Types of Values

Economists have decomposed the total economic value conferred by resources into three main components: (1) *use value*, (2) *option value*, and (3) *nonuse value*. Use value reflects the direct use of the environmental resource. Examples include fish harvested from the sea, timber harvested from the forest, water extracted from a stream for irrigation, even the scenic beauty conferred by a natural vista. If you used one of your senses—sight, sound, touch, taste or smell—to experience the resource, then you have *used* the resource. Some of these uses are called *passive use values* or *nonconsumptive use values* if the resource is not actually used up (consumed) in the process of experiencing it. Pollution can cause a loss of use value such as when air pollution increases the vulnerability to illness, an oil spill adversely affects a fishery, or when smog enshrouds a scenic vista.

Option value reflects the willingness to pay to preserve an option to use the environment in the future even if one is not currently using it. Whereas use value reflects the value derived from current use, option value reflects the desire to preserve a potential for possible future use. Are you planning to go to Yellowstone National Park next summer? Would you like to pay to preserve the option to go?

Nonuse value reflects the common observation that people are more than willing to pay for improving or preserving resources that they will never use. A pure nonuse value is also called *existence value*. Another type of nonuse value is a *bequest* value; the willingness to pay to preserve the resource for future generations. When the Bureau of Reclamation began looking at sites for dams near the Grand Canyon, groups such as the Sierra Club protested the potential loss of this unique resource. With Glen Canyon already flooded by Lake Powell, even those who never intended to visit the area recognized the potential loss. Because this value does not derive either from direct use or potential use, it represents a very different category of value.

These categories of value can be combined to produce the total willingness to pay (*TWP*):

$$TWP = \text{Use Value} + \text{Option Value} + \text{Nonuse Value}.$$

Since nonuse values are derived from motivations other than personal use, they are obviously less tangible than use values. Estimated nonuse values can be quite large. Therefore, it is not surprising that they are controversial. Indeed when the Department of Interior drew up its regulations on the appropriate procedures for performing natural resource damage assessment, it prohibited the inclusion of nonuse values unless use values for the incident under consideration were zero. A subsequent 1989 decision by the District of Columbia Court of Appeals (880 F. 2nd 432) overruled this decision and allowed nonuse values to be included as long as they could be measured.

Classifying Valuation Methods

Typically, the researcher's goal is to estimate the total willingness to pay for the good or service in question. This is the area under the demand curve up to the quantity consumed (recall the discussion from Chapter 2). For a market good, this calculation is relatively straightforward. However, as we examine in this chapter, nonmarket goods and services require the estimation of willingness to pay either through examining 1) behavior; 2) responses to surveys; or 3) extracting the willingness to pay from markets for related goods.

Several methods are available to estimate these values. This section will provide a brief overview to convey some sense of the range of possibilities and how they are related. Subsequent sections will provide more specific information about how they are actually used.

The possibilities are presented in Table 3.1. Direct revealed preference methods are those that are based on actual observable choices and from which actual resource values can be directly inferred. For example, in calculating the economic damage to local fishermen from an oil spill, the *revealed preference method* might calculate how much the catch declined and the resulting lost value of the lower catch. In this case, prices are directly observable, and their use allows the direct calculation of the loss in value.

Compare this with the direct stated preference case that might be used when the value is not directly observable. For example, the nonuse value of a panda bear, a right whale, or another endangered species is not directly observable. Hence, a survey might be used to derive this value by attempting to elicit the respondents' willingness to pay (their *stated preference*) for the preservation of the species.

This approach, called *contingent valuation*, provides a means of deriving values that cannot be obtained more traditionally. The simplest version of this approach merely asks respondents what value they would place on an environmental change (such as the loss of a wetlands or increased exposure to pollution) or on preserving the resource in its current state. Another version asks whether the respondent would pay $*X* to prevent the change or to preserve the species. The answers reveal either an upper bound (in the case of a "no" answer) or a lower bound (in the case of a "yes" answer). This version, however, presents the respondent with a more familiar question: a good with an associated price.

The major concern with the use of the contingent valuation method has been the potential for survey respondents to give biased answers. Five types of potential bias have been the focus of a large amount of research: (1) strategic bias, (2) information bias, (3) starting-point bias, (4) hypothetical bias, and (5) the observed discrepancy between willingness to pay (*WTP*) and willingness to accept (*WTA*).

Strategic bias arises when the respondent provides a biased answer in order to influence a particular outcome. For example, if a decision to preserve a stretch of river for fishing depends on whether or not the survey produces a sufficiently large value for fishing, the respondents who enjoy fishing may be tempted to provide an answer that ensures a high value rather than a lower value that reflects their true valuation.

Information bias may arise whenever respondents are forced to value attributes with which they have little or no experience. For example, the valuation by a recreationist of a loss in water quality in one body of water may be based on the ease of substituting recreation on another body

TABLE 3.1 Economic Methods for Measuring Environmental and Resource Values

Methods	Revealed Preferences	Stated Preferences
Direct	Market price	Contingent valuation
	Simulated markets	
Indirect	Travel cost	Attribute-based models
	Hedonic property values	Conjoint analysis
	Hedonic wage values	Choice experiments
	Avoidance expenditures	Contingent ranking

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of water. If the respondent has no experience using the second body of water, the valuation will be based on an entirely false perception.

Starting-point bias may arise in those survey instruments in which a respondent is asked to check off his or her answers from a predefined range of possibilities. How that range is defined by the designer of the survey may affect the resulting answers. A range of $0 to $100 may produce a valuation by respondents different from, for example, a range of $10 to $100, even if no bids are in the $0 to $10 range. In a study of willingness to pay to protect nature areas in Denmark from new highway development, Ladenburg and Olsen (2008) found a gender-specific starting point bias attributed to female respondents.

Hypothetical bias can enter the picture because the respondent is being confronted by a contrived, rather than an actual, set of choices. Since he or she will not actually have to pay the estimated value, the respondent may treat the survey casually, providing ill-considered answers. One way to get a handle on the magnitude of this bias is to compare willingness to pay estimates derived from surveys directly with actual expenditures. In one survey of 10 studies, Hannemann (1994) found that although some of the studies concluded that the willingness to pay estimates derived from surveys exceeded actual expenditures, the majority found that the differences were not statistically significant. More recently, Ehmke, Lusk, and List (2008) tested whether hypothetical bias depends on national culture. In a study based on student experiments in China, France, Indiana, Kansas, and Niger, they find significant differences in bias across nations. Given that policymakers frequently rely on existing benefits estimates when making decisions on other locations, this finding should not be taken lightly, a subject we treat in more depth in a subsequent section of this chapter.

Increasingly, environmental economists are using these types of experiments to try to determine the severity of some of these biases as well as to learn how to reduce bias. Some of these experiments are conducted in a lab, such as a computer lab or a classroom.

The biases may be quite subtle. In one such experiment on voluntary provision of public goods (donations), Landry et al. (2006) found that for door-to-door interviews, an increase in physical attractiveness of the interviewer led to sizable increases in giving. When isolated by gender, they find the effect to be even stronger with solely female interviewers, partially by inducing more households to contribute (e.g., increased participation rates when males answered the door!). Their findings suggest that personal attractiveness "elicits contributions from agents who would not otherwise elect to contribute" (p. 770).

The final source of bias addresses observed gaps between willingness to pay and willingness to accept compensation. Respondents to contingent valuation surveys tend to report much higher values when asked for their willingness to accept compensation for a specified loss than if asked for their willingness to pay for a specified improvement in quantity or quality. Economic theory suggests the two should be equal. Debate 3.1 explores some possible reasons offered for the difference.

Much experimental work has been done on contingent valuation to determine how serious a problem these biases may present. One survey (Carson et al., 1994) uncovered 1,672 contingent valuation studies. Are the results from these surveys reliable enough for the policy process?

Faced with the need to answer this question in order to compute damages from oil spills, the National Oceanic and Atmospheric Administration (NOAA) convened a panel of independent economic experts (including two Nobel Prize laureates) to evaluate the use of contingent valuation methods for determining lost passive use or nonuse values. Their report, issued on January 15, 1993 (58 FR 4602), was cautiously supportive.

DEBATE 3.1

Willingness to Pay Versus Willingness to Accept: Why So Different?

Many contingent valuation studies have found that respondents tend to report much higher values for questions that ask what compensation the respondent would be willing to accept (*WTA*) to give something up than for questions that ask for the willingness to pay (*WTP*) for an incremental improvement he or she would receive. Economic theory suggests that differences between *WTP* and *WTA* should be small, but experimental findings both in environmental economics and in other microeconomic studies have found large differences. Why?

Some economists have attributed the discrepancy to a psychological endowment effect; the value of something you own is greater than something you do not. Psychologists attribute this to a form of loss aversion; their research sugests that humans tend to value losses more highly than comparable gains (Kahneman, Knetsch, and Thaler, 1990).

Others have suggested that the difference is explainable in terms of the market context. In the absence of good substitutes, large differences between *WTA* and *WTP* would be the expected outcome. In the presence of close substitutes, *WTP* and *WTA* should not be that different, but the divergence between the two measures should increase as the degree of substitution decreases (Hanemann, 1991 and Shogren et al., 1994).

The characteristics of the good may matter as well. In their review of the evidence provided by experimental studies, Horowitz and McConnell (2002) find that for "ordinary goods" the difference between *WTA* and *WTP* is smaller than for public and nonmarket goods. Their results support the notion that property rights are not neutral.

The moral context of the valuation may matter as well. Croson et al. (draft) show that *WTA* increases with culpability as long as the party causing the damage is also paying for the repairs. If, however, a third party is paying, *WTA* is insensitive to culpability. This difference suggests that the valuation includes an amount levied in punishment for the party who caused the damage (the *WTA* becomes the lost value plus a sanction).

Ultimately, the choice of which concept to use in environmental valuation comes down to how the associated property right is allocated. If someone owns the right to the resource, using *WTA* is the appropriate approach. If the respondent does not have the right, using *WTP* is the right approach. However, as Horowitz and McConnell point out, since the holders and nonholders of "rights" value them differently, the initial allocation of property rights will have strong influence on valuation decisions for environmental amenities.

Sources: Croson, R., J. J. Rachlinski, and J. Johnston, "Culpability as an Explanation of the WTA-WTP Discrepancy in Contingent Valuation." (Draft 2005). Hanemann, W. M., "Willingness to Pay and Willingness to Accept: How Much Can They Differ?" *American Economic Review*, 81, 635–647, 1991. Horowitz, J. K. and K. E. McConnell, "A Review of WTA/WTP Studies," *Journal of Environmental Economics and Management*, 44, 426–447, 2002. Kahneman, D., J. Knetsch, and R. Thaler, "Experimental Tests of the Endowment Effect and the Coase Theorem," *Journal of Political Economy*, 98, 1325–1348, 1990. Shogren, J. F., Senung Y. Shin, D. J. Hayes, and J. B. Kliebenstein, "Resolving Differences in Willingness to Pay and Willingness to Accept." *American Economic Review* 84 (1), 1994: 255–270.

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recent addition to the economist's tool kit. GIS offers a powerful collection of tools for depicting and examining spatial relationships. Most simply, GIS can be used to produce compelling graphics that convey the spatial structure of data and communicate analytic results with a force and clarity otherwise impossible. But the technology's real value lies in the potential it brings to ask novel questions and enrich our understanding of social and economic processes by explicitly considering their spatial structure. Models that address environmental externalities have, almost by definition, a strong spatial component. GIS can contribute, for example, to incorporating spatial dimensions into such areas as benefit/cost analysis (Bateman et al., 2002) and urban and real estate economics (Clapp et al., 1997).

Hedonic property valuation models have also recently been refined using GIS technology. Since hedonic property models are fundamentally spatial in nature, their use of GIS is a natural fit. Housing prices vary systematically and predictably from neighborhood to neighborhood. Spatial characteristics, from air quality to the availability of open space, can influence property values of entire neighborhoods; if one house enjoys abundant open space or especially good air quality, it is highly likely that its neighbors do as well.

Lewis, Bohlen, and Wilson (2008) use GIS and statistical analysis to evaluate the impacts of dams and dam removal on local property values. In a unique "experiment" they collected data on property sales for 10 years (before and after) the removal of the Edwards Dam on the Kennebec River in Maine, the first federally licensed hydropower dam to be removed primarily for the purpose of river restoration. They also collected data on property sales approximately 20 miles upstream where two other dams remained in place. GIS technology enhanced this study by facilitating the calculation of the distance from each home to both the river and the nearby dams. Lewis et al. found that living close to a dam extracted a penalty relative to living further away.[7] In other words, willingness to pay for identical housing was higher, the further away from the dam the house was located. They also found that the penalty near the Edwards Dam site dropped to nearly zero after the dam was removed. Interestingly, the penalty associated with the upstream dams remained, but its magnitude also fell after the removal of the downstream dam. Can you think of reasons why?[8] Example 3.3 shows how the use of GIS can enable hedonic property value models to investigate how the view from a particular piece of property might affect its value.

Valuing Human Life

One fascinating public policy area where these various approaches have been applied is in the valuation of human life. Many government programs, from those controlling hazardous pollutants in the workplace or in drinking water, to those improving nuclear power plant safety, are designed to save human lives as well as to reduce illness. How resources should be allocated among these programs depends crucially on the value of human life. How is life to be valued?

The simple answer, of course, is that life is priceless, but that observation turns out to be not very helpful. Because the resources used to prevent loss of life are scarce, choices must be made. The economic approach to valuing lifesaving reductions in environmental risk is to calculate the

[7]And relative to other waterfront housing.

[8]Lewis, Lynne Y., Curtis Bohlen, and Sarah Wilson. 2008. "Dams, Dam Removal and River Restoration: A Hedonic Analysis," *Contemporary Economic Policy*, 26(2):175–186. Interestingly, after this study was complete, one of the two upstream dams, the Fort Halifax Dam, was removed in July 2008 after years of litigation about its removal.

EXAMPLE 3.3

Using GIS to Inform Hedonic Property Values: Visualizing the Data

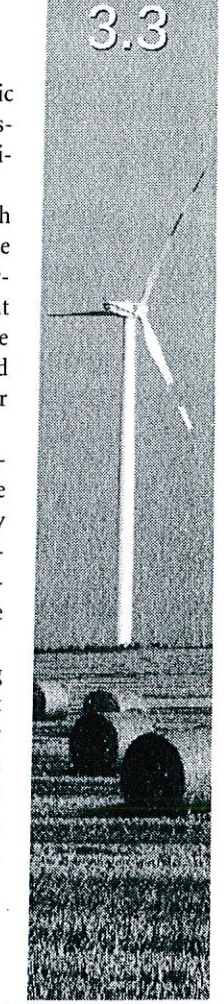
For nonmarket valuation, GIS has proven to be especially helpful in enhancing hedonic property value models by incorporating both the proximity of environmental characteristics and their size or amount. GIS studies have also allowed for the incorporation of variables that reflect nearby types and diversity of land use.

Geo-coding housing transactions assigns a latitude and longitude coordinate to each sale. GIS allows other spatial data such as land use, watercourses, and census data to be "layered" on top of the map. By drawing a circle around each house of the desired circumference, GIS can help analysts calculate the amount of each amenity that is in that circle as well as the density and types of people who live there. Numerous census data are available on variables such as income, age, education, crime rates, commuting time, and so on. GIS also makes it relatively easy to calculate straight-line distances to desired (or undesired) locations such as parks, lakes, schools, or landfills.

In their paper entitled "Out of Sight, Out of Mind? Using GIS to Incorporate Visibility in Hedonic Property Value Models," Paterson and Boyle (2002) use GIS to measure the extent to which visibility measures affect house prices in Connecticut. In their study visibility is measured as the percentage of land visible within one kilometer of the property, both in total and broken out for various land use categories. Finally, they added variables that measured the percentage of area within one kilometer of each house transaction that is developed, in agriculture, forested, or covered by water.

They find that visibility is indeed an important environmental variable in explaining property values, but the nature of the viewshed matters. While simply having a view is not a significant determinant of property values, viewing certain types of land use is. For example, proximity to development reduces property values, but only if the development is visible, suggesting that out of sight, really does mean out of mind! They conclude that any analysis that omits variables that reflect nearby environmental conditions can lead to misleading or incorrect conclusions about the impacts of land use on property values. GIS is a powerful tool for helping researchers include these important variables.

Source: Robert Paterson and Kevin Boyle, "Out of Sight, Out of Mind? Using GIS to Incorporate Visibility in Hedonic Property Value Models" *Land Economics*, 2002.



change in the probability of death resulting from the reduction in environmental risk and to place a value on the probability change. Thus, it is not life itself that is being valued, but rather a reduction in the probability that some segment of the population could be expected to die earlier than otherwise. Debate 3.2 examines this controversy.

Although it is the increased probability of death that is being evaluated, it is possible to translate the value derived from this procedure into an "implied value of human life." This is accomplished by dividing the amount each individual is willing to pay for a specific reduction in the probability of death by the probability reduction. Suppose, for example, that a particular environmental policy could be expected to reduce the average concentration of a toxic substance

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DEBATE 3.2

Is Valuing Human Life Immoral?

In 2004 economist Frank Ackerman and lawyer Lisa Heinzerling teamed up to write a book that questions the morality of using benefit/cost analysis to evaluate regulations designed to protect human life. In *Priceless: On Knowing the Price of Everything and the Value of Nothing* (2004), they argue that benefit/cost analysis is immoral because it represents a retreat from the traditional standard that all citizens have an absolute right to be free from harm caused by pollution. When it justifies a regulation that will allow some pollution-induced deaths, benefits/cost analysis violates this absolute right.

Economist Maureen Cropper responds that to the contrary it would be immoral not to consider the effectiveness of lifesaving measures. Resources are scarce and they must be allocated so as to produce the greatest good. If all pollution were reduced to zero, even if that were possible, the cost would be extremely high and the resources to cover that cost would have to be diverted from other beneficial uses. Professor Cropper also suggests that it would be immoral to impose costs on people about which they have no say—for example, the costs of additional pollution controls—without at least trying to consider what choices people would make themselves. Like it or not, hard choices must be made.

Cropper also points out that people are always making decisions that recognize a trade-off between the cost of more protection and the health consequences of not taking the protection. Thinking in terms of trade-offs is a familiar concept. She points out that people drive faster to save time, thereby increasing their risk of dying. They also decide how much money to spend on medicines to lower their risk of disease or they may take jobs that pose morbidity or even mortality risks if the wages are high enough.

In her response to Ackerman and Heinzerling, Cropper acknowledges that benefit/cost analysis has its flaws and that it should never be the only decision-making guide. Nonetheless, she argues that it does add useful information to the process and throwing that information away could prove to be detrimental to the very people that Ackerman and Heinzerling seek to protect.

Sources: Frank Ackerman and Lisa Heinzerling, *Priceless: On Knowing the Price of Everything and the Value of Nothing* (New York: The New Press, 2004); Frank Ackerman, "Morality, Cost-Benefit and the Price of Life," *Environmental Forum* (21)5, 2004: 46–47; and Maureen Cropper, "Immoral Not to Weigh Benefits Against Costs," *Environmental Forum* (21)5, 2004: 47–48.

to which one million people are exposed. Suppose further that this reduction in exposure could be expected to reduce the risk of death from 1 out of 100,000 to 1 out of 150,000. This implies that the number of expected deaths would fall from 10 to 6.67 in the exposed population as a result of this policy. If each of the one million people exposed is willing to pay $15 for this risk reduction (for a total of $15 million), then the implied value of a life is approximately $4.5 million ($15 million divided by 3.33).

What actual values have been derived from these methods? A survey (Viscusi, 1996) of a large number of studies examining reductions in a number of life-threatening risks found that most implied values for human life (in 1984 dollars) were between $3 million and $7 million. This same

survey went on to suggest that the most appropriate estimates were probably closer to the $5 million estimate. In other words, all government programs resulting in risk reductions costing less than $5 million would be justified in benefit/cost terms. Those costing more might or might not be justified, depending on the appropriate value of a life saved in the particular risk context being examined. In 2008 the Bush administration's EPA used $6.9 million in current dollars as the value of a human life. Interestingly this value represents a drop of nearly $1 million from just five years earlier. The controversial rationale for the drop was based on some newer studies that produced lower estimates, a rationale that was not even universally accepted within EPA. The agency's water division, for example, never adopted the change and as of 2006 was using $8.7 million in current dollars.

How have health, safety, and environmental regulations lived up to this recommendation? As Table 3.4 suggests, not very well. A very large number of regulations listed in that table could be justified only if the value of a life saved were much higher than the upper value of $7 million.

TABLE 3.4 The Cost of Risk-Reducing Regulations

	Agency, Year, and Status	Initial Annual Risk[a]	Annual Lives Saved	Cost Per Life Saved (millions of 1984 $)
Unvented space heaters	CPSC 1980 F	2.7 in 10^5	63.000	$ .10
Cabin fire protection	FAA 1985 F	6.5 in 10^8	15.000	.20
Passive restraints/belts	NHTSA 1984 F	9.1 in 10^5	1,850.000	.30
Seat cushion flammability	FAA 1984 F	1.6 in 10^7	37.000	.60
Floor emergency lighting	FAA 1984 F	2.2 in 10^8	5.000	.70
Concrete & masonry construction	OSHA 1988 F	1.4 in 10^5	6.500	1.40
Hazard communication	OSHA 1983 F	4.0 in 10^5	200.000	1.80
Benzene/fugitive emissions	EPA 1984 F	2.1 in 10^4	0.310	2.80
Radionuclides/uranium mines	EPA 1984 F	1.4 in 10^4	1.100	6.90
Benzene	OSHA 1987 F	8.8 in 10^4	3.800	17.10
Asbestos	EPA 1989 F	2.9 in 10^5	10.000	104.20
Benzene/storage	EPA 1984 R	6.0 in 10^7	0.043	202.00
Radionuclides/DOE facilities	EPA 1984 R	4.3 in 10^6	0.001	210.00
Radionuclides/elemental phosphorous	EPA 1984 R	1.4 in 10^5	0.046	270.00
Benzene/ethylbenzenol styrene	EPA 1984 R	2.0 in 10^6	0.006	483.00
Arsenic/low-arsenic copper	EPA 1986 R	2.6 in 10^4	0.090	764.00
Benzene/maleic anhydride	EPA 1984 R	1.1 in 10^6	0.029	820.00
Land disposal	EPA 1988 F	2.3 in 10^8	2.520	3,500.00
Formaldehyde	OSHA 1987 F	6.8 in 10^4	0.010	72,000.00

[a] "Initial Annual Risk" indicates annual deaths per exposed population; an exposed population of 10^3 is 1,000, 10^4 is 10,000, and so on. In the "Agency Year and Status" column, R and F represent Rejected and Final rule, respectively.

Sources: Adapted from Viscusi, W. Kip, "Economic Foundations of the Current Regulatory Reform Efforts," *The Journal of Economic Perspectives* 10 (1996): Tables 1 and 2, 124–125.

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costs borne by another. This admittedly extreme case does serve to illustrate a basic principle—ensuring that a particular policy is efficient provides an important, but not always the sole, basis for public policy. Other aspects, such as who reaps the benefit or bears the burden, are also important.

In summary, on the positive side, benefit/cost analysis is frequently a very useful part of the policy process. Even when the underlying data are not strictly reliable, the outcomes may not be sensitive to that unreliability. In other circumstances, the data may be reliable enough to give indications of the consequences of broad policy directions, even when they are not reliable enough to fine-tune those policies. Benefit/cost analysis, when done correctly, can provide a useful complement to the other influences on the political process by providing a constructive collection of information to decision makers.

On the negative side, benefit/cost analysis has been attacked as seeming to promise more than can actually be delivered, particularly in the absence of solid benefit information. This concern has triggered two responses. First, regulatory processes have been developed that can be implemented with very little information and yet have desirable economic properties. The recent reforms in air pollution control, which we will cover in Chapter 16, provide one powerful example.

The second response involves techniques that supply useful information to the policy process without relying on controversial techniques to monetize environmental services that are difficult to value. The rest of this chapter deals with the two most prominent of these—*cost-effectiveness analysis* and *impact analysis*.

Even when benefits are difficult or impossible to quantify, economic analysis has much to offer. Policy-makers should know, for example, how much various policy actions will cost and what their impacts on society will be, even if the efficient policy choice cannot be identified with any certainty. Cost-effectiveness analysis and impact analysis both respond to this need, albeit in different ways.

Cost-Effectiveness Analysis

What can be done to guide policy when the requisite valuation for benefit/cost analysis is either unavailable or not sufficiently reliable? Without a good measure of benefits, making an efficient choice is no longer possible.

In such cases, frequently it is possible, however, to set a policy target on some basis other than a strict comparison of benefits and costs. One example is pollution control. What level of pollution should be established as the maximum acceptable level? In many countries, studies of the effects of a particular pollutant on human health have been used as the basis for establishing that pollutant's maximum acceptable concentration. Researchers attempt to find a threshold level below which no damage seems to occur. That calculated threshold can then be further lowered to provide a margin of safety and the adjusted threshold becomes the pollution target.

Approaches could also be based upon expert opinion. Ecologists, for example, could be enlisted to define the critical numbers of certain species or the specific critical wetlands resources that should be preserved.

Once the policy target is specified, however, economic analysis can have a great deal to say about the cost consequences of choosing a means of achieving that objective. The cost consequences are important not only because eliminating wasteful expenditures is an appropriate goal in its own right, but also to assure that choices do not trigger a political backlash.

Typically, several means of achieving the specified objective are available; some will be relatively inexpensive, while others turn out to be very expensive. The problems are frequently complicated enough that identifying the cheapest manner of achieving an objective cannot be accomplished without a rather detailed analysis of the choices.

Cost-effectiveness analysis frequently involves an *optimization procedure*. An optimization procedure, in this context, is merely a systematic method for finding the lowest-cost means of accomplishing the objective. This procedure does not, in general, produce an efficient allocation because the predetermined objective may not be efficient. All efficient policies are cost-effective, but not all cost-effective policies are efficient.

In Chapter 2 we introduced the efficiency equimarginal principle. According to that principle, net benefits are maximized when the marginal benefit is equal to the marginal cost.

A similar, and equally important, equimarginal principle exists for cost effectiveness:

> *Second equimarginal principle (the cost-effectiveness equimarginal principle): The least-cost means of achieving an environmental target will have been achieved when the marginal costs of all possible means of achievement are equal.*

Suppose, for example, that we want to achieve a specific emission reduction across a region, and several possible techniques exist for reducing emissions. How much of the control responsibility should each technique bear? The cost-effectiveness equimarginal principle suggests that the techniques should be used such that the desired reduction is achieved and the cost of achieving the last unit of emission reduction (in other words, the marginal control cost) should be the same for all sources.

To demonstrate why this principle is valid, suppose that we have an allocation of control responsibility where marginal control costs are much higher for one set of techniques than for another. This cannot be the least-cost allocation since we could lower cost while retaining the same amount of emission reduction. Costs could be lowered by allocating more control to the lower marginal cost sources and less to the high marginal cost sources. Since it is possible to find a way to lower cost, then clearly the initial allocation could not have minimized cost. Once marginal costs are equalized, it becomes impossible to find any lower-cost way of achieving the same degree of emissions reduction; therefore, that allocation must be the allocation that minimizes costs.

In our pollution control example, cost-effectiveness can be used to find the least-cost means of meeting a particular standard and its associated cost. Using this cost as a benchmark case, we can estimate how much costs could be expected to increase from this minimum level if policies that are not cost-effective are implemented. Cost-effectiveness analysis can also be used to determine how much compliance costs can be expected to change if the EPA chooses a more stringent or less stringent standard. The case study presented in Example 3.5 not only illustrates the use of cost-effectiveness analysis, but also shows that costs can be very sensitive to the regulatory approach chosen by the EPA.

Impact Analysis

What can be done when the information needed to perform a benefit/cost analysis or a cost-effectiveness analysis is not available? The analytical technique designed to deal with this problem is called *impact analysis*. An impact analysis, regardless of whether it focuses on economic impact or environmental impact or both, attempts to quantify the consequences of various actions.

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NO_2 Control in Chicago: An Example of Cost-Effectiveness Analysis

In order to compare compliance costs of meeting a predetermined ambient air quality standard in Chicago, (Seskin, Anderson, and Reid, 1983) gathered information on the cost of control for each of 797 stationary sources of nitrogen oxide emissions in Chicago, along with measured air quality at 100 different locations within the city. The relationship between ambient air quality at those receptors and emissions from the 797 sources was then modeled using mathematical equations. Once these equations were estimated, the model was calibrated to ensure that it was capable of re-creating the actual situation in Chicago. Following successful calibration, this model was used to simulate what would happen if EPA were to take various regulatory actions.

The results indicated that a cost-effective strategy would cost less than one-tenth as much as the traditional approach to control and less than one-seventh as much as a more sophisticated version of the traditional approach. In absolute terms, moving to a more cost-effective policy was estimated to save more than $100 million annually in the Chicago area alone. In Chapters 15 and 16 we will examine in detail the current movement toward cost-effective policies, a movement triggered in part by studies such as this one.

In contrast to benefit/cost analysis, a pure impact analysis makes no attempt to convert all these consequences into a one-dimensional measure, such as dollars, to ensure comparability. In contrast to cost-effectiveness analysis, impact analysis does not necessarily attempt to optimize. Impact analysis places a large amount of relatively undigested information at the disposal of the policy-maker. It is up to the policymaker to assess the importance of the various consequences and act accordingly.

On January 1, 1970, President Nixon signed the National Environmental Policy Act of 1969. This Act, among other things, directed all agencies of the federal government to

> *include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment, a detailed statement by the responsible official on—*
>
> *i. the environmental impact of the proposed action,*
> *ii. any adverse environmental effects which cannot be avoided should the proposal be implemented,*
> *iii. alternatives to the proposed action,*
> *iv. the relationships between local short-term uses of man's environment and the maintenance and enhancement of long-term productivity, and*
> *v. any irreversible and irretrievable commitments of resources which would be involved in the proposed action should it be implemented.*[13]

This was the beginning of the environmental impact statement, which is now a familiar, if controversial, part of environmental policymaking.

[13] 83 Stat. 853.

Current environmental impact statements are more sophisticated than their early predecessors and may contain a benefit/cost analysis or a cost-effectiveness analysis in addition to other more traditional impact measurements. Historically, however, the tendency had been to issue huge environmental impact statements that are virtually impossible to comprehend in their entirety.

In response, the Council on Environmental Quality, which, by law, administers the environmental impact statement process, has set content standards that are now resulting in shorter, more concise statements. To the extent that they merely quantify consequences, statements can avoid the problem of "hidden value judgments" that sometimes plague benefit/cost analysis, but they do so only by bombarding the policy-makers with masses of noncomparable information. All three of the techniques discussed in this chapter are useful, but none of them can stake a claim as being universally the "best" approach. The nature of the information that is available and its reliability make a difference.

Summary

In this chapter we have examined the most prominent but certainly not the only techniques available to supply policy-makers with the information needed to implement efficient policy. Finding the total economic value of the service flows requires estimating three components of value: (1) use value, (2) option value, and (3) nonuse or passive-use value.

Our review of these various techniques available to estimate these values included direct observation, contingent valuation, contingent ranking, conjoint analysis, *travel cost*, hedonic property and wage studies, and averting or defensive expenditures.

Because benefit/cost analysis is both very powerful and very controversial, in 1996 a group of economists of quite different political persuasions got together to attempt to reach some consensus on its proper role in environmental decision-making. Their conclusion is worth reproducing in its entirety:

> *Benefit-cost analysis can play an important role in legislative and regulatory policy debates on protecting and improving health, safety, and the natural environment. Although formal benefit-cost analysis should not be viewed as either necessary or sufficient for designing sensible policy, it can provide an exceptionally useful framework for consistently organizing disparate information, and in this way, it can greatly improve the process and, hence, the outcome of policy analysis. If properly done, benefit-cost analysis can be of great help to agencies participating in the development of environmental, health and safety regulations, and it can likewise be useful in evaluating agency decision-making and in shaping statutes.*[14]

Even when benefits are difficult to calculate, however, economic analysis in the form of cost effectiveness can be valuable. This technique can establish the least expensive ways to accomplish predetermined policy goals and to assess the extra costs involved when policies other than the least-cost policy are chosen. What it cannot do is answer the question of whether those predetermined policy goals are efficient.

At the end of the spectrum is impact analysis, which merely identifies and quantifies the impacts of particular policies without any pretense of optimality or even comparability of the information generated. Impact analysis does not guarantee an efficient outcome.

[14] From Kenneth Arrow et al. "Is There a Role for Benefit-Cost Analysis in Environmental, Health and Safety Regulation?" *Science* 272 (April 12, 1996): 221–222. Reprinted with permission from AAAS.