straightforward, estimating measures of WTP for substantial or discrete (non-marginal) improvements in environmental quality is difficult. The use of hedonic property value studies for benefits assessment rests on careful interpretation of the hedonic price function and its relevance to the policy scenario being considered. Bartik

[20] To simplify the discussion, housing units are consistently used as examples. Hedonic property value studies are also completed on vacant land parcels.

(1988b) and Palmquist (1991, 1988) provide excellent, detailed discussions of benefits assessment using hedonic methods.

When using hedonic property value studies for benefits assessment, the measurement of the environmental attribute is central to the analysis. If the measurement of the environmental attribute does not match individuals' perceptions, then the results of the analysis may be biased.

The hedonic price function for housing units represents an equilibrium that results from the interaction of suppliers and demanders of housing in a market with full information. When this assumption is not met, the results of an hedonic analysis will not provide an exact representation of the tradeoffs individuals make across housing attributes and the marginal values associated with these different attributes.

General Application of Hedonic Property Value Studies to Benefits Assessment

Benefits assessment applications of hedonic property value studies focus on the relationship between property values and environmental attributes such as air quality, water quality, proximity to hazardous waste sites, and landscape characteristics. Hedonic property value studies are not widely used in environmental benefits assessments because of the limited transferability of hedonic results and the difficulties of using hedonic methods to describe the benefits associated with discrete (non-marginal) environmental improvements.

Using data on a sample of transactions, price is regressed on measures of the observable attributes and an hedonic price function is estimated. The coefficient on the environmental attribute reveals the marginal WTP for that attribute. Therefore, if the policy scenario considered results in a marginal environmental improvement, the estimated hedonic is well-suited to measure benefits. However, if the policy scenario considered results in a discrete improvement that affects numerous properties, additional information on preferences and the hedonic price function is required for assessing the true benefits of the environmental improvement.[21] When larger changes in environmental quality are considered, the analytical requirements increase because the hedonic price function and the level of utility of individuals may change.

The hedonic price function does not typically provide general information on individuals' WTP for the different attributes. Methods for identifying demand (or WTP) functions for the different attributes (e.g., using data from multiple markets or imposing assumptions about the functional form of the hedonic and/or the utility functions of individuals) exist, but they often rely on restrictive assumptions.[22] Furthermore, identifying WTP functions does not ensure the ability to measure the welfare gain from a discrete environmental improvement because markets intervene and prices change. As a result of these difficulties, approximations of welfare gains based on the hedonic price function are sometimes employed to assess benefits. See Palmquist (1991, 1988) and Bartik (1988b) for a detailed discussion of benefits assessment using hedonic methods, including guidance on developing lower and upper bound measures of benefits.

Considerations in Evaluating and Understanding Property Value Studies

- **Data requirements and implications:** Property value studies require large amounts of disaggregated data. Market transaction prices on individual parcels or housing units are preferred to aggregated data such as census tract information on average housing units because aggregation problems can be avoided. Data on attributes may include housing characteristics, sale dates, neighborhood amenities such as schools and parks, neighborhood demographic characteristics such as income, age, and race, and environmental quality.
- **Errors in variables:** Problems may arise from error in measuring prices (aggregated data) and errors in measuring product characteristics (particularly those related to the neighborhood and the environment). In addition, omitted variable bias problems may occur if relevant data are not available.
- **The measurement of environmental attributes:** The measurement of the environmental attribute included in the hedonic price function is central to

[21] There are cases when the hedonic price function can alone be used to measure welfare effects from a non-marginal change in the environmental attribute. For example, this holds if few properties are affected and the hedonic price function is not expected to shift.

[22] See Palmquist (1991) for a detailed discussion of identification issues.

the assessment of benefits. Researchers often use information available from the scientific community such as air or water quality monitoring data and then must determine how to assign the data to the individual houses in the data set. However, there may be differences between how these attributes are measured by scientists and how they are perceived by individuals. If this difference is large, the hedonic price function will not accurately represent the values of these attributes. Individual perceptions of environmental attributes are central to this type of analysis.[23] Another issue to consider is the timing of the effect from the environmental improvement. Some effects from environmental improvements change over time. Others may be understood differently over time depending on available information (e.g., hazardous waste sites). The choice of when and how to measure the environmental attribute for a given transaction price is complicated. Refer to Kiel and McClain (1995) for a discussion of price responses over time.

- **The model for estimation:** There are numerous statistical or econometric issues associated with applying hedonic methods to property value studies. These include the choice of functional form, the definition of the extent of the market, identification, and endogeneity. A brief overview of the first two estimation issues is presented below. Refer to Palmquist (1991) for a thorough treatment of the econometric issues associated with hedonic property value studies.

  Because economic theory offers limited guidance on the functional form of an hedonic price function, researchers often try several forms when estimating hedonic functions (e.g., semilogarithmic, inverse semilogarithmic, log-linear, or quadratic Box Cox). However, it is important to note that the choice of functional form has implications for benefits assessment. See Graves et al. (1988) and Cropper, Deck, and McConnell (1988) for discussions of issues related to the choice of functional form.

  The choice of data also has an effect on estimation. The extent of the market is defined by the scope of housing market data collected. Questions have been raised about how to define the extent of housing markets. Empirically, it is important to note that if the market is defined to be too big, the resulting coefficients of the hedonic price function may be biased. Conversely, if the market is defined too narrowly, the coefficients of the hedonic price function are less efficient. Refer to Michaels and Smith (1990) for information on defining the extent of the market.

- **Assessing the results an empirical study:** Two simple ways to assess the quality of a property value study are noted here. First, a review of the empirical work is informative. This involves assessing the quality of the data collected, the framing of the policy problem, the measurement of environmental attributes, and the statistical regression analysis. Second, the existing literature on hedonic methods is a valuable resource. Comparing data, modeling assumptions, and results across studies is a useful exercise. While variation is expected across studies, especially between those completed on different areas, some factors such as the signs of particular coefficients may be consistently reported.

7.5.2.4 Averting Behavior Method

The averting behavior method infers values from observations of how people change defensive behavior in response to changes in environmental quality. Defensive behaviors are usually defined as actions taken to reduce the risk of suffering environmental damages, as well as actions taken to mitigate the impact of environmental damages. The former category includes behaviors such as the use of air filters or boiling water prior to drinking it, while the latter includes the purchase of medical care or treatment. Faced with a given level of environmental risk, the averting behavior method assumes that individuals engage in these defensive behaviors to achieve an optimal level of health. By analyzing the expenditures associated with these defensive behaviors economists can attempt to estimate the value individuals place on small changes in risk.[24]

[23] For example, hedonic property value studies that address water quality often use measures such as water clarity because these are observable characteristics. In contrast, standard scientific measures such as BOD or pH may not be readily perceived by individuals.

[24] As Desvousges et al. (1998) note, the term "averting behavior study" has been used to describe at least three somewhat different approaches: attempts to estimate WTP for environmental quality; attempts to estimate WTP for health effects or other specific impacts; and simple summation of observed expenses.

Economic Foundation of Averting Behavior Method

The economic theory underlying the averting behavior method rests on a model of household production. In these models, households produce health benefits by combining an exogenous level of environmental quality with inputs such as defensive behaviors. The underlying theory predicts that a person will continue to take protective action as long as the perceived 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 cost just equals his or her WTP for these reductions. Thus, the benefit for a small reduction in health, or health risk, is 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.

Averting behaviors methods can provide theoretically correct measures of WTP to avoid a decline in environmental quality or an increase in environmental risks. To do so, however, they require a great deal of data and particular assumptions about consumer preferences. In practice, it has proven difficult to meet these requirements. More detail on the difficulties inherent in applying the averting behavior model can be found in Cropper and Freeman (1991).

One approach to estimation is to use observable expenditures on averting and mitigating activities to generate values that may be interpreted as a lower bound on WTP. Harrington and Portney (1987) demonstrate this by showing that WTP for small changes in environmental quality can be expressed as the sum of the values of four components: lost time, changes in averting expenditures, changes in mitigating expenditures, 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 using changes in these expenditures observed 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. These estimates can be an improvement over cost-of-illness estimates alone, because the latter usually captures only mitigating expenditures and lost time.[25]

Averting behavior results cannot always be interpreted as lower bounds on WTP, however, because conclusions may depend critically upon modeling conditions. For example, Shogren and Crocker (1991) use a theoretical model to show that the impact of changes in risk on defensive expenditures is ambiguous and that these expenditures need not be a lower bound value. Using the same model Quiggin (1992) imposes restrictions under which defensive expenditures will increase in response to increases in risk, providing support for self-protection expenditures as a method for benefits valuation. Recently, Shogren and Crocker (1999) show that averting behavior need not be a lower bound on value when both private and collective risk reduction strategies are considered.

Large, or non-marginal, changes in environmental quality require a somewhat different valuation strategy. Generally, it is not possible to obtain exact estimates of WTP for these changes. However, Bartik (1988a) details the conditions under which upper and lower bounds on WTP may be estimated in this circumstance. These bounds effectively bracket WTP.

Finally, analysts should remember that consumers base their actions on perceived benefits from defensive behaviors. If these perceptions differ from objective estimates of, for example, risk changes, the analysis will produce biased WTP estimates for a given change in objective risk. Surveys may be necessary in order to determine the benefits individuals perceive they are receiving when engaging in defensive activities. These perceived benefits can then be used as the object of the valuation estimates.

General Application of Averting Behavior Method to Benefits Assessment

The averting behavior method can, in theory, provide WTP estimates for a wide range of environmental benefits, including changes in mortality risks, morbidity risks, and damage to materials. Most recent research, however, has focused on health risk changes.

Mortality risks can be estimated with the averting behavior method by observing purchases of items that reduce the

[25] Cropper and Freeman (1991) note that the full costs of medical expenditures and lost work time may not be borne by individuals making these decisions due to insurance and paid sick leave. An analysis of social benefits would need to include the costs that have been shifted from the consumer to others.

risks of dying in an accident. These applications are sometimes known as consumer market studies. One of the difficulties with the use of averting behavior methods in this context is that many of the risk reduction actions are discrete rather than continuous, leading to estimates that are likely to understate the value of risk reduction to the average purchaser. These studies can also be sensitive to assumptions about unobserved costs such as the time required for employing or maintaining the risk-reducing good.

The most common focus of averting behavior models has been the estimation of values for non-fatal health (morbidity) risk changes. There have been many analyses of observable averting and mitigating expenditures. Some of these studies focus on behaviors that prevent or mitigate the impact of particular symptoms (e.g., shortness of breath, headaches), while others have examined averting expenditures in response to specific episodes of contamination (e.g., groundwater contamination). The difference in these endpoints is important. Because many contaminants can produce similar symptoms, studies that estimate values for symptoms may be more amenable to benefit transfer than those that are episode-specific. The latter may be more useful, however, if assessing the benefits of a regulation expected to reduce the probability of similar contamination episodes.

Considerations in Evaluating and Understanding Averting Behaviors Studies

- **Data requirements and implications:** Cropper and Freeman (1991) describe the data required for estimating WTP using the averting behavior method. These requirements are quite burdensome and include information detailing the severity, frequency, and duration of symptoms; exposure to environmental contaminants; actions taken to avert or mitigate damages; the costs of those behaviors and activities; and other variables that affect health outcomes (e.g., age, health status, chronic conditions).

  Often, data availability will limit the analysis to an examination of observed defensive expenditures. These results can be cautiously interpreted as a lower bound on WTP. Analysts should note that costs associated with pain and suffering will not be included in the estimate.

- **Separability of other benefits:** Many defensive behaviors not only avert or mitigate against environmental damages, but also provide other benefits. For example, air conditioners obviously provide cooling in addition to air filtering, and bottled water may not only reduce health risks, but may also be better tasting. The degree to which individuals engage in averting behaviors to obtain these benefits provides evidence of the value of these qualities, but disentangling the value of different components is not an easy task. In order to accurately produce estimates of WTP for a risk change, for example, averting behaviors studies must isolate the value for the effect of interest from the value of the other benefits conferred by the defensive activity. It is also 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 "other" benefits and disutilities associated with averting behaviors will result in biased estimates of WTP. Analysts should exercise caution in interpreting the results of studies that focus on goods in which there may be significant interrelated costs and benefits.

- **Modeling assumptions:** As noted above, restrictive assumptions are sometimes needed to make averting behavior models tractable. For example, assuming that the economy and the environment are additively separable may lead to unambiguous results, but it may be plausible only in particular circumstances. Shogren and Crocker (1999) note this fact and suggest that this assumption be justified whenever invoked. Analysts drawing upon averting behavior studies will need to review and assess the implications of these assumptions for the valuation estimates.

7.5.2.5 Cost-of-Illness Method

The health economics literature relies heavily upon the cost-of-illness method to value morbidity changes. The cost-of-illness method does not estimate WTP, but rather estimates the change in explicit market costs resulting from a change in the incidence of a given illness. Two types of costs measured in a typical cost-of-illness study are direct costs (such as diagnosis, treatment, rehabilitation, and accommodation) and indirect costs (including loss of work time).

Economic Foundation of Cost-of-Illness

The theoretical basis for the cost-of-illness method relies on two major assumptions (1) direct costs of morbidity reflect the economic value of goods and services used to treat illness and (2) a person's earnings reflect the economic value of lost production. Because of distortions in medical and labor markets, an argument could be made about whether these assumptions hold, but they are broadly consistent with neoclassical economics.

It is important to note that the cost of illness is not a measure of WTP. The cost-of-illness approach simply measures *ex post* costs and does not attempt to measure the loss in utility due to pain and suffering or the costs of any averting behaviors that individuals have taken to avoid the illness altogether (see Section 7.5.2.3 on averting behaviors). However, the cost-of-illness estimate may be considered a lower bound estimate of WTP (Harrington and Portney, 1987; Berger et al., 1987). The main reason that the cost of illness understates total WTP is the failure to account for many effects of disease. It ignores pain and suffering, defensive expenditures, lost leisure time, and any potential altruistic benefits. As a simple hypothetical example, if an individual spends five dollars on pain medication to treat a headache, and does not miss any time from work due to the headache, his or her cost of illness is five dollars. The individual's actual WTP to avoid the headache is likely to be greater than five dollars, assuming he experiences disutility from the pain the headache causes prior to taking the pain medication. Available comparisons of cost-of-illness and total WTP estimates suggest that the difference can be large (Rowe et al., 1995). However, this difference varies greatly across health effects and across individuals.

Most existing cost-of-illness studies estimate indirect costs based on the typical hours lost from a work schedule or home production, evaluated at an average hourly wage. The direct medical costs of illness are generally derived in one of two ways. The empirical approach uses a database of actual costs incurred for patients with the illness to estimate the total medical costs of the disease. The theoretical approach uses a panel of physicians to develop a generic treatment profile for the illness. Illness costs are estimated by multiplying the probability of a patient receiving a treatment by the cost of the treatment. For any particular application, the preferred approach will depend on availability of reliable actual cost data as well as characteristics of the illness under study.

Detailed descriptions of the cost-of-illness approach can be found in Cooper and Rice (1976), Hartunian et al. (1981), Hu and Sandifer (1981), Rice (1966), Rice et al. (1985) and EPA's *Cost of Illness Handbook* (EPA, forthcoming).

General Application of Cost-of-Illness Method to Benefits Assessment

Because the cost-of-illness approach does not rely on elaborate econometric models, and data are often readily available, implementation of this approach is relatively straightforward. For these same reasons, the approach is easy to explain to policy makers and the general public and tends to be less resource intensive than other approaches to health valuation. The method is generally suited for illnesses such as non-fatal cancers and other incidents of morbidity.

Cost-of-illness measures will understate WTP because they do not capture the disutility associated with anxiety, pain and suffering, or averting costs. On the other hand, some WTP estimates may understate social costs because they are unlikely to account for health care costs passed on to third parties (e.g., health insurance companies or hospitals in the case of direct medical expenses, and employers who offer sick leave in the case of time/productivity loss).

Considerations in Evaluating and Understanding Cost-of-Illness Studies

- ***Ex post* vs. *ex ante* measure:** As noted above, the cost-of-illness measures *ex post* costs of an illness rather than WTP to avoid the illness. Although the approach may account for costs shifted from the individual experiencing the illness to third parties, it fails to account for the disutility of pain and suffering, or any costs that may have been incurred in order to avoid the illness. Also, *ex post* measures cannot capture any value associated with risk attitudes. These attitudes may have a significant effect on WTP to reduce risks of more severe illnesses.

  It is also important to keep in mind that this measure captures the costs of choices that individuals make. Individuals generally choose when and how often to go to the doctor and when and for how long to stay home from work. These choices may be affected by the existence of health insurance, sick leave, and socioeconomic status.

- **Technological change:** Medical treatment technologies and methods are constantly changing, and this could push the true cost estimate for a given illness either higher or lower. When using previous cost-of-illness studies, the analyst should be sure to research whether and how the generally accepted treatment has changed from the time of the study.
- **Measuring the value of lost productivity:** Several issues arise in the indirect cost portion of a cost-of-illness study. Simply valuing the actual lost work time due to an illness may not capture the full loss of an individual's productivity in the case of a long-term chronic illness. Chronic illness may force an individual to work less than a full-time schedule, take a job at a lower pay rate than he or she would otherwise qualify for as a healthy person, or drop out of the labor force altogether. A second issue involved with estimating the value of lost productivity is the choice of wage rate. Even if the direct medical costs are estimated using individual actual cost data, it is highly unlikely that the individual data will include wages. Therefore, the wage rate chosen should reflect the demographic distribution of the illness under study. Furthermore, the value of lost time should include the productivity of those persons not involved in paid jobs. Homemakers' household upkeep and childcare services, retired persons' volunteering efforts, and students' time in school all directly or indirectly contribute to the productivity of society. Finally, the value of lost leisure time to an individual and his family is not included in most cost-of-illness studies.

7.5.3 Stated Preference Methods

Stated preference approaches attempt to measure WTP values directly. Unlike the revealed preference methods that infer values for environmental goods and services from observed behavior, stated preference methods rely on data from surveys that directly question respondents about their preferences to measure the value of environmental goods and services. This class of methods comprises several related techniques, including contingent valuation (CV), stated choice or conjoint analysis (CA), and less frequently, contingent ranking (CR). The common feature of these methods is direct questioning of members of a population about their likely choices in a hypothetical market. These three techniques are discussed below.

Economic Foundation of Stated Preference Methods

There are some situations in which data on actual behavior and choices cannot be used to derive estimates of the value of environmental goods and services. Stated preference methods rely on survey data rather than on data on observed behavior, therefore, they can be used to measure the value of environmental goods and services in most situations. The responses elicited from the surveys, if truthful, are either direct expressions of WTP or can be used to estimate WTP for the good in question.

- **Contingent Valuation:** Contingent valuation (CV) is the most well developed of the stated preference methods. CV surveys either ask respondents if they would pay a specified amount for a described hypothetical commodity or ask their highest WTP for it (for a good overview of the method see Hanemann, 1991; Mitchell and Carson, 1989; Carson, 2000; or Kopp et al. 1997). Concerns about the reliability of value estimates that come from CV studies have dominated debates about the methodology, since research has shown that bias can be introduced easily into these studies, especially if they are not carefully done. In particular, the concern that CV surveys do not require respondents to make actual payments has led critics to argue that responses to CV surveys are biased because of the hypothetical nature of the good. Reliability tests on the data that conform to expectations from both economic and psychological theory can enhance the credibility of a CV survey. Surveys without these tests should be suspect; surveys whose results fail the tests may be discredited.

  The result of the debates about the reliability of the CV method has been an infusion of methods and theory, particularly from the disciplines of psychology and survey research, to enhance questionnaire design to mitigate these concerns (Krosnick, 1991; Fischhoff, 1997). In addition, the National Oceanic and Atmospheric Administration (NOAA) convened a panel of well-known economists to review and evaluate the methodology in 1993. The panel devised a set of "best practices" recommendations for the method, particularly as it relates to natural resource damage assessments (NOAA, 1993). EPA subsequently prepared comments on the panel recommendations and regulations NOAA proposed that drew upon the panel's report (EPA, 1994).

- **Conjoint Analysis and Contingent Ranking:** Conjoint analysis (CA) and contingent ranking (CR) studies ask respondents to make choices between two or more (in the case of CA), or rank several (in the case of CR), similar commodities with different attributes and prices, in order to tease out the marginal value of particular attributes of the commodity of interest (Johnson et al., 1994). These methods are a variation on stated preference methods that aim to evaluate marginal tradeoffs rather than the total value for a described change that is evaluated in CV studies. Arising out of the marketing discipline, these methods rely on respondents' ability to make choices between commodities whose attributes differ in relation to one another. These methods often present respondents with a series of binary choice questions (e.g., "Given the descriptions of A and B, would you prefer A or B?") or multiple choice questions that ask respondents to make tradeoffs between prices and other features of commodities that are presented to them.

General Application of Stated Preference Methods to Benefits Assessment

More than 2,000 stated preference studies have been undertaken since the early 1970's. Among other things, these have been used to value changes in visibility (Chestnut and Rowe, 1990; Tolley et al. 1985), changes in surface water quality (Mitchell and Carson, 1984, 1986b), groundwater protection (McClelland et al., 1992), recreation services (Cameron and James, 1987; Bishop and Heberlien, 1979) and changes in health effects attributable to pollution (Krupnick and Cropper, 1989; Mitchell and Carson, 1986a; Viscusi et al., 1991).

Currently, contingent valuation is the only established method capable of estimating non-use values; however, most CV studies are designed to elicit respondents' total value for a given commodity. A number of researchers have attempted to disaggregate WTP values into "use" and "non-use" components. Examples of studies where non-use values have been specifically evaluated include McClelland et al. (1992) and Schulze et al. (1993). A more practical approach is to represent non-use values by employing the total WTP amounts given by persons who do not use the resource. The downside to this convenience is that there might be significant differences between those who use the resource and those who do not. Applying non-use values from the latter population to the former one may result in biased estimates.

In the context of environmental valuation, the commodity being purchased is usually a described change in environmental quality. This is a Hicksian measure, since it asks respondents to state the amount of income that they would be willing to forgo in order to have the described commodity, while making them as well off as they were without it and the payment. Similarly, they might be asked how much they would be willing to accept to put up with a nuisance or a loss. However, willingness to accept applications of CV are much more problematic because, unlike the case of WTP, there is no upper limit on the size of the opportunity set available to the respondent. This results in a strong potential for respondents to overstate the amount they would need to receive to compensate them for a loss.

While conjoint analysis (CA) has been used in marketing for some time, its application to environmental valuation began in the late 1980's. To date, it has not been subject to the level of testing and scrutiny that CV has had, so much less is known about the reliability (and how to enhance it) of these studies. The main methodological concerns that arise with CA studies are the viability of disaggregating the good in question into attributes that can be separately traded off in respondents' minds, and the problem that many respondents display intransitive preferences over the numerous, and often complex, set of choices. As a result of this complexity, heteroskedasticity is a pervasive problem with these methods.

An important limitation to using contingent valuation and other stated preference techniques is that it is expensive and time-consuming to survey the public about their preferences. Samples must be drawn, questionnaires developed, surveys administered either by mail, telephone or in person, and results coded and analyzed. In-person interviews are most expensive, but in some contexts are unavoidable due to the need to present complicated information to respondents or when they are required as criterion for legal evidence. Mail and phone surveys carry much lower costs and are often sufficient for use in the analysis of EPA policies.

Considerations in Evaluating Contingent Valuation Studies

Accurately measuring WTP for environmental goods and services using contingent valuation depends on the reliability and validity of the data collected. There are several issues to consider when evaluating study quality.

- **Content validity:** To evaluate a survey instrument itself, the analyst should look for a number of features that the researchers should have incorporated into the survey scenario. First, the commodity being valued must be clearly and concisely defined. A detailed explanation of the salient features of the environmental change being valued (the "commodity") begins with 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 (policy change) should be described, including an illustration of how and when the policy action would affect aspects of the environment that people might care about. Finally, the way the payment will be made (e.g., through taxes, user fees, etc.) may have large implications for the outcome, so careful attention should be paid to the rationale given for the choice of payment mechanism. Respondent attitudes about the provider and the implied property rights of the survey scenario can be used to evaluate the appropriateness of these features of the commodity description (Fischhoff and Furby, 1988). Questions that probe for respondent comprehension and acceptance of the commodity scenario can offer important indications about the potential for the study to be reliable.
- **Construct validity:** In CV studies, the main indicators of study quality are tests of internal validity that can be incorporated into study design. Internal validity is supported when variables that are expected by theory to be important determinants of preferences actually are statistically significant with the correct sign. For example, with normal goods, price is expected to have a negative effect on demand for a good, while household income is expected to have a positive effect, all else equal. Thus, respondents with higher income are expected to demand more of the good than respondents with low income. Familiarity with the good or its context can also be an important indicator of internal validity. One would intuitively expect that someone who fishes would know more about, and be willing to pay more for, a commodity that improves conditions for fishing than someone who never engages in outdoor recreation. Tests of sensitivity to scope, where the amount of the commodity is varied randomly over different sub-samples of survey respondents, can increase confidence in the results where the findings are consistent with theoretical expectations (Carson et al., 1993).
- **Criterion validity:** In order to assess criterion validity, the analyst needs to have an indicator of true value against which to evaluate values from contingent valuation studies. Given the lack of actual market prices, it is often impossible to conduct criterion validity tests. However, the quality of a CV study can also be gauged by comparing valuation estimates obtained using CV with those obtained using other techniques. At least one study that has compared CV valuation estimates with estimates derived using other valuation techniques has shown that, where the CV study was carefully designed, CV estimates are not inflated relative to the other estimates for the same commodity (Carson, 1996).

In conclusion, because of the issues raised here, among other factors, there is a divergence of views within the economic profession concerning whether stated preference methods can provide useful information on economic values and on validity of individuals' responses to hypothetical questions. Nonetheless, for goods providing non-use value, stated preference methods may provide the only analytic method currently available for benefits estimation.

7.5.4 Benefit Transfer

Benefit transfer can be a feasible alternative to using one of the primary stated or revealed preference research methods described in previous sections. Rather than collecting primary data, the benefit transfer approach relies on information from existing studies that have applied other methods. More precisely, Boyle and Bergstrom (1992) define benefit transfer as "the transfer of existing estimates of nonmarket values to a new study which is different from the study for which the values were originally estimated." The case for which the existing estimates were obtained is often referred to as the "study case," while the case under consideration for a new policy is termed the "policy case."

Existing applications of benefit transfer often focus on recreation demand. For an example of such a study, see Walsh et al. (1992). Applications of benefit transfer to value health effects have also been completed. See, for example, EPA's retrospective and prospective reports on the benefits and costs of the Clean Air Act (EPA, 1997a; EPA 1999a). Here, ranges of values for multiple-symptom health effects were calculated by combining results of studies that valued individual health effects. More information on benefit transfer in general and some of the issues discussed below can be found in EPA (1993) and a special issue of *Water Resources Research* (1992, Volume 28, Number 3) dedicated to the topic. More recently, Desvousges et al. (1998) discusses transfer studies in general, not only for valuation purposes. The authors illustrate the transfer method with a case-study estimating externalities associated with electric utility generation.

Is Benefit Transfer the Appropriate Technique?

The advantages to benefit transfer are clear. Original studies are time consuming and expensive; benefit transfer can reduce both the time and financial resources needed to develop benefits estimates of a proposed policy. Given the demands of the regulatory process, these considerations may be extremely important. Additionally, while the quality of primary research is unknown in advance, the analyst performing a benefit transfer is able to gauge the quality of existing studies prior to conducting the transfer exercise.

However, benefit transfer is not without drawbacks. Most important, estimates derived using benefit transfer techniques are unlikely to be as accurate as primary research tailored specifically to the new policy case. Of concern to the analyst is whether more accurate benefits information make a difference in the decision-making process. There are many situations in which a benefit transfer may provide adequate information. For example, if the entire range of benefits estimates from the transfer exercise falls well above or below the costs of the policy being considered, more accurate estimates will probably not alter the efficiency conclusion.

Other factors to consider when deciding whether to conduct a benefit transfer include the availability of relevant, high-quality existing studies and the degree to which additional primary research would reduce the uncertainty of the current benefits estimate.

Considerations in Evaluating and Understanding Benefit Transfer Studies

Currently, a systematic process for conducting benefit transfer does not exist. There are, however, well-accepted steps involved in the process. When conducting a benefit transfer, one should make certain that each of the following steps are carried out carefully:

- **Describe the policy case.** The first step in a benefit transfer is to describe the policy case so that its characteristics and consequences are understood. It is equally important to describe the population impacted by the proposed policy. As part of this step, it is important to determine whether effects of the policy will be felt by the general population or by specific subsets of individuals (e.g., users of a particular recreation site or children). Information on the affected population will generally be used to convert per person (or household) values to an aggregate benefits estimate.
- **Identify existing, relevant studies.** Existing, relevant studies are identified by conducting a literature search. This literature search should, ideally, include searches of published literature, reviews of survey articles, examination of databases, and consultation with researchers to identify government publications, unpublished research, works in progress, and other "gray literature."
- **Review available studies for quality and applicability.** In the third step, the analyst should review and assess the studies identified in the literature review for their quality and applicability to the policy case. The quality of the study case estimates will, in part, determine the quality of the benefit transfer. Indicators of quality will generally depend on the method used. See the previous discussions on each of the primary research methods for more information on assessing the quality of studies. Assessing studies for applicability involves determining whether

available studies are comparable to the policy case. Specifically:

— the basic commodities must be essentially equivalent;

— the baseline and extent of the change should be similar; and

— the affected populations should be similar.

The analyst should also determine whether adjustments can be made for important differences between the policy case and the study case. In some cases, it may prove enlightening to discuss your interpretation and intended use of the study case with the original authors. See Desvousges et al. (1992) for additional information on criteria used to determine quality and applicability. For more information on applicability as related to specific benefit categories, see the draft *Handbook for Non-Cancer Valuation* (EPA, 1999c), the *Children's Health Valuation Handbook* (EPA, forthcoming), and Desvousges et al. (1998).

- **Transfer the benefit estimates.** This step involves the actual transfer. There are four types of benefit transfer studies: point estimate, benefit function, meta-analysis, and Bayesian techniques. The point estimate approach involves taking the mean value (or range of values) from the study case and applying it directly to the policy case. As it is rare that a policy case and study case will be identical, this approach is not generally recommended. Rather than directly using existing values, analysts will often adjust point estimates based on judged differences between the study and policy cases. Judgments of this type should be based on economic theory, empirical evidence, and experience (Brookshire and Neill, 1992). The benefit function transfer approach is more refined but also more complex. If the study case provides a WTP function, valuation estimates can be updated by substituting applicable values of key variables, such as baseline risk and population characteristics (e.g., mean or median income, racial or age distribution) from the policy case into the benefit function. The most rigorous benefit transfer exercise uses meta-analysis. Meta-analysis is a statistical method of combining a number of valuation estimates that allows the analyst to systematically explore variation in existing value estimates across studies. As with the benefit function transfer approach, key variables from the policy case are inserted into the resulting benefit function. An alternative to the meta-analytic approach is the Bayesian approach. These techniques provide a systematic way of incorporating study case information with policy case information. Studies that have explored these concepts include Atkinson et al. (1992) and Boyle et al. (1994). A discussion of Bayesian approaches appears in Desvousges et al. (1998). Regardless of the procedure used, estimates are generally aggregated over the affected population to compute an overall benefits estimate.

- **Address uncertainty.** Benefit transfer involves judgements and assumptions. Throughout the analysis, the researcher should clearly describe all judgements and assumptions and their potential impact on final estimates, as well as any other sources of uncertainty inherent in the analysis.

7.6 Values for Major Benefit Categories

As noted earlier, EPA policies may reduce the risk of premature death, typically measured as the number of statistical lives "saved" as a result of the policy action. The benefits of these risk reductions are usually measured using the concept of the "value of a statistical life" (VSL). VSL estimates are derived from aggregated estimates of individual values for small changes in mortality risks. If 10,000 individuals are each willing to pay, for example, $500 for a reduction in risk of 1/10,000, then the value of saving one statistical life equals $500 times 10,000—or $5 million. This does not mean that any identifiable life is valued at this amount, but rather that the aggregate value of reducing a collection of small individual risks is worth $5 million in this case.

7.6.1 Human Health: Mortality Risks

EPA policies reduce a wide array of mortality risks. Some risks are experienced by young persons and others by older persons. Some risks result in death shortly after exposure, while others take years to manifest. For benefits analysis, mortality risks can generally be classified across two broad dimensions: the characteristics of the affected population and the characteristics of the risk itself, such as timing. These dimensions can be expected to affect the value of reducing mortality risks.

An ideal value estimate for fatal risk reduction would account for all of these demographic and risk characteristics. It would be derived from the preferences of the population affected by the policy, based on the type of risk that the policy is expected to reduce. For example, if a policy were designed to remove carcinogens at a suburban hazardous waste site, the ideal measure would represent the preferences for reduced cancer risks for the typical suburban dweller in the area. Unfortunately, it is simply too expensive and time-consuming to obtain such unique risk value estimates for each EPA policy.

Because original research is usually infeasible, analysts at EPA will need to draw from existing VSL estimates that have been obtained using well-established methods. However, virtually all available applications of these methods focus on risks that differ from environmental risks in a number of ways. Applying existing VSL estimates found in the economics literature is an exercise in benefit transfer and raises a number of issues associated with this technique.

This section characterizes and assesses these issues, recognizing that there are limitations to how effectively analysts can make adjustments in the benefit transfer process. The discussion is sometimes necessarily broad, given that there are a variety of different types of mortality risks affected by EPA policies. First, this section briefly reviews relevant economic valuation methods and the VSL estimates they provide. Then the bulk of this section highlights key considerations when considering and transferring these values for use in EPA benefits analysis. In order to focus the discussion, this section emphasizes those considerations that may be unique to benefit transfer in the context of mortality risk changes. Benefit transfer considerations that are common to all applications, including most demographic characteristics of the study and policy populations, are described in the section of the benefit transfer method itself.

7.6.1.1 Available Methods for Estimating Mortality Risk Values

The value of small changes in mortality risk is well-studied, although researchers generally acknowledge that there are formidable difficulties in measuring risk-dollar tradeoffs. Economists have developed three broad methods to estimate a value of mortality risk reduction, each of which is described below. When using any of these methods, researchers encounter uncertainties not only in isolating the amount of compensation received for assuming higher mortality risk, but also in estimating the actual and perceived risk increment inherent in the transaction.

- **Wage-risk analysis:** This method is well-established and the economics literature contains at least twenty high-quality wage-risk studies. The resulting VSL estimates range from about $0.7 million to more than $16 million (1997$) and are included in Exhibit 7-3. Wage-risk studies have been performed in a number of different industries and countries and their estimates appear to be somewhat sensitive to the data and econometric model used.[26] Workplace mortality risks, which tend to be dominated by deaths associated with accidents or other immediate causes, form the basis for VSL estimates from these studies. Environmental risks affected by EPA policies often differ from these types of risk in a number of ways.
- **Contingent valuation:** There are at least five high-quality published estimates of VSL based on the contingent valuation method. These estimates are broadly consistent with those generated by the wage-risk method and are included in Exhibit 7-3. These studies have not employed a fatal risk scenario involving an environmental cause and, therefore, suffer from some of the same "risk context" differences as wage-risk studies when transferred for use in EPA policy analyses. Recently, however, researchers have exhibited renewed interest in using the contingent valuation method to explore how particular factors affect WTP to reduce risks.[27]
- **Averting behavior studies:** The published literature contains several examples of averting behavior

[26] Viscusi (1992, 1993) discusses the implications of different specifications and data sets.

[27] Johannesson and Johansson (1996), for example, attempt to value extensions to life expectancy. The design of this study has, however, received some criticism (Krupnick et al., 1999).

Exhibit 7-3 Value of Statistcal Life Estimates (mean values in 1997 dollars)

Study	Method	Value of Statistical Life
Kneisner and Leeth (1991 - U.S.)	Labor Market	$0.7 million
Smith and Gilbert (1984)	Labor Market	$0.8 million
Dillingham (1985)	Labor Market	$1.1 million
Butler (1983)	Labor Market	$1.3 million
Miller and Guria (1991)	Contingent Valuation	$1.5 million
Moore and Viscusi (1988)	Labor Market	$3.0 million
Viscusi, Magat and Huber (1991)	Contingent Valuation	$3.3 million
Marin and Psacharopoulos (1982)	Labor Market	$3.4 million
Gegax et al. (1985)	Contingent Valuation	$4.0 million
Kneisner and Leeth (1991 - Australia)	Labor Market	$4.0 million
Gerking, de Haan and Schulze (1988)	Contingent Valuation	$4.1 million
Cousineau, Lecroix and Girard (1988)	Labor Market	$4.4 million
Jones-Lee (1989)	Contingent Valuation	$4.6 million
Dillingham (1985)	Labor Market	$4.7 million
Viscusi (1978, 1979)	Labor Market	$5.0 million
R.S. Smith (1976)	Labor Market	$5.6 million
V.K. Smith (1976)	Labor Market	$5.7 million
Olson (1981)	Labor Market	$6.3 million
Viscusi (1981)	Labor Market	$7.9 million
R.S. Smith (1974)	Labor Market	$8.7 million
Moore and Viscusi (1988)	Labor Market	$8.8 million
Kneisner and Leeth (1991 - Japan)	Labor Market	$9.2 million
Herzog and Schlottman (1987)	Labor Market	$11.0 million
Leigh and Folsom (1984)	Labor Market	$11.7 million
Leigh (1987)	Labor Market	$12.6 million
Garen (1988)	Labor Market	$16.3 million
Derived from EPA (1997) and Viscusi (1992).		

studies, also known as "consumer market studies." Consumer market studies have examined risk-dollar tradeoffs associated with highway speed (Ghosh et al., 1975), seatbelt use (Blomquist, 1979), use of smoke detectors (Dardis, 1980; Garbacz, 1989), and the use of child safety seats (Carlin and Sandy, 1991). All of these studies suffer from problems in estimating the full costs of consumer actions to reduce risks. For example, it is difficult to quantify the added expense and "cost of time" involved in purchasing, installing, and maintaining a smoke detector. Further, these studies cannot generally control for reductions in non-fatal risks that are associated with the averting action. Some researchers argue that these and other limitations lead consumer market studies to produce downwardly biased VSL estimates.[28]

[28] These criticisms include Fisher et al. (1989) and Viscusi (1992).

No clear consensus establishes one of these three methods or any particular study as exhibiting superior features for use in regulatory analyses. However, the relative abundance of available VSL estimates from wage-risk studies provides a range of broadly applicable values for reduced mortality risk. As in other benefit transfer exercises, a range or distribution of values serves as a starting point when seeking to identify values appropriated for a particular policy context.

7.6.1.2 Existing Reviews of Value of Statistical Life Estimates

Literature surveys found in Viscusi (1993) and Fisher (1989) represent the best starting points for VSL estimates. In both cases, the authors' goals included presenting a broadly applicable range of values rather than a point estimate. Viscusi (1993) is more recent and includes some studies not considered by Fisher.[29]

Drawing from these reviews, EPA identified 26 policy-relevant risk VSL studies as part of an extensive assessment titled *The Benefits and Costs of the Clean Air Act*, 1970 to 1990 (EPA, 1997a).[30] These are summarized in Exhibit 7-3 (IEc 1992, 1993a, 1993b). Five of the 26 studies are contingent valuation studies, the rest are wage-risk studies. To allow for probabilistic modeling of mortality risk reduction benefits, the analysts reviewed a number of common distributions to determine which best fit the distribution of mean values from the studies. A Weibull distribution was selected with a central tendency (or mean) of $5.8 million (1997$).

Although these studies are generally of high-quality, the $5.8 million measure of central tendency does not account for variation in study-specific factors underlying these VSL estimates. Further research on synthesizing the results of these and other studies, including the use of meta-analysis, may provide estimates better suited for benefit transfer to environmental policies.

7.6.1.3 Benefit Transfer Considerations for Using Existing VSL Estimates

Exhibit 7-3 contains the best range of estimates available at this time. For use in benefits analyses, EPA recommends a central estimate of $4.8 million (1990$), updated to the base year of the analysis. For example, updating this figure for inflation produces an estimate of $6.1 million in 1999 dollars.[31]

However, as with any benefit transfer exercise, it is important to consider differences in the nature of the base and policy cases. As noted earlier, for fatal risks these differences fall into two major categories:

- differences in the characteristics of the population; and
- differences in the characteristics of the risks being valued.

Particular differences in these categories are detailed below. Following this presentation is a summary assessment of how analysts might assess the impact of these population and risk dimensions. Generally, policy analysts considering mortality-related benefits should include at least a qualitative discussion of the potential impact of these factors on the overall results. It is important to recognize that the ultimate objective of the benefit transfer exercise is to adjust or correct for all of the factors that significantly affect the value of mortality risk reduction in the context of the policy. Analysts should carefully consider the implications of making adjustments for some relevant factors, but not for others.

[29] A third literature survey, Miller (1990), reviews a broad range of value of life studies, including estimates from averting behavior studies that others forcefully argue are not appropriate for environmental policy purposes. In addition, Miller's results are dependent on adjustments he makes to wage-risk data. These adjustments are the subject of debate among economists and may be difficult to defend in environmental contexts.

[30] This approach for valuing mortality risks was subject to extensive external peer review during the development process of this report. Since the report's release, this approach has been adopted in other EPA benefit analyses. Peer reviewers have recently confirmed the approach for use in a prospective analysis of the Clean Air Act (EPA, 1999a).

[31] This was estimated using the Consumer Price Index (CPI) for all goods and services. Many economists prefer to use the Gross Domestic Product (GDP) Deflator inflation index in some applications. The key issue for EPA analysts is that the chosen index be used consistently throughout the analysis.

Factors Associated with Demographic Characteristics

- **Age (longevity):** 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.[32] In the case of reductions in mortality risks, a young person is assumed to experience a greater expected benefit in total lifetime utility than an older person. This hypothesis may be confounded by the finding that older persons reveal a greater demand for reducing mortality risks and hence have a greater implicit value of a life year (Ehrlich and Chuma, 1990). Though few in number, empirical studies and theoretic models suggest that the value of a life follows a consistent "inverted-U" life-cycle, peaking in the region of mean age.[33]

  Two alternative adjustment techniques have been derived from this literature. The first, *valuation of statistical life-years*, is based on the concept of statistical life years introduced in Section 7.4.1. The most common application of this approach is illustrated in Moore and Viscusi (1988) and presumes that 1) the value of statistical life equals the sum of discounted values for each life year and 2) each life year has the same value. This method was applied as an alternative case in an effort to evaluate the sensitivity of the benefits estimates prepared for EPA's retrospective study of the costs and benefits of the Clean Air Act (EPA 1997). A second technique is to apply a distinct value or suite of values for mortality risk reduction depending on the age of incidence. However, there is relatively little available literature upon which to base such adjustments.[34]

- **Health status:** Individual health status also appears to affect WTP for mortality risk reduction. This is a relevant factor for valuation of environmental risks because individuals with impaired health are often the most vulnerable to mortality risks from environmental causes (for example, particulate air pollution appears to disproportionately affect individuals in an already impaired state of health). Health status is distinct from age (a "quality versus quantity" distinction) but the two factors are clearly correlated and therefore must be addressed jointly when considering the need for an adjustment. At least one pilot study has found that WTP for increased longevity decreases with a declining baseline health state (Desvousges et al., 1996).[35]

Factors Associated with Characteristics of Risk and Other Considerations

- **Risk characteristics** appear to affect the value that people place on risk reduction. A large body of work identifies eight dimensions of risk that affect human risk perception:[36]

  — voluntary/involuntary;

  — ordinary/catastrophic;

  — delayed/immediate;

  — natural/man-made;

  — old/new;

  — controllable/uncontrollable;

  — necessary/unnecessary; and

  — occasional/continuous.

[32] See, for example, Cropper and Sussman (1990) and Moore and Viscusi (1988).

[33] Jones-Lee et al. (1985) reach this conclusion empirically, considering both remaining years of life and the value of a life year. This conclusion supports theoretical predictions by Shepard and Zeckhauser (1982).

[34] This second approach was illustrated in one EPA study (EPA, 1995) for valuation of air pollution mortality risks, drawing upon adjustments measured in Jones-Lee et al. (1985).

[35] The fields of health economics and public health often account for health status through the use of quality adjusted life years (QALYs) or disability adjusted life years (DALYs). These measures have their place in evaluating the cost effectiveness of medical interventions and other policy contexts, but have not been fully integrated into the welfare economic literature on risk valuation. More information on QALYs can be found in Gold et al. (1996) and additional information on DALYs can be found in Murray (1994).

[36] A review of issues in risk perception is found in Slovic (1987). Other informative sources include Rowe (1977), Otway (1977), and Fischhoff et al. (1978).

Transferring VSL estimates between these categories may introduce bias.[37] There have been some recent efforts attempting to quantitatively assess these sources of bias. These studies generally conclude that voluntariness, control, and responsibility affect individual values for safety, although the direction and magnitude of these effects are somewhat uncertain.

Environmental risks may differ from those that form the basis of VSL estimates in many of these dimensions. Occupational risks, for example, are generally considered to be more voluntary in nature than are environmental risks, and may be more controllable.

- **Latency periods:** Many environmental policies are targeted at reducing the risk of effects such as cancer, where there may be an extended period of latency between the time of exposure and eventual death from the disease.[38] While the benefit of a reduction in exposure is an immediate reduction in the risk of the associated health endpoint, latency periods between exposure and manifestation may affect the value of that risk reduction. Existing VSL estimates are based upon risks of relatively immediate fatalities, making them an imperfect fit for a benefits analysis of many policies. Economic theory suggests that reducing the risk of a delayed health effect will be valued less than reducing the risk of a more immediate one, when controlling for other factors.

  A simple ad hoc approach to adjusting existing VSL estimates is to apply a financial discount rate over the expected latency period. However, defining latency periods with existing risk assessment methods may be difficult and empirical estimates may be highly uncertain. Further, the underlying assumptions supporting this procedure may oversimplify how individuals appear to consider delayed health effects.[39] Cropper and Sussman (1990) develop an alternative procedure to account for the influence of time on fatal risk reduction values, but their demonstration is data-intensive, requiring detailed life tables and age-specific VSL estimates.

- **Altruism:** The existing VSL literature focuses on individual risk tradeoffs, but there is evidence that people are willing to pay to reduce risks incurred by others. Although the literature on altruism is limited, several studies suggest that these values may be significant.[40] Other analysts advocate caution in attempting to inflate value of life estimates to reflect altruism, primarily because of concerns over the potential for double-counting.[41]

7.6.1.4 Summary of Advice from the Economics Literature

It is important to recognize the limitations of a single VSL point estimate and to consider whether any of the factors discussed above may have a significant impact on the benefits estimated for mortality risk reductions from environmental policies. In any given policy context, there may be several components that are both relevant and important and that could act to increase or decrease the appropriate risk reduction value used to estimate benefits. Adjustments for each these factors may offset one another to some extent.[42] Analysts should exercise caution in accounting for some important risk and population characteristics when unable to account for others.

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[37] Examples include Mendeloff and Kaplan (1990), McDaniels et al. (1992), Savage (1993), Jones-Lee and Loomes (1994, 1995, 1996), and Covey et al. (1995).

[38] Although latency is defined here as the time between exposure and fatality from illness, alternative definitions may be used in other contexts. For example, "latency" may refer to the time between exposure and the onset of symptoms. These symptoms may be experienced for an extended period of time before ultimately resulting in fatality.

[39] See, for example, the choice of discount rate discussion in Horowitz et al. (1990) and Rowlatt et al. (1998).

[40] In a study that included willingness to pay to reduce others' risk of illness from insecticides, Viscusi et al. (1988) found evidence of significant altruistic values. Jones-Lee et al. (1985) suggests an adjustment to value of statistical life estimates of about one-third to account for people's concern for the safety of others.

[41] Examples include Bergstrom (1982) and Viscusi (1992).

[42] Sometimes this might mean that very different risks will be valued similarly. There are relatively few studies that assess responses to different types of risk in an individual choice framework. A notable exception is Magat et al. (1996), which finds individuals indifferent between mortality risks from an automobile accident and those from fatal lymph cancer.

Because of the inherent uncertainty in any analysis, however, analysts should consider qualitative evaluations of these factors and explore where sensitivity analysis can satisfactorily address some of these concerns. The importance and relevance of each of the risk and demographic characteristics need to be considered. Depending upon specific policy context, there may be multiple alternatives for supplemental analysis.

For example, when policies do not affect the entire population equally, a sensitivity analysis may show the cost per life saved. In some contexts these values may provide useful information to decision makers on the relative merits of alternative policy options. However, cost-per-life-saved measures implicitly assume that all costs are associated with mortality reduction. For policies that provide other types of benefits, cost-per-life-saved measures may be misleading unless the value of those benefits are first deducted from cost estimates, but it is impossible to make these deductions when some benefits are either non-quantified or non-monetized. Because of these shortcomings, analysts will need to assess the usefulness of cost-per-life-saved measures on a case-by-case basis.

In general, the decision to perform sensitivity analysis will also depend upon the relative importance of mortality values in the overall benefits estimates and upon having sound theoretical and empirical economic literature upon which to structure the analysis. Parameter values used to formulate the sensitivity analyses must also be supported by the underlying risk assessment data.

What support does the economics literature currently offer for making these potential adjustments? Existing, feasible methods for age (or longevity) adjustments have significant limitations. Age adjustments may be desirable from a theoretical standpoint, but the relationship between the value of risk reductions and expected remaining life span is complex. Application of existing valuation of statistical life years approaches implicitly assumes a linear relationship in which each discounted life year is valued equally. As OMB (1996) notes, although "there are theoretical advantages to using a value of statistical life-year-extended (VSLY) approach, current research does not provide a definitive way of developing estimates of VSLY that are sensitive to such factors as current age, latency of effect, life years remaining, and social valuation of different risk reductions." The second alternative, applying a suite of values for these risks, lacks broad empirical support in the economics literature. However, the potential importance of this benefit transfer factor suggests that analysts consider sensitivity analysis when risk data—essentially risk estimates for specific age groups—are available. Emerging literature on the value of life expectancy extensions, based primarily on stated preference techniques, is beginning to help establish a basis for valuation in cases where the mortality risk reduction involves relatively short extensions of life.[43]

A small body of literature on the quantitative impact of risk characteristics on risk valuation exists. Although there is some qualitative consistency in the results of these studies, the risk valuation literature is not sufficiently robust to support quantitative adjustments for these factors at this time. Considerations associated with risk characteristics may deserve qualitative discussion in some policy contexts.

Both of the procedures available for accounting for latency have potentially serious shortcomings. For example, neither procedure addresses the dread of death or the morbidity that occurs prior to fatality from protracted diseases, such as that experienced with many cancers. As noted earlier, the simple "discounted VSL" approach may also oversimplify how individuals consider latency in their expressed WTP for reduced mortality risks. This literature does, however, suggest one alternative for conducting sensitivity analysis on this benefit transfer component.

7.6.1.5 Conclusion

In summary, these guidelines recognize the theoretically ideal measure of mortality risk reduction benefits and this section has discussed the many variables affecting such a measure. Due to current limitations in the existing economic literature, these guidelines conclude that an appropriate default approach for valuing these benefits is

[43] It should be noted that many observers have expressed reservations over adjusting the value of mortality risk reduction on the basis of population characteristics such as age. One of the ethical bases for these reservations is a concern that adjustments for population characteristics may imply support for variation in protection from environmental risks. Another consideration is that existing economic methods may not capture social willingness to pay to reduce health risks. Chapter 9 details how some these considerations may be informed by a separate assessment of equity. Chapter 10 describes the potential for efficiency and equity considerations to be considered together in a social welfare function.

provided by the central VSL estimate described earlier. However, analysts should carefully present the limitations of this estimate. Economic analyses should also fully characterize the nature of the risk and populations affected by the policy action and should confirm that these parameters are within the scope of the situations considered in these guidelines. While a qualitative discussion of these issues is generally warranted in EPA economic analyses, analysts should also consider a variety of quantitative sensitivity analyses on a case-by-case basis as data allow. The analytical goal is to characterize the impact of key attributes that differ between the policy and study cases. These attributes, and the degree to which they affect the value of risk reduction, may vary with each benefit transfer exercise, but analysts should consider the characteristics described above (e.g., age, health status, voluntariness of risk, latency) and values arising from altruism.

As the economic literature in this area evolves, WTP estimates for mortality risk reductions that more closely resemble those from environmental hazards may support more precise benefit transfers. Literature on the specific methods available to account for individual benefit transfer considerations will also continue to develop. EPA will continue to conduct annual reviews of the risk valuation literature and will reconsider and revise the recommendations in these guidelines accordingly. EPA will seek advice from the Science Advisory Board as guidance recommendations are revised.[44]

Despite the limitations described in this section, analysts should remember that mortality risk valuation remains one of the most studied benefit categories for environmental policies. Wage-risk studies, while not without limitations, nonetheless provide revealed preference estimates based on a well-tested method. Estimating mortality related benefits will often be relatively straightforward to implement, while other benefit categories will require more time and attention.

7.6.2 Human Health: Morbidity Risks

Morbidity valuation, or the valuation of non-fatal health effects, often requires addressing a more diverse set of issues than mortality valuation. First, there is a tremendous variety in the health endpoints considered for valuation. These endpoints vary with respect to their severity, including the degree to which other activities can be pursued and the degree of discomfort or pain associated with the ailment. The duration of the effect also varies considerably, from short term effects to those that may be permanently debilitating. Non-fatal health effects differ considerably with respect to the availability of existing value estimates. Some of these health effects have been valued multiple times with different methods, while others have not been the subject of any valuation studies.

Willingness to pay to reduce the risk of experiencing an illness is the preferred measure of value for morbidity effects. This measure includes several components. Illness imposes direct costs, such as expenses for medical care and medication, and indirect costs, such as lost time from paid work, maintaining a home, and pursuing leisure activities. Illness also imposes less easily measured, but equally real costs of discomfort, anxiety, pain, and suffering. Methods used to estimate WTP vary in the extent to which they capture these components.

A commonly used alternative to WTP is the avoided costs of illness (COI). For a given health effect, the COI approach will generally understate true WTP. By focusing on market measures of the value of health effects, it leaves out important components such as the value of avoiding pain and suffering. By focusing on *ex post* costs, it also does not capture the risk attitudes associated with *ex ante* WTP measures. However, for many effects, estimates of WTP are not currently available or are highly uncertain. Where estimates of WTP are not available, the potential

[44] A second review on this subject was recently completed by the Science Advisory Board (SAB), the results of which can be found in "An SAB Report on EPA's White Paper *Valuing the Benefits of Fatal Cancer Risk Reductions*," EPA-SAB-EEAC-00-013, July 27, 2000 (website address http://www.epa.gov/sab/eeacf013.pdf, accessed 8/28/00). The SAB review elaborates further on using the wage-risk literature for valuing mortality risk reductions, concluding that among the demographic and risk factors that might affect VSL estimates, the current literature can only support empirical adjustments related to the timing of the risk. First, the review supports adjusting willingness-to-pay estimates to account for higher future income levels, though not for cross-sectional differences in income. The second time-related adjustment recommended is to discount for risk reductions that are brought about in the future by current policy initiatives (that is, after a latency period), using the same rates used to discount other future benefits and costs. More information on the SAB review and its implications for the Guidelines will be released in a forthcoming supplement to this document.

bias inherent in relying on cost-of-illness estimates should, at minimum, be discussed qualitatively.

Time and resources will often not allow for original morbidity valuation research to support specific benefit analyses. As with other types of benefits, analysts will then need to look for estimates available from existing sources, and apply these values to the policy case using benefit transfer techniques. The discussion here is presented with this benefit transfer exercise in mind. The remaining parts of this section present a summary of methods commonly used to value reductions in morbidity, useful references for obtaining existing values, and issues that arise in transferring existing values to the analysis of EPA policies.

7.6.2.1 Available Methods for Estimating Morbidity Values

Researchers use a wide range of methods to value changes in morbidity risks. Some available methods measure the theoretically-preferred value of individual WTP to avoid a health effect, while others provide useful data but are less well-grounded in economic theory. Methods also differ in the perspective from which valuation is measured (e.g., before or after the incidence of morbidity) and the degree to which they account for all of the components of total WTP.

The three primary research methods used most often to value environmental morbidity are cost of illness, contingent valuation, and averting behavior, as described earlier. Several other primary valuation methods have been used less frequently to value morbidity from environmental causes: hedonic methods, risk-risk tradeoffs, health-state indexes, and studies of jury awards. However, these methods often do not provide monetary estimates of WTP or suffer from other methodological flaws, and are generally less useful for policy analysis.

- **Cost-of-illness:** The cost-of-illness method is straightforward to implement and explain to policy makers and has a number of other advantages. It has been applied for many years, is well-developed, and measures of direct and indirect costs are easily explained without reference to complex economic theory. Collection of additional data is often less expensive than for other methods, perhaps making it feasible to develop original cost-of-illness estimates in support of a specific policy or set of policies. Estimates for many illnesses are available from existing studies and span a wide range of health effects. EPA's *Cost of Illness Handbook* (EPA, forthcoming) contains an extensive collection of cost-of-illness estimates. As noted earlier, however, the cost-of-illness method has several shortcomings and its theoretical basis is quite limited. Generally, cost-of-illness estimates should be considered lower bounds on WTP.

- **Averting behavior:** In the case of morbidity valuation, the averting behavior method can provide WTP estimates based on actual behavior. These measures can account for all of the effects of health on individual well-being, including altruism toward other household members if averting actions are taken jointly (e.g., if everyone in the household drinks bottled water). However, the method has several weaknesses, as described in Section 7.5. Existing studies vary in their analytical approach. Some existing studies have attempted to estimate WTP for particular sets of illnesses (Gerking and Stanley, 1986; Dickie and Gerking, 1991; Bresnahan and Dickie, 1995). Others do not attempt to estimate *ex ante* WTP, but focus instead on actual household expenditures in response to a particular contamination episode or event (Harrington et al., 1989; Abdallah, 1990). In practice, most averting behavior estimates should be interpreted cautiously as a lower-bound estimate on WTP. Also, because behaviors generally avert a range of symptoms (e.g., a water filter removes contaminants with several potential health effects), it is difficult to isolate the value of avoiding those individual health effects that may be attributable to a particular EPA policy. Indiscriminate use of this method may raise significant problems with double counting or overestimation.

- **Stated preference methods:** Contingent valuation and other stated preference methods can be used to account for all the effects of illness on individual well-being, including pain and suffering. These methods appear to be the only ones capable of eliciting dollar values for altruism toward persons outside the household. Unlike the averting behavior or cost-of-illness methods, these can be applied to value the risks of illness lacking any connection to market transactions. Stated preference methods have been used to value a number of different health outcomes including accidental poisoning (Viscusi and Magat, 1987; Viscusi et al., 1988); coughing, congestion, and other minor symptoms (Berger et al., 1987); chronic bronchitis (Viscusi et al., 1991; Krupnick and Cropper, 1992); and nonfatal nerve disease (Magat et al., 1996). Some economists, however, express concerns about the hypothetical nature of the transaction and the difficulties inherent in ensuring that respondents understand the change in health status they are being asked to value.[45]

7.6.2.2 Existing References for Morbidity Values

Analysts have a number of resources available for obtaining information on morbidity values. While these references provide valuable information, they are not substitutes for careful evaluation of original studies when considering a benefit transfer. Useful general references for valuing non-fatal health effects include Tolley et al. (1994) and Johanneson (1995). Both of these books provide references to many existing health valuation studies and discuss issues associated with using these estimates for policy valuation. Desvousges et al. (1998) assess a number of existing studies in the context of performing a benefit transfer for a benefits analysis of improved air quality. Another good starting point for reviewing available estimates is EPA's *Handbook for Non-Cancer Valuation* (EPA, 1999c). This report will provide available, published estimates for many illnesses and reproductive and developmental effects.

Because estimates of WTP will not always be available for particular health effects, another useful resource is EPA's *Cost of Illness Handbook* (EPA, forthcoming). This handbook includes cost-of-illness estimates for many cancers, developmental illnesses and disabilities, and other illnesses. Work on the handbook is ongoing and new estimates will be included as they become available.[46]

Existing EPA economic analyses may also provide useful insights. For example, the *Benefits and Costs of the Clean Air Act* (EPA, 1997a) draws upon a number of existing studies to obtain values for reductions of a variety of health effects. The report describes the central estimates used in the analysis, how these estimates were derived, and attempts to quantify the uncertainty associated with using the estimates.

7.6.2.3 Benefit Transfer Using Existing Morbidity Value Estimates

Benefit transfer was detailed earlier in this chapter; however, there are issues associated with benefit transfer particular to morbidity valuation. As with any benefit transfer, analysts should:

- carefully describe the policy case;
- assess the quality of the studies and their applicability to the policy case;
- evaluate the plausibility of the findings;
- consider possible adjustments for differences between the subject of the study and the policy case; and
- explicitly address uncertainty.

EPA's *Handbook for Non-Cancer Valuation* (EPA, 1999c) contains additional information on this subject.

Matching the Study Case to the Policy Case

- **Assessing applicability:** A key element in evaluating the applicability of a study is the correspondence between the health effect valued in the study and the health effect influenced by the policy. An assessment of this correspondence must consider the set of symptoms covered in the study. The analyst should consider whether the study case consists of a larger or smaller set of symptoms than the policy case. The severity of the symptoms should also be commensurate, including the degree to which the illness limits activities and the extent of any discomfort, pain and suffering. Analysts should also assess whether the duration of the base and policy cases are similar.

  A second key factor is the similarity between the population examined in the study and the population affected by the policy. Key considerations include the

[45] See, for example, Mitchell and Carson, 1989; Cummings et al., 1986; NOAA, 1993; Bjornstad and Kahn, 1996; NRDA, 1994; Diamond and Hausman, 1994.

[46] The *Cost of Illness Handbook* will be available online.The website will be continuously updated as additional COI estimates are completed.

baseline health status of the populations, the age of the populations, and other demographic characteristics.

- **Evaluating plausibility:** The analyst should conduct some initial checks to evaluate whether the study case values are plausible or reasonable. For example, if the estimated value of avoiding an acute, reversible effect exceeds other reasonable estimated values for avoiding long-term, chronic effects, then the value for the acute effect is probably too large and will be difficult to defend. On the other hand, WTP values that are less than cost-of-illness values for the same effect are probably too low, particularly if the effect clearly results in pain or otherwise impairs activity.
- **Using the results of multiple studies:** After reviewing the quality and applicability of available studies, the analyst can apply the valuation estimates to the data on cases averted by each policy option. Because the value of morbidity avoidance is difficult to quantify precisely, it is useful where possible to apply estimates from more than one valuation technique. Where multiple studies are available that provide differing estimates, the range of values should be presented with a discussion of the advantages and limitations of the studies used. Estimates based solely on cost-of-illness values should be flagged as potentially understating total values. WTP studies of health effects that are similar in severity and duration may be used as a point of comparison.

Addressing Uncertainty and Related Concerns

Available estimates of non-fatal effects may suffer from several limitations. They may be derived from cost-of-illness methods that do not fully measure WTP to avoid the effect or may be transferred from studies of effects that are similar, but not identical to, the effect of concern. The extent to which adjustments or new research are needed to address these concerns will depend largely on the value of new information to the decision-making process. If morbidity values are a small component of total benefits and unlikely to influence the choice among policy options, then a qualitative discussion of uncertainty may be appropriate. Where morbidity values are a significant concern, quantified sensitivity analysis and additional data collection may be desired.

Some of the major sources of uncertainty are described below. Because of the diversity of the health effects of concern and of the studies used to value morbidity effects, this discussion is relatively general. The limitations and potential adjustments or analyses of uncertainty that are appropriate will vary greatly depending on the approach used for a particular policy analysis. More information on these issues is provided in EPA's *Non-Cancer Valuation Handbook* (EPA, 1999c).

- ***Ex ante* and *ex post* valuation estimates:** Environmental contamination will generally not cause an adverse health effect with certainty, but rather will increase the probability that the effect occurs, increase its severity given that it occurs, or both. People are likely to value these changes in risk differently than they would value certain changes in health status. While contingent valuation and other methods can adopt an *ex ante* perspective and obtain estimates for risk changes, many available studies provide *ex post* value estimates for morbidity effects. For minor health effects this difference in perspective may not be important, but for severe health effects the difference may be significant and *ex post* estimates may understate the benefits of a policy action. Analysts should address this issue at least qualitatively in these cases.
- **Incomplete estimates of willingness to pay:** The widespread availability of health insurance and paid sick leave shift the costs of illness from individuals to others. While this cost-shifting can be addressed explicitly in cost-of-illness studies, it may lead to problems in estimating total WTP through contingent valuation surveys. If the researcher does not adequately address these concerns, respondents may understate their WTP, assuming that some related costs will be borne by others.
- **Timing of health effects:** Environmental contamination may cause immediate or delayed health effects and the value of avoiding a given health effect likely depends on whether it occurs now or in the future. Recent empirical research confirms that workers discount future risks of fatal injuries on the job; that is, they are willing to pay less to reduce a future risk than a present risk of equal magnitude (Viscusi and Moore, 1989). In addition, a separate study concluded that individuals value policies that yield health benefits in the present more highly than policies that

yield the same benefits in the future (Cropper et al., 1994).

7.6.2.4 Summary

Morbidity benefits valuation can be a difficult process, often requiring careful judgment decisions by the analyst. Whether the analyst is conducting original research that supports the policy action or is drawing upon existing studies, clarity and transparency in the analysis is vital. When employing benefits transfer, some shortcomings in the "fit" of the study case to the policy case is to be expected. Addressing these shortcomings explicitly, conducting appropriate sensitivity analysis, and clearly stating assumptions can greatly enhance the credibility of the benefits analysis.

7.6.3 Ecological Benefit Valuation

In estimating ecological benefits, one is generally forced to value individual ecological service flows separately and then sum these estimates rather than constructing prices for changes in the structure and function of entire ecosystems. Alternative approaches that estimate the total value of ecosystems based on the replacement cost of the entire ecosystem or its embodied energy (e.g., Costanza et al., 1997; Ehrlich and Ehrlich, 1997; Pearce, 1998; Pimentel et al., 1997) have received considerable attention as of late. However, the results of these studies should not be incorporated into benefit assessments. The methods adopted in these studies are not well grounded in economic theory nor are they typically applicable to policy analysis. Pearce (1998) contains a critical review of the total value approach, as does Bockstael et al. (2000).

Although the economics literature is replete with benefit studies, the coverage is patchy considering the broad range of services and stressors addressed by EPA programs. Especially rare in the literature are examples of wide-scale changes, very small changes, or the consequences of long term ecological and economic change. Ongoing research has begun to address these data limitations. Examples include recent contingent valuation studies undertaken for purposes of natural resource damage assessments that attempt to elicit WTP for marginal changes in long-term environmental quality (Kopp et al., 1994). In addition, Layton and Brown (1997) attempt to elicit from respondents the value of one attribute of the long-term ecological changes expected to be associated with climate change.

Available Methods for Estimating Ecological Benefits

Economists have employed a variety of methods to estimate the benefits of improved ecological conditions. Issues particular to their implementation for this benefit category are discussed below.

- **Market models:** The benefit of changes in commercial crop, timber, or fish harvests can be estimated using a variety of available market models. Several studies have assessed the social welfare implications of changes in yields for a number of crop species. For example, Taylor et al. (1993) apply the Agricultural Sector Model and Kopp et al. (1985) apply the Regional Model Farm Agricultural Benefits Assessment Model to estimate welfare impacts of agricultural yield changes. Adams et al. (1997) use the Agricultural Simulation Model to estimate the economic effects associated with yield changes resulting from climate change.

  When dealing with timber or fisheries, bioeconomic models are designed to deal explicitly with time to account for the fact that environmental and market changes are not coincident. EPA has used the Timber Assessment Market Model (TAMM) to estimate ozone effects on commercial timber harvesting. The welfare impacts of changes in commercial fish harvests have also been examined, e.g., Alaskan king crab in Greenberg et al. (1994); herring in Mendelsohn (1993); and lobster in Wang and Kellogg (1988).

  If changes in service flows are small, current market prices can be used as a proxy for expected benefit. For example, a change in the commercial fish catch might be valued using the market price for the affected species. This approach can only be used in cases where fishing effort and price are unlikely to be affected by the policies. If these conditions do not hold, a market model should be applied to assess the effects of increased catch rates on supply conditions and market price, as discussed earlier in the section on consumer and producer surplus.

- **Production function approach:** Values for indirect, non-market benefits can be estimated when their contribution to production processes are expressed explicitly in a production function. As service flows increase, the welfare gain is essentially the marginal product of the service for small changes or is reflected by the shift in the marginal cost curve. Several studies have examined the relationship between environmental quality and crop production, such as measurement of ozone impacts by Heck et al. (1983). Moreover, researchers have applied this approach using household production functions to examine ecosystem services that benefit individuals directly or to establish the link between some services and their off-site benefits. Bell (1997) values wetland contributions to recreational fishing and Barbier (1994) values a range of indirect wetland benefits. Smith et al. (1993) estimate the impact of nitrogen and pesticide loadings on coastal water recreational fishing.

- **Averting behaviors approaches:** One such approach, the replacement cost method, uses purchases of market goods to infer the value of indirect, non-market services. Willingness to pay is revealed by efforts made to substitute for services provided by ecosystems. For example, since water treatment infrastructure replaces wetland functions, investment and operations and maintenance costs provide an estimate of the value of the water filtration service provided by wetlands. This method is justified only when individuals are proven willing to incur such replacement costs, through either their voluntary purchases or their support for public works projects. If so, the value of the service is at least as much as the replacement cost.

  Another variation on this theme applies to actions that reduce the cost of complying with existing policies. For example, a reduction in atmospheric nitrogen deposition in the watershed of an estuary may ultimately reduce the costs incurred in reducing other sources of nitrogen to the system, such as added controls on POTWs. This approach is generally useful in situations satisfying two criteria: alternative pollution control methods are prescribed through existing policies and the new policies under consideration would provide a lower cost method for achieving the desired level of environmental protection.

- **Hedonic methods:** Hedonic property models can isolate the relationship between environmental quality and housing prices from the effects of variation in other attributes such as size, location, and security. This method is often used to use to value regional difference in air quality. Smith and Huang (1995) conduct a meta-analysis to examine how well the models perform in this context and discern a statistically significant relationship between housing prices and air quality measures in general. Hedonic models have also been used to estimate the impact of landfill closure (Kohlhase, 1991), nuisances from odors (Palmquist et al., 1997), and the existence and remediation of toxic contaminated sites (Kiel, 1995) on nearby property values. Other applications have addressed the costs of land-use restrictions (Parsons, 1992) and the benefits of water quality improvements (Rich and Moffitt, 1982). A promising development involves the application of hedonic methods in a model of land-use change to explore the ecological and economic consequences of landscape alteration (Bockstael, 1996). In addition, Geoghegan et al. (1997) estimate the impact of land uses adjacent to and near one's home.

- **Recreation demand models:** Recreation demand models are discussed in detail in Section 5.2.1 of this chapter. Recreation demand models are based on the tradeoff between travel expense and environmental quality. While early travel cost models dealt with single sites, did not consider environmental quality, and suffered from a range of limitations and biases, more recent efforts overcome these problems. For example, a travel cost model was used to estimate the WTP by Chesapeake Bay beach users for a 20 percent improvement in water quality as measured by nitrogen and phosphorous loadings (Bockstael et al., 1989).

  Random utility models have been used in a wide variety of studies to estimate the recreational fishing benefit of fresh water quality improvements. Montgomery and Needelman (1997) apply a random utility model to fishing behavior and estimate the benefits of eliminating toxic contamination from New York lakes and ponds. Recreation demand models also have been used in the context of forest management (Englin and Mendelsohn, 1991; Dwyer et al., 1983), the ecological effects of natural resource damages (Hausman et al., 1995; Morey and Rowe, 1995), health risks associated with fish advisories (Jakus et. al., 1997), and non-point source pollution controls in estuaries (Kaoru et al., 1995).

- **Stated preference approaches:** As discussed earlier, stated preference methods represent the only means of obtaining a value for non-use benefits. For instance, research has tried to measure how much better people feel that various wildlife species are alive and well (Stevens et al., 1991). Individuals have also expressed a willingness to pay to protect visibility in national parks, whether or not they plan to visit these parks (Crocker and Shogren, 1991). Contingent valuation has been applied to the other ecological benefit categories as well. EPA has regularly used the results of one such study to value water quality improvements in fresh water (Mitchell and Carson, 1986b).

7.6.4 Materials Damage

Market methods are the primary technique used to quantify benefits falling in this category. Materials damages can include changes in both the quantity of the materials and in the quality. Linking changes in environmental quality with the provision of service flows from materials can be difficult because of the limited understanding of the physical effects (e.g., scientific information), the timing of some effects (e.g., long-term), and risk responses of producers and consumers of these service flows. When feasible, assessment typically involves combining the output of an environmental model with stressor-response function and/or price information to estimate the impact of the change in environmental quality on production (inputs) or consumption (output) of the material service flows. The market response to this impact serves as the basis for the welfare change and benefits assessment. In practice, these market methods may be implemented as reduced from economic models that relate averting or mitigating expenditures to ambient pollutant levels. The degree to which behavioral adjustments are considered when measuring the market response are important and models that incorporate behavioral responses are preferred to those that do not. Refer to Adams and Crocker (1991) for a detailed discussion of this and other features of materials-damages-benefits assessment. See the analysis of household soiling in EPA (1997b) for an example that employs a reduced form economic model relating defensive expenditures to ambient pollutant levels.

7.7 References

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