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Has Economic Research Answered the Needs of Environmental Policy?[1]

Maureen L. Cropper

University of Maryland, World Bank and Resources for the Future, 1818 H Street NW, Washington, DC 20433

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Efficient environmental regulation requires empirical estimates of the benefits and costs of environmental standards and policies. In the last 25 years, economists have made considerable progress in developing methods for measuring environmental benefits and in providing empirical estimates of costs and benefits. This paper reviews this progress. It examines by category of benefit the advancements made in providing estimates that can be used in benefit-cost analyses and identifies areas where additional research is required.

1. INTRODUCTION

Environmental economics is fundamentally a policy-oriented discipline. Its goal is to promote the efficient use of environmental resources, which, by their open-access nature, are unlikely to be allocated efficiently by private markets. This means that government policies are needed to ensure efficient use of the environment. It is therefore appropriate to ask whether researchers in environmental economics have provided the tools that environmental policymakers would need to formulate and implement environmental policy.

The goal of this article is to assess whether researchers have developed the tools that environmental managers need to make efficient decisions. Typically, these decisions fall into two categories: (1) how to set goals or standards and (2) how to achieve stated environmental goals, e.g., how to achieve the National Ambient Air Quality Standards or restore an environment that has been damaged. Efficiency requires that goals (or standards) be set to balance benefits and costs and that stated goals be achieved at least cost. To limit the scope of the paper, we focus on the first topic, asking whether economists have provided the tools policy makers would need to set environmental goals in an economically efficient manner.

This is a slightly different question than those addressed by other authors in this volume. Robert Hahn asks: "How have environmental and natural resource economists influenced U.S. environmental policy?" In answering this question

[1]This paper has benefited from conversations with the following people, none of whom is responsible for the views expressed herein: Nancy Bockstael, Curtis Carlson, Richard Carson, David Chapman, John Charbonneau, Lauraine Chestnut, A. Myrick Freeman III, Virginia Kibler, Alan Krupnick, Elizabeth McClelland, Ted McConnell, William O'Neil, George Parsons, Nathalie Simon, Ivar Strand, Jeff Vaughan, and W. Kip Viscusi. I especially thank V. Kerry Smith and Rob Stavins for helpful comments on an earlier version of the paper.

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Hahn points out that economists have had limited success in inducing policymakers to balance benefits and costs in setting environmental standards. Indeed, two major statutes—the Clean Air Act and the Clean Water Act—prohibit the balancing of benefits and costs. As Kerry Smith, however, points out in his review of JEEM and non-market valuation, the valuation of environmental damages has influenced regulatory outcomes, if not the setting of the National Ambient Air Quality Standards.

Whether or not economists are always successful in promoting efficiency in environmental regulation, it is appropriate to ask whether they have provided the tools that would enable policymakers to make efficient decisions were they willing and able to do so. This paper asks what progress economists have made in publishing studies that could be used to make environmental policy more efficient. It also attempts to identify which topics require more research.

Scope and Organization of the Paper

The tools required to set environmental standards efficiently are the tools of benefit-cost analysis. We examine to what extent economists have provided the methods necessary to estimate the benefits and costs of various environmental programs, as well as actual estimates of benefits and costs. Since a particular category of benefits—e.g., health or recreation—may be associated with a variety of environmental programs, the discussion is organized by the type of benefit measured. To limit the scope of the paper, we focus on the types of benefits associated with pollution control. Environmental management is thus equated with pollution regulation, including air pollution, water pollution, hazardous and solid waste disposal, and the regulation of toxic substances in consumption and production.

The discussion begins with an overview of the advances made in valuing non-market benefits over the past 25 years. This is followed by a discussion of each of the major categories of benefits associated with pollution control: health benefits, recreation benefits, the non-use values associated with improving water quality, visibility benefits, and ecological benefits. The benefits of pollution control policies to agriculture and other reductions in materials damage are treated more briefly, as is the measurement of costs. The paper concludes with a summary of remaining research needs and considers how these might be met.

2. THE ADEQUACY OF TOOLS FOR BENEFIT MEASUREMENT: OVERVIEW

During the last 25 years economists have made significant progress in developing methods for valuing environmental benefits. Prior to 1974, stated preference techniques were in their infancy. The last 25 years have witnessed the development of the Contingent Valuation Method (Mitchell and Carson [87]), and the adoption of related techniques, such as conjoint analysis, to value virtually all categories of environmental benefits. The period was also one of major advances in the development of revealed preference techniques. The theory of hedonic markets was formally stated in 1974 (Rosen [101]) and subsequently applied to both housing and labor markets to value health and aesthetic benefits. The household production model was used to develop a theoretical basis for valuing the benefits of health (Harrington and Portney [62]) and recreation (Bockstael and McConnell [15]).

Advances in statistical techniques have paralleled advances in the theory of benefit measurement. This includes the development of methods to analyze data from contingent valuation surveys (Cameron and James [21]; Hanemann *et al.* [60]), as well as the application of random utility models to analyze the choice of recreation sites (Hanemann [58]; Feenberg and Mills [46]).

Where progress has been slower is in using the preceding techniques to value specific environmental benefits. Although many high-quality empirical studies have been published over the last 25 years, there are major gaps in the set of benefit values needed to evaluate environmental policies. In some cases, such as valuing changes in ecosystems, the problem is that the relevant outcomes are difficult to define. In other cases, the outcome is defined but is inherently difficult to value. For example, it is difficult to value reductions in mortality risks because the magnitudes of such risk reductions are difficult to communicate and because people do not think in terms of trading money for them. In other cases, benefit estimates exist for specific geographic areas (e.g., travel cost studies to value water quality benefits), but not for the entire country.

The discussion below reviews the major accomplishments that have been made in valuing environmental benefits during the past 25 years. Each section highlights the major achievements, both theoretical and empirical, that have been made in valuing each category of benefits. Then, remaining research needs are discussed.

3. VALUING HEALTH BENEFITS

Improvements in human health are regarded as the most important benefits of air quality regulations, drinking water regulations, and regulations affecting hazardous waste disposal. They are also an important category of benefits associated with pesticide regulation and are the primary reason for regulating toxic substances in consumption and production (e.g., asbestos). In the case of air quality improvements, health benefits include reductions in premature mortality, reductions in the incidence of asthma and chronic bronchitis, and reductions in the number of hospital admissions (and symptom days) experienced by people with chronic heart and lung disease. Air pollution has also been linked to episodes of acute illness, e.g., acute respiratory infections, respiratory restricted activity days, and respiratory symptom days. Cancer is the health effect most often associated with exposure to toxic substances, either from hazardous waste sites, pesticide residues, or from hazardous sources of air pollution, such as lead smelters or coke ovens. Other health effects associated with exposure to toxic substances include fetal defects. Drinking water treatment may reduce water-borne infections (e.g., giardiasis) as well as cancers.

Studies that quantify the impacts of pollution on health typically measure impacts on mortality rates—the proportion of people in an age/sex cohort that die each year—and on the incidence of various diseases—cancers, chronic heart and lung disease, gastrointestinal illness, and acute respiratory illness. We discuss separately attempts to value reductions in risk of death and risk of illness.

Mortality Benefits

Main achievements. Twenty-five years ago, the value of preventing premature mortality was measured using the human capital approach; i.e., the value of reducing an individual's risk of death was measured by the corresponding increase in his expected lifetime earnings. The notion that it should be measured by what a person would pay for the risk reduction (holding expected utility constant) had been suggested by Schelling [105] and Mishan [86], but had not been developed formally, nor had many attempts been made to measure willingness to pay empirically.[2]

In 1974 M. Jones-Lee [68] formalized the concept of willingness to pay in the context of a static expected utility model and showed that WTP should equal the difference in utility between being alive and being dead, divided by the expected marginal utility of money. In the 25 years following Jones-Lee's contribution, economists have informed attempts to value reductions in mortality risks by extending the theory of mortality risk valuation. The theoretical basis for valuing reductions in risk of death is the life-cycle consumption model with uncertain lifetime (Yaari [128]), which has been used to examine willingness to pay for reductions in the conditional probability of dying at age t over the life cycle.[3] Shepard and Zeckhauser [107] were among the first economists to simulate the life-cycle model to examine how WTP for a change in the conditional probability of dying varies with age. Their finding—that WTP first increases, then decreases with age—suggests that empirical estimates of WTP for a change in risk of death should vary with the age of those affected.

WTP should also vary with the time between reduction in exposure and reduction in risk. A lag between the two may result from a latency period (in the case of exposure to a carcinogen, such as asbestos) or because health effects depend on cumulative exposure to a pollutant. As Cropper and Sussman [40] have shown, the life-cycle model implies that WTP for a future risk reduction should be discounted to the present at the consumption rate of interest—the rate at which a person is willing to substitute consumption tomorrow for consumption today. This result suggests that one could estimate a 40-year-old's WTP for a reduction in his risk of death 20 years hence (e.g., at the end of a 20-year latency period) by taking the WTP of a 60-year-old for a current risk reduction and discounting it over the 20-year period.

Economists (Conley [33] and Bergstrom [11]) also showed that the human capital approach is likely to understate WTP. Specifically, foregone earnings are a lower bound to WTP as long average utility of consumption exceeds marginal utility of consumption, a condition that holds for all strictly concave utility functions for consumption in excess of some minimum level. Replacing foregone earnings by WTP in government decision-making depended, however, on having reliable estimates of WTP for risk reductions. It was the development of hedonic labor market models that provided these estimates.

[2]A notable exception is Acton [2] who asked people what they would pay for ambulance service that would expedite treatment of heart attacks.

[3]The conditional probability of dying at age t is probability that a person dies between his t and $(t + 1)$st birthdays, given that he is alive on his tth birthday.

The use of compensating wage differentials to estimate WTP was first suggested by Thaler and Rosen [113], who applied Rosen's theory of hedonic markets to the labor market for the purpose of valuing mortality risks. Under competitive conditions, workers would locate along an equilibrium wage-risk locus at the point where their willingness to substitute risk for income just equaled the rate at which the market rewarded workers' willingness to accept additional risk of death on the job. This allowed the partial derivative of the hedonic wage equation with respect to fatal job risk to be interpreted as the marginal value of a risk change.

Thaler and Rosen's work paved the way for empirical estimates of compensating wage differentials. W. K. Viscusi, M. Moore, and others (see Viscusi [123, 124] for a summary) undertook the challenging task of obtaining credible estimates of hedonic wage equations. This has required matching micro datasets on wages and worker characteristics with estimates of fatal and non-fatal job risk, as well as compensation received for on-the-job injury. As J. P. Leigh [78] has noted, information on risk of injury on the job is usually available only at the industry level, which may make it difficult to control for inter-industry wage differentials that arise from other causes. Moore and Viscusi's use of injury rates [91] that vary by state as well as by industry, however, provides a potential solution to this problem.

As the hedonic wage literature was developing, economists also relied on two other approaches to value mortality risks. Averting behavior studies used expenditures of time (in the case of seatbelt use—Blomquist [13]) or money (in the case of smoke detector purchase—Dardis [41]), together with data on the size of the risk reduction achieved to infer the value of the risk reduction. These studies yielded smaller values for mortality risk reductions than the hedonic wage literature. The other approach was to ask people directly what they would pay for a reduction in their risk of dying, either on the job (Gerking *et al.* [49]) or in a traffic accident (Jones-Lee *et al.* [69]).

While contingent valuation studies of mortality risk reductions often yielded estimates of the value of a statistical life in the same range as hedonic wage studies, they also revealed people's difficulties in comprehending and valuing risk changes.[4] In the study of Jones Lee *et al.*, for example, 42% of respondents placed the same value on a 7 in 100,000 risk reduction as on a 4 in 100,000 risk reduction. Smith and Desvousges [111] found that respondents were sometimes willing to pay more for smaller risk changes than for larger ones. Similar insensitivity to the size of risk changes has been found by Hammitt and Graham [57] who report that WTP to reduce risk of death in an auto accident does not increase in proportion to the size of the risk change. This may either be due to a failure to correctly perceive the size of the risk change (32% of respondents in Hammitt and Graham's survey did not know that 9 in 100,000 was smaller than 1 in 10,000) or to a lack of familiarity in trading money for risk.

In spite of these difficulties, the emergence of a body of literature that reported values of mortality risk reductions within an order of magnitude of one another led to their use in benefit-cost analyses. In preparing "The Benefits and Costs of the

[4] The value of a statistical life is the sum of individuals' WTPs for risk reductions that sum to one statistical life. If each of 10,000 workers is willing to pay $500 to reduce his risk of dying by 1 in 10,000, the sum of the WTPs = $5,000,000. Because, on average, one statistical life is saved, the value of a statistical life is $5,000,000.

Clean Air Act 1970 to 1990" [117] the U.S. Environmental Protection Agency selected 26 studies of the value of mortality risk changes—21 hedonic wage studies and 5 contingent valuation studies—and used the average value of a statistical life (VSL) from these studies—$4.8 million 1990 USD—to value statistical lives saved by the Clean Air Act. This value is also suggested in the Agency's "Guidelines for Regulatory Impact Analyses" [118] as the central VSL estimate to be used in Regulatory Impact Analyses.

Remaining research needs. While hedonic wage equations may reliably estimate workers' willingness to accept (WTA) compensation for increased risk of death, there is concern that these values may either under- or over-state WTP for reductions in mortality risks provided by environmental programs. The main concerns here are: (1) that consumers' WTP for reductions in their risk of dying are lower than workers' WTA compensation to bear additional risk; (2) that the average age of workers in compensating wage studies differs from the ages of people who benefit from reductions in environmental risks; and (3) that the nature of the risks reduced by environmental programs are less voluntary than job risks and may be associated with prolonged illness.

With regard to the first point, studies that use consumers' purchases of safer products to infer WTP for risk reductions typically result in values that are an order of magnitude lower than the VSLs in labor market studies (Viscusi [123]). Although WTA compensation for bearing increased risk should, for small risk changes, equal WTP for risk reductions, the two may differ (Khaneman *et al.* [72]) In valuing an air quality regulation that increases the price of fuels or the cost of automobiles, it is consumers' WTP for the regulation that is the relevant concept.

With regard to age, the average age of workers in hedonic labor market studies is approximately 40. Many environmental regulations, however, benefit disproportionately either the very old or the very young. In the case of air quality improvements, for example, it is primarily older persons who benefit. "The Benefits and Costs of the Clean Air Act 1970 to 1990" [117] estimates the number of statistical lives saved by the Clean Air Act in 1990: the difference between the number of deaths that would have occurred in 1990 without the CAA and the number that actually occurred. This number—approximately 200,000—is distributed in proportion to the age distribution of deaths in the U.S. in 1990. This implies that three-quarters of the statistical lives saved are persons 65 years of age or older. There are, unfortunately, no estimates of the values of risk reductions to persons over 65 years of age. Simulations of the life-cycle model suggest that WTP for reductions in risk of death are lower at age 60 than at age 40, a result that is confirmed by Jones-Lee *et al.* [69].

This led the Council on Clean Air Act Compliance Analysis [35] to suggest that the USEPA apportion the $4.8 million value of a statistical life (VSL) over the average remaining life expectancy in the 26 studies—35 years—to calculate a value per life-year saved. The value of statistical lives saved at different ages could then be computed based on remaining life expectancy. This approach is, however, not fully satisfactory as there is no reason to believe that WTP for reductions in risk of death is proportional to remaining life expectancy.

What is required is a large-scale study that values risk reductions among older persons. Contingent valuation is the most likely approach to take in valuing risk reductions among the elderly, however, this will require successfully communicating the magnitude and timing of risk changes to respondents and making them comfortable with trading money for quantitative risk changes (Krupnick *et al.* [76]). If successful, such a survey could also address the WTP/WTA disparity. The survey could also provide information as to how quality of life (extent of functional limitation) affects WTP as people age.

Further research is also required on the value of mortality risk reductions to children. Here, two approaches are possible: (1) attributing to a child the value he would pay if he knew his future earnings and could make rational decisions, and (2) using parents' willingness to pay (WTP) to reduce risks to their children. The latter approach is what has been followed in valuing reductions in morbidity to children (Agee and Crocker [8]). In addition, it is important to consider societal effects of reducing mortality rates among children that may not be reflected in parents' WTP.

Finally, a remaining issue for research is how WTP for mortality risk reductions is affected by the voluntariness of the risk or by the cause of death itself. Much has been written in the psychological literature about the importance of psychological aspects of risk—dread, controllability, voluntariness, blame—in explaining how people prioritize human health risks (Slovic *et al.* [109]). These characteristics of risk may well influence WTP for changes in risk of death, although Cropper and Subramanian [39] have found that they do not have much influence on people's choices between life-saving programs when people are given information about the number of lives each program saves.[5] With regard to cause of death, it may be possible to separate the morbidity associated with an illness (e.g., a case of cancer that involves radiation therapy) from the shortening of life that the illness causes and value the two separately.

Morbidity Benefits

Main achievements. In 1974 the most common approach to valuing reduced morbidity was to compute the avoided Costs of Illness (Cooper and Rice [34]). These include the value of medical expenditures avoided and the associated gain in productive time. While the Cost of Illness approach to valuing morbidity is sometimes used by default, Cost of Illness estimates are now widely regarded as inferior to measuring what individuals would pay to avoid illness or the risk of illness.

The triumph of WTP over the Cost of Illness may be traced to utility-theoretic models of health production that allowed an individual's WTP for reduced illness to be compared to changes in income and medical expenditures. For commonly occurring illnesses, the household production model has provided a convenient framework for analyzing the value of changes in health or in pollutants that affect health. M. Grossman [55] was the first economist to include a health production function in the household production model and time spent ill in an individual's utility function and budget constraint. Harrington and Portney [62] extended

[5]Cropper and Subramanian [39] confronted respondents with pairs of life-saving programs that differed in the number of lives saved and in their psychological risk attributes, which respondents rated on 10-point scales. While the psychological aspects of risk were statistically significant in explaining respondent choices, the rate of substitution between lives saved and psychological risk characteristics in respondents' utility functions was quite low.

Grossman's model to allow exposure to a pollutant to enter the health production function and to allow exposure to depend on averting behavior.

Harrington and Portney showed that WTP for a marginal change in the pollutant could be written as the sum of four components—the value of income lost while ill, the change in expenditures on medical care, the change in expenditures on averting goods, and the disutility of time spent ill. This result implies that, under reasonable assumptions, the Cost of Illness approach to valuing morbidity, which counts only productivity losses and medical costs, is a lower bound to WTP. This has been a significant result for policymakers, who, when forced to rely on Cost of Illness estimates, can claim that they underestimate the appropriate value of reduced morbidity.

The household production model has also provided a means of estimating WTP for a reduction in pollution by using expenditures on averting or mitigating inputs. As Courant and Porter [36] showed, the value of a marginal change in pollution equals the marginal rate of substitution in the health production function between pollution and an averting (or mitigating) input times the price of the input. The value of a non-marginal change in pollution can be bounded by the area between the marginal cost of healthy time before and after the pollution change, evaluated at the current level of healthy time (Bartik [10]).

Although there have been several attempts to implement the averting behavior approach empirically (Gerking and Stanley [50]; Dickie *et al.* [42] and Chestnut *et al.* [28]) it has not become the primary source of estimates of the value of reduced morbidity. This is because most averting inputs (air-conditioners, bottled water) produce joint products and because others (staying inside to avoid air pollution) have prices that are hard to measure.[6]

The main source of estimates of the value of reduced morbidity for benefit-cost analyses has been contingent valuation studies. For valuing minor respiratory illness (days of upper or lower respiratory symptoms, minor restricted activity days) most policy analyses rely on contingent valuation studies that were conducted in the 1970s and 1980s. Three studies (Loehman *et al.* [80], Tolley *et al.* [114], and Dickie *et al.* [43]) valued minor respiratory symptoms. Respondents were presented with a description of a symptom complex and WTP was elicited using either an open-ended question or payment card. Rowe and Chestnut [102] valued reductions in symptom frequency for asthmatics (a 50% reduction in the number of bad asthma days experienced each year) while Chestnut *et al.* [28] asked patients with angina what they would pay to reduce by one the number of angina episodes experienced per month. A recent meta-analysis of these studies (Johnson *et al.* [67]) attempts to link WTP to quality of well-being (QWB) indices, which allow different illnesses to be compared along a common scale.

To value reductions in the risk of contracting a chronic illness such as chronic bronchitis or asthma, economists have also relied on stated preference methods. Studies that have asked people directly to trade income for reduced risk of illness (e.g., Lin and Milon [79]) often find that WTP does not increase in proportion to the size of the risk reduction. A more successful approach has been confronting respondents with risk–risk trade-offs (Viscusi *et al.* [125]). Respondents are asked to choose among different combinations of risk of death (e.g., in an auto accident)

[6]Notable exceptions are Abdalla [1] and Harrington *et al.* [61] who use averting expenditures to value water pollution.

and risk of chronic illness (e.g., chronic bronchitis). This allows the researcher to relate the value of a statistical case of chronic bronchitis to the value of a statistical life. Although some respondents have exhibited inconsistencies in making risk–risk trade-offs, the value of avoiding chronic bronchitis using this approach has been found to be robust to different information treatments (Krupnick and Cropper [75]).

Because estimates of WTP to avoid risk of serious illness do not exist for all diseases, policy makers are sometimes forced to rely on Cost of Illness (COI) estimates.[7] Bartel and Taubman [9] and Cropper and Krupnick [38] have provided estimates of the impact of chronic heart and lung diseases on labor force participation and earnings. The medical costs associated with various chronic illnesses have been estimated by Salkever [104], Oster *et al.* [93], and Cropper and Krupnick [38].

Remaining research needs. The main shortcoming in the area of morbidity valuation is the dearth of empirical studies that value morbidity endpoints of interest. As noted above, revealed preference approaches have been less successful in valuing morbidity than in valuing mortality risks. Conducting stated preference studies requires original data collection, which is often expensive. Unfortunately, the five contingent valuation studies that are used to estimate WTP for morbidity reductions associated with air pollution were conducted in the early days of the contingent valuation method and suffer from serious shortcomings (Cropper and Freeman [37] and Johnson *et al.* [67]). Studies of WTP to reduce risk of chronic illness have been estimated for only a few diseases. This has forced policymakers to rely on COI estimates for most cancers and for some of the more serious morbidity effects associated with air pollution.

Given the expense of conducting stated preference studies for morbidity endpoints, it is important to try to prioritize research needs. Mortality benefits have, thus far, dominated estimates of the value of reducing air pollution. A value of information analysis for the setting of air quality standards (Evans *et al.* [45]) suggests a large payoff to refining estimates of the value of reducing mortality risks. If, however, these estimates should be found to be smaller than those currently in use, the value of avoiding chronic illness would become a more important benefit category.

4. VALUING THE BENEFITS OF RECREATION AND WATER QUALITY

Recreation benefits constitute one of the largest categories of benefits from improving surface water quality. Improvements in water quality and/or in fish catch may increase participation in water-based recreation and can enhance the value of a recreation experience. They are one of the most important benefits associated with the Clean Water Act and are also an important source of benefits in natural resource damage cases under CERCLA. Recreation benefits are also important in the multiple use management of forests and river basins. In these cases the issue is often one of valuing a site per se—assessing the value of creating

[7] If people do not consider the impact of chronic disease on labor force participation, earnings, and medical costs when expressing their WTP to avoid serious illness, then COI estimates must be added to WTP estimates.

a lake when a river is dammed or of protecting a wilderness area—rather than valuing the impact of changes in environmental quality.

To limit the scope of the discussion, we focus on recreation benefits associated with improvements in water quality. Because the nonuse benefits of improvements in water quality are often difficult to separate from use values, we discuss the techniques that have been used to value both types of benefits.

Main achievements. By 1974 the travel cost method had been developed (Clawson and Knetsch [31]) and used to estimate the value of recreation sites (Knetsch [73]); however, it had not been used to value site quality. It was K.-G. Mäler's achievement [82] to show that a shift in the demand for site visits in response to a change in the quality of the site could be used to value the quality change. Mäler showed that the change in consumer surplus corresponding to the shift in the demand curve measures the value of site quality if site quality is weakly complementary to the demand for site visits, specifically, if (a) there is a choke price above which site visits are zero, (b) the marginal utility of site quality is zero if the site is not visited.

Empirical applications of the weak complementarity approach were hampered by difficulties in predicting how the demand for sites visits shifts in response to a change in site quality: Since all visitors to a site experience the same level of dissolved oxygen or average fish catch, it is difficult to estimate the impact of quality on site visits with a single cross-section of data on visits to a single site. Solutions to this problem were proposed by Vaughan and Russell [121], using data on visits to a set of sites. Multiple site data allow the researcher to interact site quality with income and travel cost, variables that vary across visitors to a site. This approach, termed the varying parameters model, was used by Smith *et al.* [112] and by Smith and Desvousges [110] to value changes in water quality at recreation sites.

Over time, discrete choice models have become the most popular variant of the travel cost method used to value site quality. One reason for this is the inability of the varying parameters model to incorporate the prices of substitute sites. The Random Utility Model, which assumes that each person chooses the site he will visit by comparing the utility of a set of sites, allows travel cost and quality variables for each site to be incorporated in a straightforward manner. Early applications of the random utility model (RUM) (Hanemann [58] and Feenberg and Mills [46]) used a multinomial logit model to explain the choice of site to visit on a given day. Visits on different days of the season were assumed independent of one another, and the total number of visits made was explained separately.

More sophisticated versions of the random utility model (Carson *et al.* [24], Parsons and Kealy [95], and Bockstael *et al.* [14]) have been developed that simultaneously estimate the participation and site choice decisions and that allow site choices on different days of the season to be determined simultaneously. At the same time, Hanemann [59] spelled out how compensating and equivalent variation for changes in site attributes could be measured in the random utility framework.

From a policy perspective, the important question is what site attributes have been valued using travel cost models and how these attributes can be related to the characteristics of water quality affected by pollution regulations. A large portion of the travel cost literature has focused on recreational fishing. The site attributes in some models include average fish catch, by species (Carson *et al.* [24]). These models have been quite useful in site management decisions and in allocating fish between commercial and recreational uses; however, to inform water quality decisions, changes in fish catch must be linked to water characteristics (e.g., dissolved oxygen, fecal coliform count, etc.). Other models of recreational fishing (Smith and Desvousges [110]) include water quality measures (e.g., dissolved oxygen) directly in the model.[8]

Travel cost models have also been used to value water quality at beaches (Hanemann [58] and Bockstael *et al.* [14]) and water pollution associated with acid rain and toxic substances. The benefits of reducing acid rain to fishermen at lakes in the Adirondacks have been estimated by Mullen and Menz [92], Morey and Shaw [89], Englin *et al.* [44], and, most recently, by Montgomery and Needelman [88]. The latter study was used most recently by the USEPA [119] to value the benefits of reduced acid rain associated with Title IV of the 1990 Clean Air Act Amendments. The impact of toxic water pollution that can result in fish advisories (i.e., that fish are unsafe to eat) has been studied by Montgomery and Needelman [88] in New York state and by Jakus *et al.* [66] in Tennessee.

One drawback of the travel cost approach is that it cannot capture non-use values. This has led researchers to value water quality benefits using stated preference approaches which, in principle, can capture both use and non-use values. Because it is difficult to ask people to value changes in dissolved oxygen levels or fecal coliform counts, many contingent valuation studies have used a water quality ladder that indicates whether water is boatable, fishable for rough fish, fishable for game fish, or swimmable. The ladder, first developed by Vaughan and Russell [120], has been used to value water quality improvements at specific sites, such as the Monongahela River (Smith and Desvousges [110]) and throughout the U.S. (Carson and Mitchell [25]).

Carson and Mitchell's study, which has been used by the USEPA to value national water quality improvements, asked a national random sample of households to value improvements in water quality from boatable to fishable and from fishable to swimmable throughout the U.S. Respondents were then asked to allocate some fraction of this value to water quality improvements in their own state. These figures have in some cases been used to value water quality improvements along specific river segments by multiplying the state-wide value by the proportion of total miles of river in the state represented by the segment in question.[9] Plans are underway to update the Carson–Mitchell survey, originally conducted in 1988, using a computer-administered survey instrument developed by Magat *et al.* [81].

Remaining research needs. The main question for policy is whether existing estimates of the use and non-use values associated with water quality improvements are adequate to make informed environmental management decisions. The stated preference studies funded by the USEPA (Carson and Mitchell [25] and

[8]Yet another approach is to view changes in water quality as creating or destroying recreation sites. Vaughan and Russell [122] adopt this approach in valuing the benefits of the Clean Water Act. It is, however, less likely to be applicable when valuing marginal changes in water quality. As we emphasize below, it is in valuing marginal policy changes that economists' analyses can be most useful in informing policy.

[9]WTP for water quality improvements outside one's own state would be similarly apportioned to a given river segment.

Magat *et al.* [81]) are designed to measure the benefits of national improvements in water quality—to allow the EPA to compare the benefits with the costs of the Clean Water Act on a national scale. Benefit-cost analyses at this level of aggregation are, however, of limited usefulness in improving environmental policy. As Freeman [48] has noted, efficient water pollution control policy would replace the national goal of fishable and swimmable water quality with ambient standards (e.g., for dissolved oxygen, fecal coliform) that vary across water bodies and river segments. These standards would be based on a comparison of marginal benefits and marginal costs.

In assessing research needs we must therefore ask whether existing estimates of the use and non-use values associated with water quality improvements (a) can be linked at the local level to the attributes of site quality that are relevant to environmental management decisions and (b) can be estimated for enough sites to be useful in informing pollution control decisions.

At present, the existing set of travel cost studies is not comprehensive enough—either geographically or in terms of the measures of water quality valued—to provide the answers needed for policy decisions. One way to develop a comprehensive set of travel cost models for valuing water quality benefits would be to link a national recreation survey to measures of water quality that are available throughout the country. Such an effort has been undertaken by the USEPA, which has funded a model (RF1) to link point source emissions along 633,000 miles of rivers and streams to measures of water quality (dissolved oxygen, total suspended solids, fecal coliform). These measures have, in turn, been linked to Vaughan and Russell's water quality ladder (Bingham *et al.* [12] and Bondelid *et al.* [16]). In a pilot effort, George Parsons has estimated travel cost models for New England by linking data from the 1993 National Recreation Survey to the EPA's water quality data, measured using the water quality ladder.

An important question for research is whether this approach should be pursued on a national level and, if so, what measures of water quality should be used. The advantages of a uniform approach to valuing recreational use values is clear—values would be consistent, estimated using models that are comparable across locations. There are, however, issues as to how water quality and other site attributes should be measured. Can a uniform set of site attributes be chosen that would be applicable in all locations? An alternate approach would be to conduct perhaps a dozen "Cadillac" travel cost studies that used site-specific data, informed by on-the-ground knowledge of the sites, that valued the same set of water quality characteristics. Benefits transfer procedures could then be used to apply values from these studies to other stretches of river.

If the travel cost approach were to be used to value the use values associated with water quality improvements, how would non-use values be measured? Contingent valuation can, of course, be used to value water quality improvements; however, it is usually impossible to separate use from non-use values in a contingent valuation survey. A possible solution to this problem is to estimate non-users' WTP for water quality improvements (i.e., the WTP of people who do not engage in water-based recreation) and use this as a lower bound to the non-use values of the general population.

Another possible research strategy would be to try to obtain both use and non-use values in a stated preference study, but one that linked values more closely to the geographic sites where improvements would be made. Although neither

Carson and Mitchell [25] nor Magat *et al.* [81] are designed to value changes in water quality along particular rivers, it would indeed be possible to tailor the valuation scenario to specific rivers near the respondent's home using the RF1 database.

5. VALUING BENEFITS OF IMPROVED VISIBILITY

Improved visibility, both locally and throughout the country, is an important benefit of reducing air pollution. Twenty-five years ago, policy makers had no estimates of the value of improved visibility, unless one interprets hedonic property value studies as providing an upper bound on the value of visibility improvements. During the past 25 years a body of work has developed that values local visibility improvements as well as improvements in visibility in national parks.

Main achievements. By 1974 economists had begun to examine the impact of local air pollution on residential property values (Ridker and Henning [100]). Work by Rosen [101] and Freeman [47] allowed the gradient of the hedonic price equation with respect to air pollution to be interpreted as consumers' WTP for a marginal change in air quality; however, it was unclear what categories of benefits this captured. While most economists claimed that changes in property values captured the aesthetic benefits of cleaner air (Trijonis *et al.* [116]), including improved visibility, other economists interpreted residential property value studies as valuing health effects (Portney [97]). In reality both were correct.

The difficulty in using housing prices to value air quality benefits, however, is that it is unclear that they capture *all* of the health benefits of air quality. To assert this, one would have to argue that people are very well educated about the health risks of dirty air. It is more likely that property values capture all of the aesthetic benefits of improved air quality (including improved visibility) and a portion of the health benefits. This makes it difficult to add these values to independently valued health benefits or to use them as a comprehensive measure of the value of health and visibility benefits.

This realization has led to the use of stated preference techniques to value local air quality improvements. One advantage of the stated preference approach is that through the use of conjoint analysis, one can vary not only visibility but the health effects associated with it. It is thus possible to separate the values attached to improved visibility from the values attached to health improvements.[10] Unfortunately, early studies of the benefits of improved residential visibility did not adequately account for health effects, and, like the morbidity studies cited above, used contingent valuation methods that would be judged unacceptable by current standards. Unfortunately, the majority of studies that value local visibility improvements (Rae [98] and Tolley *et al.* [115]) date from this period.[11] More recent studies (Carson *et al.* [26] and McClelland *et al.* [85]) cover only three cities (Cincinnati, Chicago, and Atlanta).

[10]An alternative approach is to ask people to allocate WTP for improved air quality between improved visibility and better health.

[11]Tolley *et al.* [115] valued visibility improvements in 6 eastern cities—Chicago, Atlanta, Boston, Mobile, Washington, DC, Cincinnati, and Miami; Rae [98] valued improvements in air quality in Cincinnati. Their data were reanalyzed by Chestnut and Rowe [29].

Studies have also valued improvements in visibility in national parks in the southeastern and southwestern U.S. Early studies focused on the Grand Canyon (Schulze and Brookshire [106]) and found extremely large average WTP values for all households in the U.S. This value, however, was much lower (Tolley *et al.* [115]) when households were asked to value local improvements in visibility at the same time as improvements in visibility at the Grand Canyon. The most recent research in this area, the National Parks Visibility Value Study improves upon previous studies by asking respondents to value visibility improvements at all national parks in their region, and by attempting to separate WTP for park protection from WTP for improved visibility (Chestnut and Rowe [30] and Chestnut and Dennis [27]). The regions covered in the study include California, the Southwestern U.S., and the Southeastern U.S.

Remaining research needs. There are few studies using modern contingent valuation methods that value visibility improvements, especially local visibility improvements. In order to avoid the "adding up" problem, it would seem appropriate to ask a national random sample of households to value visibility improvements where they live and, at the same time, improvements in visibility at major recreation sites in their region and throughout the U.S. Care would have to be taken to separate health effects from visibility improvements when valuing local visibility changes, and, similarly, to separate WTP to preserve recreation areas from WTP for visibility improvements at these areas.

6. VALUING ECOLOGICAL BENEFITS

If one takes an anthropocentric approach to valuing environmental benefits, then ecological benefits include changes in all of the ecosystem services that humans value. For the purposes of valuation, it is convenient to categorize the impact of pollution on ecosystems as follows: (1) impacts that occur through markets (e.g., impacts of pollution on commercial timber stands or fish populations); (2) impacts that affect recreation (e.g., damage to national parks from air pollution or to recreational fishing from oil spills and other forms of water pollution); (3) impacts on ecosystems for which people have well-defined non-use values (e.g., damage to remote forests or to groundwater, the value of reduced fish populations to non-anglers); (4) impacts on ecosystem services that are not readily understood by laypersons—water and nutrient recycling, storage of biodiversity, climate stabilization.

To an economist, valuing the first category of ecosystem benefits is easy. As long changes in pollution can be translated into changes in timber yield or in commercial fish harvests, the latter can be translated into changes in consumer or producer surplus. These studies are discussed in Section 7. The second category of benefits can be handled using the recreation demand models described in Section 4. The challenge is to value ecosystem changes in categories (3) and (4).

Main achievements. Before 1974, there were no empirical studies that measured the non-use values associated with damages to ecosystems. Since that time a large literature has developed that can be classified according to the nature of the endpoint valued:

(1) There have been studies conducted to value damages to ecosystems that are the subject of natural resource damage cases. Examples include the Exxon Valdez case (Carson *et al.* [23]) and the Nestucca oil spill (Rowe *et al.* [103]).

(2) There have been studies to estimate the value of preserving specific ecosystems, such as wetlands (Whitehead and Blomquist [127]) or wilderness areas (Walsh *et al.* [126]).

(3) Studies have measured WTP to improve visibility in National Parks (see Section 5) and at other sites.

(4) Studies have measured WTP to improve surface water quality (see Section 4) and groundwater quality (Boyle *et al.* [20]).

(5) Studies have valued damages to forests that could be associated with air pollution (either acid rain or ozone). Peterson *et al.* [96] valued aesthetic damages to forests in southern California that could be caused by ozone, while Holmes and Kramer [65] examined WTP to prevent damage to forests in North Carolina.

(6) Finally, there are over half a dozen studies (see Randall [99] for a summary) that estimate the value of preserving particular species, such as the bald eagle, striped shiner, grizzly bear, whooping crane and bighorn sheep.

How useful have these studies been in informing environmental management decisions? As Kerry Smith notes, studies in category (1) have been used in actual court cases, as have some studies in category (3). Studies that value preserving particular ecosystems could be used in land use planning decisions. Studies such as Boyle and Bishop's estimate [19] of WTP to preserve the bald eagle in the state of Wisconsin could be useful for environmental management decisions at the state level.

For the purposes of regulating air, water, and hazardous waste pollution on a continuing basis, however, what is needed are studies in categories (3)–(5). The issue, as it has been throughout this paper, is (a) what to endpoint value and (b) how to link the endpoint to specific pollutants. The endpoints that have been valued—water that is boatable–fishable–swimmable in the case of water quality and changes in visual range or in leaf damage to trees—are clearly ones that people can perceive. The main difficulties in valuing them are in deciding on (a) the geographical scope of what is to be valued (Are people asked only for non-use values in their state?); (b) the nature of substitutes (i.e., conditions at other locations); (c) how many endpoints to value at the same time. In regard to the latter, if a regulation to reduce nitrogen oxides (NOx) affects forests through ozone and fish population through acid rain, people should be asked to value the entire package of ecosystem benefits brought about by NOx reduction. Adding up WTP values from individual studies will most likely overstate the value of the NOx reduction program (Hoehn and Randall [64]).

Remaining research needs. Existing studies of non-use values associated with air and water pollution are limited in number and in geographic scope. The main ecological endpoints that have been linked to air and water pollution, for which people could be expected to have well-defined non-use values, include aesthetic damages to forests (from acid rain and deposition of toxic air pollutants such as mercury and dioxin), damages to fish populations (from nitrogen saturation, eutrophication, and heavy metals) and wildlife populations (from air toxics). This would suggest further studies are needed of these WTP to prevent damage to these endpoints; however, future studies must adequately handle the problem of describing substitutes for the good being valued, and they must deal with the "adding up" problem.

A problem for policy analysis is that these endpoints do not capture the totality of ecosystem damages associated with pollution control decisions. In particular, they do not capture services of nutrient recycling and habitat provision. Nor do they capture the more subtle changes in ecosystem functioning that may to lead to non-marginal changes in ecosystem performance. Should we try to have people value these more subtle endpoints? The answer, in my opinion, is no. Incorporating these more subtle effects into pollution control decisions requires characterizing ecosystem outcomes (or indicators) that are important to ecosystem functioning and then relating these outcomes (or indicators) to particular activities or pollutants. This information is essential before any attempt is made at valuation.

7. VALUING REDUCED MATERIALS DAMAGE

Materials damage is usually defined as damages to buildings and structures (e.g., from air pollution or acid rain), as well as damages to biological resources (fish, forests, and agriculture) that make up the "material" environment (Adams and Crocker [5]). Reducing ground-level ozone can improve crop yields and reduce damage to timber. Reducing nitrogen deposition or toxic water pollutants can improve commercial fish yields.

Main achievements. The main empirical contributions that economists have made in this area have been to measure the impacts of ground-level ozone on field crops and forests. In some cases, a damage function approach has been used; i.e., impacts of ozone on crop yield, predicted from controlled experiments, have been used to estimate changes in consumer and producer surplus for the crop in question (Adams *et al.* [6] and Shortle *et al.* [108]). Most models, however, allow both for input and output substitution by farmers (Adams *et al.* [4] and Kopp and Krupnick [74]). Adams *et al.* [3] have studied the impacts of acid rain on soybean production. In the area of forestry, the USDA's TAMM model (Adams and Haynes [7]) has been used to estimate the benefits of reducing ground level ozone on commercial timber yields [119].

Fewer studies have been conducted to value the impacts of air pollution on buildings and household soiling. Two studies that are repeatedly cited value the impact of sulfur dioxide on building materials (Lareau *et al.* [77]) and the impact of SO_2 and particulates on household soiling (Manuel *et al.* [83]). The latter is an ambitious attempt to examine the impact of air pollution on a system of household expenditure equations for cleaning and personal care.

There are also a few studies of the impacts of water pollution on commercial fisheries, especially those that link changes in a pollutant to changes in fish yield or fish quality and then value the latter. A notable exception is Kahn and Kemp [71] who examined the impact on the Chesapeake Bay striped bass fishery of submerged aquatic vegetation. The latter can, in principle, be linked to nutrient loadings in the Bay.

Remaining research needs. The economics of valuing materials damage is straightforward; however, there are areas in which additional empirical studies would be useful. These include updated studies of the value of reduced damage to buildings and reduced household cleaning expenditures associated with air quality improvements. In the case of water pollution, few studies exist of the impact of water pollution on commercial fisheries.

8. VALUING THE COST OF ENVIRONMENTAL PROGRAMS

In most regulatory impact analyses, the direct costs of pollution regulations are estimated using an engineering cost approach. The engineering cost approach may overstate direct compliance costs to a firm or household if it fails to correctly anticipate technical improvements that will occur in the future, or if it fails to take account of substitution possibilities that can be used to reduce compliance costs. Economists can shed light on the extent of these problems by (a) providing econometric estimates of compliance costs that allow for input and output substitution and (b) comparing ex ante and ex post estimates of compliance costs. Econometrically estimated marginal abatement cost functions exist for SO_2 abatement by electric utilities (Gollop and Roberts [51], Coggins and Swinton [32] and Carlson *et al.* [22]) and for water pollution abatement by the pulp and paper industry (Gray and Shadbegian [54] and McClelland and Horowitz [84]). Studies that compare ex ante and ex post estimates of the direct costs of regulations include Hammitt [56] and Morgenstern *et al.* [90].

The direct costs of a regulation to a firm or household are not, of course, an estimate of the social costs of the regulation. In the last 25 years, economists have made important advances in estimating regulatory costs in a general equilibrium framework using appropriate welfare measures (Jorgenson and Wilcoxen [70] and Hazilla and Kopp [63]). General equilibrium models capture three types of costs not included in estimates of direct compliance costs: substitution effects that result from the price changes associated with environmental regulations (e.g., the substitution of "clean" for "dirty" goods in consumption, the substitution of leisure for labor as goods become more expensive), investment effects, and effects on productivity growth. Jorgenson and Wilcoxen [70] find that the net effect of these forces over the period 1974–1985 was to reduce real GDP by 1.7 percent in 1985, a cost that must be added to the direct compliance cost of environmental regulation.[12]

One of the most important insights to emerge in the past 25 years is that environmental regulations, by exacerbating existing distortions in the economy, can have social costs considerably in excess of direct compliance costs. An environmental regulation that raises the price of purchased goods and lowers the real wage, will, other things equal, cause a substitution of leisure for labor. This compounds the deadweight loss of the income tax system, which, by driving a wedge between the gross and net of tax wages, causes individuals to substitute leisure for labor

[12] Hazilla and Kopp [63] likewise find that the social costs of the Clean Air Act and Clean Water Act exceed expenditures on abatement by consumers and firms due to reductions in investment and labor supply.

(Bovenberg and de Mooij [17], Parry [94], and Bovenberg and Goulder [18]). This tax interaction effect can, in some cases, double the costs of a regulation (Goulder *et al.* [53]). The tax interaction effect can to some degree be offset if the environmental program raises revenues, which are used to reduce the rates of other, distorting taxes.

The literature on second best environmental taxation implies that the costs of a program will depend on how a standard is achieved. For example, a permit market will have lower social costs if permits are auctioned and revenues recycled than if permits are given away (Goulder *et al.* [52]). It also implies that tax interaction effects can be substantial (as a percent of direct costs)—even for small programs.[13]

9. CONCLUSIONS

One theme that emerges clearly from the preceding review, and from discussions with policy makers, is the need for a greater number of applied studies to value the benefits (and costs) of pollution control. This raises at least three issues: (1) Which studies do policy makers need the most? (2) Who will fund these studies? (3) How will the studies be peer-reviewed and circulated to the profession?

In answer to the first question, it is the categories of benefits that are likely to drive benefit-cost analyses—to make the greatest difference to environmental management decisions and bring about the greatest change in net social benefits—that should be undertaken first. To answer this question would, of course, require a full value of information analysis for all major environmental regulations. Without conducting such an analysis, however, it is often clear what are the "big ticket" items that drive environmental benefits.

Which topics receive the greatest attention from researchers will depend in part on funding. The logical source of funding for applied, policy-oriented research is the agencies that use this research. In allocating funds, agencies implicitly reveal their research priorities. There is, however, sometimes a tension between the types of research that agencies need and the research that is considered prestigious from a professional standpoint. Articles that develop new techniques have a higher chance of being published than studies that do not have a methodological "twist," even if the latter are carefully done.

This relates directly to question number three. If policymakers require more, applied studies, and if such studies are difficult to publish, how will it be possible for policy to rely on peer-reviewed articles? Is this possible given the current set of journals in environmental economics, or will some changes have to be made?

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[13] This point deserves emphasis. One usually thinks of general equilibrium effects as important only for "large" regulations. Tax interaction effects, however, exist even at the margin.

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