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10 Demands for Data and Analysis Induced by Environmental Policy

Clifford S. Russell and V. Kerry Smith

10.1 Introduction

Economic analysis of environmental policies is, if not uniquely, at least unusually difficult. Resolution of these difficulties requires substantial investment in data collection and model construction, only some of which is directly economic. Some of the reasons for the difficulties of environmental benefit-cost analysis are well known, appearing in intermediate and even elementary microeconomic and policy analysis texts (Baumol and Oates 1975; Fisher 1981). Thus, even the average economics undergraduate major can be expected to appreciate that there is a problem finding demand functions for many services of the natural environment because they are public goods. At more advanced levels, they will learn about such thorny technical issues in implementing proposed solutions to this problem. Those interested in policy learn about the conflicting maze of environmental legislation, including problems of overlapping jurisdiction, differences in burdens of proof, and, most significantly, disagreements between laws over what role, if any, economic analysis should play.

But neither the technical economic matters nor the special policy problems, challenging though they may be, provide the principal explanation for our assertion that the benefit-cost analysis of environmental policy may well be uniquely difficult. Rather, that assertion is based on the central place in such

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analyses of the complex relationship between policy implementation choices on the one hand and the relevant natural systems (especially atmosphere, water bodies, soil and resident plant communities, and ground water) on the other.[1]

To set the stage for a more careful examination of this assertion, let us consider the nature of the system of environmental regulation and the origin of the complications in which we are especially interested. Figure 10.1 combines an overview of the linkage between policy choice and resulting benefits, with indications of the complications arising at each stage in the linkage. In the next three sections we shall examine in turn each of the links in the figure. In section 10.2, we shall describe some of the problems implied by the way standard setting is constrained and practiced, and by the necessity of choosing an accompanying implementation plan. In section 10.3, we concentrate on the central role of knowledge of natural systems interacting with choice of implementation system. In section 10.4, we come to some of the more obviously and traditionally economic issues connected with valuing environmental services.

The final section of the paper brings together the analysis of sections 10.2–10.4 with a brief assessment of key emerging policy issues to produce our version of a catalog of data-gathering strategies likely to be most relevant and valuable for analyzing future decisions on the allocation and management of environmental resources.

While our approach to identifying data needs builds from specific examples of current policy issues, the questions raised are general ones. Thus, we do not attempt to catalog what we consider to be the most important environmental policy issues in the late 1980s and then base an evaluation of data requirements on them. Instead we argue that the interactions between the statutes defining the character of environmental policy and the role of natural systems for economic agents' behavior affect the problems that would appear on any list that might be composed. Thus, regardless of whether one believes global warming or indoor air pollution is among the most pressing environmental questions, economic analysts will need to incorporate what is known about a form of these interactions in developing their analyses.

10.2 Choosing Standards and Implementation Plans

Table 10.1 (adapted from U.S. Environmental Protection Agency [EPA] 1987a) summarizes the major criteria to be considered by the administrator of EPA in deciding on standards under a variety of legislative mandates. Two features of this table are especially striking. First, the criteria used to choose standards frequently focus on a subset of the information that would be part of a full benefit-cost analysis. For example, under the Clean Air Act the primary standards for criteria air pollutants are to be based on human health effects but cannot include compliance costs in the process of defining the stan-

Logical Structure for Analysis	Rationale for the Logical Structure
Definition of technology, discharge or environmental quality standard by type of pollutant and media Development of implementation plan to meet standard Change in effluent loadings	The institutional structure governing the definition and implementation of environmental policy is complex. As a result of multiple, overlapping statutes defined by both environmental media (e.g., Clean Air Act and Clean Water Act) and the types of residuals generated (e.g., Resource Conservation and Recovery Act, Comprehensive Environmental Response, Compensation and Liability Act, etc.), policies must be responsive to multiple objectives. Moreover, they can involve the definition of standards in a format inconsistent with available environmental data or in generic terms that require considerable judgment to implement, enforce, or evaluate.
↓ POLLUTION DISCHARGES TO THE ENVIRONMENT	
Changes in one or more dimensions of resources Changes in ecological habitat and non-human species Change in human species	The services of environmental resources are produced within a complex physical system where the effects of different patterns and types of uses depend on temporal aspects of those uses. In particular, the pattern of environmental quality corresponding to a chosen standard is influenced by the implementation program to be used to attain the standard.
↓ AMBIENT ENVIRONMENTAL QUALITY CONDITIONS (SERVICE LEVELS)	
Economic agents change their patterns of consumption of other related goods or their use or valuation of environmental resources	The services of environmental resources exchange outside markets, and therefore, the information normally present from market exchanges is not available. Indeed, as part of their ordinary consumption choices, individuals may not have been required to consider changes comparable to those envisioned in any specific policy analysis. Information on the quality and character of these services can be quite technical, involve subtle measurement problems, and is unlikely to be generated through the informal processes individuals use to learn about other commodities they consume.

Fig. 10.1 Schematic description of issues in using economic analysis for environmental policies

Table 10.1 Components of Economic Analysis Identified in EPA's Enabling Legislation

	Benefits			Costs		
Economic Legislation/Regulation	Human Health	Other[b] Effects	Welfare[c]	Compliance	Cost Effective	Impacts
Clean Air Act:						
Primary NAAQS[a]	x	x				
Secondary NAAQS			x			
Hazardous air pollutants	x					
New source performance	*	x	*	**	**	**
Motor vehicle emissions[d]	x	x	x	x	x	x
Fuel standards[d]	x	x	x	x		
Aircraft emissions	x	x	x	x		x
Clean Water Act:						
Private treatment		e	***	x	x	x
Public treatment		e				
Safe Drinking Water Act:						
Maximum contaminant level goals	x					
Maximum contaminant levels				x		x
Toxic Substances Act	x	x	x	x	x	x
Resource Conservation and Recovery Act	x		x			
CERCLA (SARA):						
Reportable quantities	x		x			
National contingency		x			x	
Federal Insecticide, Fungicide and Rodenticide Act:						
Data requirements			x			
Minor uses	x	x	x	x	x	x
Atomic Energy Act						
Radioactive waste	x	x	x			
Uranium mill tailings	x	x	x	x	x	x

Source: Adapted from U.S. EPA (1987a), table 3-1, p. 3-2.
Note: *includes non-air-quality health and environmental impacts. **statute refers only to cost. ***includes non-water-quality environmental impacts only.
[a]NAAQS designates the National Ambient Air Quality Standards and relates to the criteria air pollutants.
[b]"Other Effects" refer to nonhealth effects on humans and firms.
[c]"Welfare Effects" refer to visibility and aesthetics, effects on nonhuman species, crops, sodding, materials damage.
[d]The type of analysis here depends on the grounds for control.
[e]There is some question about whether any benefit information may be considered. One school of thought is that national aggregate benefit estimates might be allowed into this process. Such estimates would here reflect especially recreation as a pathway for accrual of benefits to society.

dard.[2] In contrast, under the Clean Water Act, the definition of one type of technology-based standard, best conventional treatment (BCT), can be based on costs (in comparison with the marginal costs of secondary treatment at municipal waste treatment facilities) but not on the specific benefits to be realized at individual water bodies.

Second, the mandates involve significant areas of overlap where different regulatory analyses are intended to influence the same types of pollutants in the ambient environment, for example, primary standards for criteria air pollutants and New Source Performance Standards defined on the basis of the effects new discharge sources would have on the concentrations of these pollutants.

One important implication is that economic analysis (in this case, benefit-cost analysis) usually involves an evaluation of the net effect of standards chosen on some basis other than economic efficiency. Another is that standards set under one provision of one law may well overlap in their effects with those set under another provision or law. This raises difficulties for the definition of benefits—at least whenever marginal benefits are nonlinear—because of the interdependence of baselines.

Other serious problems introduced by the standard setting operation can be considered in a few specific examples. Setting an environmental standard of either the discharge or ambient sort requires that the regulator must (Richmond 1983):

1. identify the pollutant to be regulated;
2. select the form of the standard (i.e., a technology to achieve an emissions rate or an ambient concentration);
3. choose the concentration or discharge amount that will be the average target;
4. pick the averaging time over which the target is to be met (an hour, a day, a week, a year, etc.);
5. define the exceedance rate(s) of interest (e.g., a weekly average standard might be paired with a daily upper limit);
6. define what constitutes a violation, taking account of the statistical error structure displayed by the monitoring equipment and other relevant sources of uncertainty (such as measurements made across a sample of different applications of a technology where the standard is technology based).

Thus, evaluating the net benefits of an environmental standard is a complicated business (Portney 1984; Smith 1984). Not only do we need information on the effects of average (or peak, as applicable) exposures to particular pollutants (or ecological effects of average concentrations), we also should be able to evaluate alternative patterns of allowed exceedance. In practice we are fortunate if we have the data from which to estimate dose-response functions over *any* range and averaging time.

The case of the particulate matter (PM) ambient air quality standards permits us to see some of the troubles that can arise even within this limited context. The benefit-cost analysis done for PM was the most expensive of those discussed in the EPA report cited above (U.S. EPA 1987a; see our n. 1). It seems reasonable to assume that the quality of the analysis reflects these expenditures.

The first and largest problem in analyzing PM benefits was that the available laboratory evidence on the health effects of airborne particulates did not match up with the available ambient measurements. Laboratory toxicology suggested that particles smaller than 10 microns across were responsible for whatever health damage was observed. Since preventing health damage was the mandated basis of the standard, the ambient standard had to be written in terms of these small particles. Ambient measurements, with a few isolated exceptions, had for years been done in terms of total suspended particulates (TSP). As a consequence, epidemiological studies aimed at finding health effects associated with airborne particulars inevitably labored under an imposed errors-in-variables problem.

More fundamentally, however, analyzing the total net benefits of the 10-micron PM standard required that the relation between TSP and the distribution of particles by size, both before and after a standard, be understood. In addition, the analysis does not end with health because other benefits could be identified that depended on other sizes of particulate matter. In fact, the PM study conducted by EPA (and subcontractors) involved separate assessments of the health benefits (including mortality and morbidity effects), the household benefits from reduced soiling and materials damages, and the benefits to the manufacturing sector from reduced soiling and materials damage. The first two relied on judgmental appraisals (see MathTech 1982) of the "best" estimates of dose-response relationships and the last two involved the development of new models linking consumer expenditures (on commodities related to household cleaning) and sectoral cost functions to measures of particulate concentrations.

The importance of the institutional setting in combination with the technical and natural systems also can be seen in the cost estimates for the PM standard. Developing these estimates required a specification of how states would formulate their state implementation plans (SIPs), the degree of compliance with the plans, and the resulting estimated levels of particulate emissions. Emissions then had to be translated into estimates of the ambient concentrations of particulates. Of course, uniform ambient air quality standards do not imply uniform levels of actual air quality, a point we return to in section 10.3. The changes in air quality from a specified baseline defined spatially will depend on how the assumed SIP describes the process (the set of discharge reductions) used to meet the standard in each air quality control region.

To stress the analytically arbitrary nature of the institutional context, we report an example drawn from Liroff (1986). When states decide how to

Table 10.2 **Average Emission Reductions of Volatile Organic Compounds Predicted to be Required to Meet Ozone NAAQS in Selected Ohio Cities**

City	Technique 1[a] (%)	Technique 2[b] (%)	Technique Selected
Cleveland	87	50	1
Akron	35	18	2
Toledo	47	25	2
Columbus	43	25	2
Canton	22	10	2
Youngstown	64	44	2
Dayton	61	40	2
Cincinnati	40	50	1

Source: Adapted from Pacific Environmental Services, Study of the 1979 State Implementation Plan Submittals (Elmhurst, Ill.: Report prepared for U.S. National Commissioner on Air Quality [December 1980], 7–12) and published in Liroff (1986).
[a]Known as "EKMA."
[b]Known as "rollback."

achieve the National Ambient Air Quality Standard (NAAQS) for a pollutant, they may have a choice among different average levels of emission reduction depending on which mathematical model of the local atmospheric system they choose to use to predict ambient concentrations. Table 10.2, based on Liroff's table 2.2, shows the choice facing Ohio in designing its implementation plan for ground-level ozone. The two alternative models lead to alternative patterns of predicted ambient concentrations, though both would show no violation at any monitoring site. Thus, the predicted net benefits of the ozone NAAQS in Ohio (and generally in any state) will depend on the choice of modeling technique, not just on the average level of the standard. Of course, it is possible that either or both models may be wrong. Neither pattern of reductions might in fact result in meeting the NAAQS.

We now shift our focus and turn to natural system information and modeling and the implications of how we handle such matters for our estimates of the benefits of environmental standards.

10.3 Bringing in the Natural World

The evaluation of environmental policies inevitably involves economists with the systems that make up the ambient environment. If a policy mandates a reduction in polluted waste water discharge from industrial and publicly owned sources, the streams, rivers, lakes, and ponds that constitute the receiving waters translate the discharge reduction into ambient quality improvements that are valued by individuals. If we turn this notion around—if public policy involves mandated upper limits for concentrations of pollutants in the ambient atmosphere, the transportation, dilution, and transformation processes at work in that atmosphere must have a key role in determining how much discharge reduction has to be accomplished to meet the standard.

While this seems intuitively clear, the importance of knowledge of those processes is greater than these observations suggest. There are two reasons for this. One is ubiquitous; the other is found to be central to some situations and not to others. The ubiquitous influence is *space,* the differential location of pollution sources and pollution receptors in the two-dimensional plane.[3] Additional complication is introduced by the nonlinearity of most environmental processes.

Consider the role of location. In the simple situation, a policy is represented graphically or mathematically by a single marginal benefit (or damage) and a single marginal cost function. These may have as arguments either ambient pollutant concentrations or pollution discharged. The optimum policy is defined by the usual MB = MC condition. The addition of spatial detail merely replicates this condition at each location. That is, efficient policies equate the marginal benefits to the marginal costs of realizing a given level of ambient quality at each location. In conventional Pareto efficiency terms this corresponds to equality of the relevant sum (for that location and its residents) of the marginal rates of substitution for environmental quality (in relation to a numeraire) to the corresponding shadow price describing the real costs of attaining it. The natural system is implicit in the definition of the real marginal costs. When perfect mixing of all pollution discharges produces uniform concentrations of pollutants everywhere in the ambient environment—as is roughly true for some air pollutants under certain physical and meteorological conditions—the simple model offers a reasonably good approximation.

But in the largest number of cases, it does not. For a mandated policy of emission reductions, even if that policy involves equal percentage reductions at all sources, the amount of ambient quality improvement will, in general, be different at every point in the relevant environmental medium. If the policy to be evaluated involves mandated ambient quality standards, the situation is even more at odds with the simple model. Not only will the concentration in the standard characterize only a few points in the environment after the policy is implemented, but which points those are and by how much the quality at every other point is better than the standard will, in general, depend on exactly how the standard is implemented.

Both environmental quality levels and, more important, improvements in quality attributable to a policy are different at every point in the environment. Moreover, every point is usually characterized by different levels of human "use." Thus, for example, some points in the atmosphere coincide with dense residential population, some with sparse; some coincide with industrial plants, some with office buildings, some with vacant space. Similarly, along a river some segments will have heavy recreational use (or prospectively have such use) because of conditions of access, bank type, current, and temperature. Other segments may be unattractive to recreationists for reasons having nothing to do with the level of pollution at that location.

Therefore, the estimates of benefits of proposed (or actual) environmental management policies are intrinsically dependent on the accuracy of our knowledge of the natural world processes, upon the detail required for the spatial resolution, and on the implementation plan assumed in the analysis.[4] The net benefits of a given policy, $\bar{P}$, can be written in fairly general terms as follows:

$$
\begin{aligned}
(1)\quad NB(\bar{P}) = {} & B_1\{f_{11}[D_1(\bar{P})] + f_{21}[D_2(\bar{P})] + \ldots + f_{n1}[D_n(\bar{P})]\} \\
& + B_2\{f_{12}[D_1(\bar{P})] + f_{22}[D_2(\bar{P})] + \ldots + f_{n2}[D_n(\bar{P})]\} \\
& + B_m\{f_{1m}[D_1(\bar{P})] + f_{2m}[D_2(\bar{P})] + \ldots + f_{nm}[D_n(\bar{P})]\} \\
& - C_1(X_1 - D_1(\bar{P})) - C_2(X_2 - D_2(\bar{P})) - \ldots \\
& - C_n(X_n - D_n(\bar{P})),
\end{aligned}
$$

where there are m points (call them receptor locations) at which we agree to measure ambient quality and infer benefits, and n sources of pollution. The functions f_{ij} $[D^i(\bar{P})]$ represent the environmental transformation of discharge level D_i $(\bar{P})$ into a contribution to ambient quality at point j. Writing D_i as a function of $\bar{P}$, the policy, we can emphasize the point that (in most cases) pollution management policies operate through affecting discharges of pollutants. The $C_i(\cdot)$ functions describe the costs to source i of reducing emissions by $X_i - D_i(\bar{P})$, with X_i the uncontrolled emissions of that source.

In general, the benefit functions $[B_j(\cdot)]$ are different for every j because of the factors mentioned above. Thus, every discharge level is a function of the policy choice, and the ambient quality at every receptor location can, in principle, be a different function of every discharge level. For example, if $\bar{P}$ consists of a required 50% reduction of prepolicy discharge at every source, that defines the vector $\{D_1(\bar{P}), \ldots, D_n(\bar{P})\}$.[5] These discharges are transformed by the functions f_{ij} (D_i) into ambient pollution (or quality levels; and the resulting quality at each receptor location is valued using the functions $B_k(\ )$. If, on the other hand, the policy $\bar{P}$ requires an upper limit on ambient pollution at any receptor location, call it S_k, analytical implementation implies finding a vector of discharges satisfying the requirement. This will depend on the functions f_{ij} (D_i), for we are solving a problem of the following form:

$$\text{find } D_i(\bar{P}) \text{ such that } \forall_j \sum_i f_{ij}[D_i(\bar{P})] \le S_k.$$

This is different than the description in textbooks because the policy is not defined to meet an efficiency criterion. We simply use (1) to evaluate how its implications relate to the net benefits realized with some baseline or status quo position. There may be no such vector. More often, since $n > m$, there will be an infinite number. The benefits flowing from the choice S_k will depend on which vector $D(\bar{P})$ is evaluated. This is because every such vector will, in general, produce a different pattern of ambient quality across receptor locations. Further, in this general formulation, there is no presumption that quality better than the standard is valued at zero.[6]

To illustrate what happens if we ignore the natural system, we offer one very simple and two not-so-simple examples. First, consider a hypothetical region with two sources of air pollution and three receptors or agreed-on monitoring locations. The sources are, in fact, linked to the receptors by an atmospheric system that can be characterized by a matrix of transfer coefficients, T, as follows:

	Receptor		
Source	I	II	III
A	2	1	.5
B	1	2	2

Ambient quality, Q, is determined on the basis of source discharge as:[7]

(2) $$Q = DT, \quad \text{where } D = (D_A, D_B).$$

The benefits of discharge reductions are assumed obtainable, as damages avoided, from a quadratic damage function.

(3) $$G_i(Q) = Q_i^2 \quad \text{for each receptor } i^8.$$

If initial discharges are $D_{AO} = 4$, $D_{BO} = 2$, the base or initial quality levels are:[9]

(4) $$(Q_{io}) = (10, 8, 6),$$

with resulting damages

(5) $$\sum_{i=\text{I}}^{\text{III}} G_i = 200.$$

The effect of what we might call environmental ignorance is illustrated by considering three different methods of evaluating the benefits of setting increasingly stiff ambient quality standards, S_j:

(i) We know nothing about the environment (in particular, we do not know T), so we simply work from the regional average concentration before the standard is set and assume that the standard is the average concentration after it is set. Let us denote this approach to estimating benefits as method (i), designated B^1. Then

(6) $$B_j^1 = 3\left[G\left(\frac{\Sigma Q_{io}}{3}\right) - G(S_j)\right],$$

where j indexes the severity of the standard.

(ii) We still know nothing about the atmospheric system (T) but disaggregate benefits. In this formulation, benefits are calculated only for receptors where the initial quality level is worse than the standard. Moreover, it assumes that at every such point, after the imposition of the standard, quality just equals the standard. This is method (ii) (B^2), given by (7):

$$B_j^2 = \sum_i G(Q_{io}) - G(S_j), \tag{7}$$

for all i such that $Q_{io} > S_j$.

(iii) We know and use T. Implementation policy is a "rollback" rule from base period discharges. That is, with particular standard, S_j, the rollback rule specifies that each discharge is reduced by the proportion R_j, given by:

$$R_j = \frac{\max (Q_{io}) - S_j}{\max\limits_i(Q_{io})}, \tag{8}$$

so that benefits are

$$B_j^3 = \sum_i [G(Q_{io}) - G(Q(R_{ij}))], \tag{9}$$

where

$$Q(R_{ij}) = (1 - R_j)(D_{AO}, D_{BO})[T]. \tag{10}$$

Each of these methods provides a definition of the aggregate benefit function and with it describes our knowledge of the environment. Table 10.3 summarizes the aggregate benefits under the three definitions and four levels of ambient quality standard: 8, 6, 4, and 2. Both total and marginal benefits are

Table 10.3 Aggregate and Marginal Benefits: The Two Source–Three Receptor Example

		Aggregate Total Benefits by Standard				Aggregate Marginal Benefits by Standard			
Method	Definition	8	6	4	2	8	6	4	2
B^1	Average initial regional concept of quality relative to standard	0	84	144	180	0	42	30	18
B^2	Actual initial quality relative to standard	36	92	152	188	18	28	30	18
B^3	Actual initial quality relative to actual quality as determined for rollback implementation	72	128	168	192	36	28	20	12

shown, with the latter defined as the difference between the benefit at standard $S_j + 2$ and at S_j divided by two.

At those standards with small improvements over the baseline quality, the three measures exhibit the least agreement for both total and marginal benefits. Methods (i) and (ii) ignore benefits produced by improvements beyond the standard required by the control actions necessary to meet the standards at the binding receptor.[10] As the standard is tightened they exhibit closer correspondence. This is not surprising because as the standard is tightened toward zero pollution, the variation around the average ambient level is reduced. Thus, the difference between the standard and the quality level at any particular nonbinding receptor is reduced, and with it the sources for the differences between B^1, B^2, and B^3 diminish.

The marginal benefits calculated ignoring the natural system are an especially unreliable guide to optimal policy choice. These results are not simply artifacts of our example. Two more realistic cases illustrate the peril of ignorance of the natural world's systems. The first is based on the data developed for the Baltimore, Maryland, region in the paper by Oates, Portney, and McGartland (1989), using their air quality results (for total suspended particulates) and translating them into versions of our surrogate benefit measures. The primary difference is that method (iii) reflects a least-cost rather than a rollback scheme for implementation. So we refer to it as method (iii′) (see app. A for data and methods). Table 10.4 contains a summary of the results

Table 10.4 **Surrogate Benefits of Reductions in Total Suspended Particulates for Baltimore by Level of Ignorance and Standard (millions per year)**

	Level of Standard (ug/m³)						
	115	110	105	100	95	90	85
Method (i):							
Total	0	0	0	0	0	0	0
·thod (i) (modified):[a]							
Total	12.3	23.6	34.5	45.2	55.1	64.5	73.6
Marginal	12.3	11.3	10.9	10.7	9.9	9.4	9.1
Method (ii):							
Total	2.6	6.0	9.7	15.4	21.2	28.2	35.2
Marginal	2.6	3.4	3.7	5.7	5.8	7.0	7.0
Method (iii′):							
Total	7.7	19.7	27.7	34.9	46.2	59.1	73.7
Marginal	7.7	12.0	8.0	7.2	11.3	12.9	14.6
Marginal benefits[b]	7.2	12.9	9.1	8.5	13.2	15.1	16.4

Source: See the appendix for a description of the data and method.

[a]The modification consists of comparing initial average concentration to projected average concentrations for each standard, where the projection depended on the percentage change in the standard.

[b]Taken from Oates, Portney, and McGartland (1989); amounts given are in millions of 1980 dollars.

for total and marginal (surrogate) benefits for each estimation approach or level of knowledge. The marginal benefits calculated by Oates et al. (1989) are shown at the bottom of the table.

Thus, in a much more realistic example, the methods that ignore the natural environment produce problematic estimates of marginal and total benefits. Method (i) shows no benefits because the base case average TSP concentration is already below all the standards considered. Method (ii) produces substantial underestimation of both marginal and total benefits. It ignores improvements at receptors that have quality better than the standard before it is imposed.

Modified method (i) depends on simple reduction of the average TSP concentration for the region for each standard level by the same percentage as that standard represents a reduction of the baseline standard that McGartland et al. (1988) use in their benefit calculations: 120 micrograms per cubic meter (120 ug/m^3). It produces total benefit numbers roughly similar to those obtained in method (iii), the method reflecting best available knowledge of the environment. But this apparent improvement does not extend to marginal benefits. The actual pattern obtained via method (iii) shows an early peak at 100 ug/m^3, followed by a dip and, then, subsequent increases. Indeed, marginal benefits are still increasing at the strictest standard shown.[11]

Of course, one might criticize this example as well, noting that we are not working with a "real" damage function. Our last example does just that, using data on water quality changes, as measured by dissolved oxygen, generated by a complex and quite realistic model of the Delaware River estuary; a mapping of dissolved oxygen (DO) into sustainable recreation types from a second source; and an annual per capita willingness to pay for the availability of water-based recreation by type from a third source. (The details of the data and calculations are set out in the appendix.) The results for total and marginal benefit estimates are given in table 10.5.

Table 10.5 **Surrogate Benefits of Improvements in Water Quality in the Delaware Estuary by Level of Ignorance of Standard**

	Water Quality Standard (ppm of Dissolved Oxygen)	
	3.5	5.0
Method (i):		
Total	0	420.2
Marginal	0	420.2
Method (ii):		
Total	184.5	510.6
Marginal	184.5	326.1
Method (iii):		
Total	372.7	581.4
Marginal	372.7	208.7

The patterns of marginal benefits once again display the largest effects from ignorance of the natural world. Method (i) implies there would be no benefits of going from the baseline situation to a standard of at least 3.5 parts per million (ppm) of DO for every reach of the river. But the marginal benefit of tightening the standard from 3.5 ppm to 5.0 ppm of DO is 420.2. Under method (ii)—reach-by-reach disaggregation, but assuming benefits only for reaches that are initially worse than the standard—the marginal benefit of the 3.5 ppm standard is 184.5 and that of the 5.0 ppm standard is 326.1. This pattern is almost exactly the reverse of that observed when complete knowledge is used in method (iii). In this case, the marginal benefits associated with the lower standard are 372.7, while those associated with the next improvement to 5.0 ppm are 208.7. Thus, even though the total benefits estimated to be associated with the tougher standard are roughly similar for methods (ii) and (iii), the marginal benefit patterns are very different.

The results of these examples may be so obvious that their applicability seems doubtful. Who would ever use methods such as (i) or (ii)? The answer—and this is the key to our later recommendations—is just about everyone. An examination of the invaluable compilation of benefit estimates published by Freeman (1982) reveals that every one of the reported air pollution benefit studies uses a version of B^1 or B^2, with most relying on a method very like method (i). The water pollution benefit studies he summarizes all use a version of B^2 in which full attainment of the most ambitious standards (or ambient quality goals) of the Clean Water Act (CWA) is assumed.

As important as pointing out the prevalence of benefit estimates based on ignorance of natural systems is, an attempt to understand why this is the case also merits consideration. In the case of water pollution control benefits, the answer is generally that insufficient resources have been invested in the research needed to reduce our ignorance. Translating the technology-based discharge standard definitions of the CWA into actual discharges from tens of thousands of point sources of water pollution is hard enough. But then translating such changes in discharges, were they available, into changes in water quality indicators that in turn can be valued by individuals, involves data gathering, modeling, and basic conceptual research efforts beyond what the sponsors of such research have been willing to pay.[12] Finally, the data on valuation that is available generally is in the form of step functions unsuited to the valuation of benefits of small improvements in quality, especially at reasonably clean receptor locations.

For air pollution benefits, the state of the art of emission inventories and air quality modeling has for some time been capable of supporting the sort of disaggregated, location-specific benefit estimates obtained by Oates et al. (1989) for Baltimore. When national total benefit estimates have been the object of the exercise, however, it apparently has been too daunting a task to manage the necessarily massive data banks and atmospheric models.

Finally, before we turn to the next concern of this paper, the valuation of

Table 10.6 **Marginal Benefits of Reductions in Total Suspended Particulates for Baltimore by Implementation Method and Standard (millions of 1980 dollars)**

	Level of Standard (ug/m³)						
Implementation Method	115	110	105	100	95	90	85
Command and control	2.2	10.5	9.7	11.5	7.5	10.0	6.5
Least cost	7.2	12.9	9.0	8.5	13.2	15.1	16.4

Source: Oates, Portney, and McGartland (1989), table 1.

environmental quality changes, we should consider the effects of implementation plans on benefits. This is the primary focus of Oates et al. (1989). While their paper actually is addressed to the relevance of benefit estimates for the choice between regulatory approaches ("command and control" vs. use of economic incentives), their results provide a fine illustration of the point that for any given level of environmental knowledge, estimates of benefits will depend on the method of implementation—the pattern of discharges—assumed.

Thus, in table 10.6 we reproduce their marginal benefit estimates for the command and control and "least cost" implementation approaches. In this case, neither set of estimates can be characterized as "wrong." Both reflect the best environmental information available. Nonetheless, they are very different. Thus, the statement that a particular standard yields particular benefits has meaning only when an implementation method is explicitly assumed.

The same standard, treated as an upper bound on a pollutant's allowable concentrations, can imply an infinite number of aggregate marginal benefit patterns because these benefits will depend on how the standard is implemented and on what the natural system implies this implementation plan will yield as the ambient concentrations for each receptor location. In most theoretical treatments of these issues, this problem is avoided by simplifying assumption. The benefits are taken to be measured at a single, representative point in the environment. The costs of improving quality at that point are assumed to reflect the environmental transformations implicitly.

10.4 Evaluation Benefits: Learning from Past Research and Identifying New Initiatives[13]

The statutory guidelines creating the demand for valuation measures for environmental resources and the time horizons written into the statutes make it impossible to develop new benefit-cost studies for each decision. This has led to growing interest in the methods used to transfer valuation (or demand) estimates derived in one situation to a new one. Both the Oates et al. (1989) study of air quality in Baltimore and our own analysis of water quality in the

Delaware River used valuation estimates derived from one or more studies in the literature (see the Appendix). For the most part, these were derived from judgmental reviews of the literature and propose a best estimate (or a range of values).

Because the services of environmental resources exchange outside markets, the methods used to estimate consumers' values for them have developed along two lines. The first focuses on observable behavior that can be linked by assumptions to the resource of interest. Methods relying on this strategy have usually been labeled the indirect approaches. They include: the travel cost recreation-demand models, hedonic price functions (property value and wage rate), hedonic travel cost functions, damage-averting cost models, and factor productivity (or reverse value-added) methods. In each case, an individual's (or a firm's) actions are assumed to be partially motivated by a desire to obtain the service of an environmental resource (or to avoid the detrimental effects of pollution to that resource). Using models based on these actions, researchers attempt to estimate the marginal value of changes in the quantity or the quality of the nonmarketed resource.

The second group of methods relies on survey techniques that ask respondents how they would value (contingent valuation) or change their behavior (contingent behavior) in response to a postulated, hypothetical change in the services of an environmental resource.[14] This method assumes that an individual's response to a hypothetical situation provides an authentic description of how he (or she) would respond to an actual change.

The purpose of this section is to suggest that efforts to summarize and evaluate benefit estimates offer another kind of opportunity—to evaluate what we have learned about the values of environmental resources; to examine the sensitivity of these estimates to the modeling decisions required to develop them; and, based on these two appraisals, to identify new data and analyses required to resolve the uncertainties leading to the disparities in valuation estimates. The required analyses treat the results from past studies as data to "test" whether differences in the estimates (across studies) reflect systematic variations in the resources being valued or in the assumptions and the methods underlying them.

While this approach appears to be a new one for evaluating empirical research in economics, it is not new to other social and health sciences.[15] "Meta-analysis" describes a research method that seeks to provide systematic summaries of the findings from empirical evaluations of educational or social programs. Du Mouchel and Harris (1983), for example, proposed a similar strategy for the transfer of risk assessment models from animal to human populations.

Our objective is broadly similar. However, we seek to evaluate whether there are systematic influences on the values estimated for specific types of environmental resources and whether these influences can be distinguished from the assumptions and features of the methods. Ideally, such an analysis would be undertaken within a single empirical study in which consistency in data sources, reporting conventions, and statistical modeling criteria could be maintained across the resources and models studied. Unfortunately, this was not possible. Consequently, we summarize the results of a pilot study conducted by Smith and Kaoru (1990) that uses the existing literature as the basis for an examination of the determinants of valuation estimates for recreation resources. The focus on value estimates is deliberate because, regardless of the original objective of the research, benefit estimates have been the single most important policy use of the outputs this type of research.

Equation (1) defines the basic model. To use it, we maintain that the valuation estimate relevant for our example, the real consumer surplus, RCS, per unit of use of a site is a function of four types of variables: the type of recreation site, X_S; the assumptions inherent in the model specification, X_A; the form of the demand model, X_D; and the estimator used, X_E.

$$\text{RCS}_i = \alpha_0 + \alpha_S X_{Si} + \alpha_A X_{Ai} + \alpha_D X_{Di} + \alpha_E X_{Ei} + e_i, \tag{11}$$

where X_{ji} and α_j, $j = S, A, D, E$ are conformably dimensioned vectors and e_i is the stochastic error for the ith estimate.

Smith and Kaoru (1987, 1988) have reviewed over 200 published and unpublished travel cost demand models prepared over the period 1970–86 and developed a data set summarizing the valuation estimates, features of the resources involved in these demand studies, and characteristics of the models involved. The results reported here are based on 77 studies. They yield 734 observations for the consumer surplus per unit of use. The individual observations vary by recreation sites, demand specification, modeling assumptions, and estimator used.

There was enormous variability in the information reported across studies. Often the objective of the research was something other than estimating the values for a recreational facility. It may have been testing a specific hypothesis, with the results reported confined to the specifics of the hypothesis test. Smith and Kaoru did not attempt to contact individual authors to supplement (or check) what was reported in the individual papers. Rather their data set relies exclusively on the information reported within these limitations. Table 10.7 defines some of the variables that could be consistently defined across the studies in each class of variable.

To interpret the results obtained from statistical analyses of valuation estimates across different studies, we must formulate specific hypotheses concerning how and in what dimensions these estimates might be sensitive to modeling judgments. A beginning step in this process can be found in past literature reviews (i.e., Ward and Loomis 1986; Smith and Desvousges 1986; Bockstael et al. 1987), as well as in what seem to be established conventions in developing travel cost demand models. A few such protocols would include:

Table 10.7 **Description of Variables for Analysis**

Name	Definition
RCS (real consumer surplus)	Marshallian consumer surplus estimated per unit of use, as measured by each study (i.e., per day or per trip) deflated by consumer price index (base = 1967)
Surtype	Qualitative variable for measure of site use = 1 for per trip measure, 0 for per day measure
Recreation site variable	Lake, river, coastal area of wetlands, forest or mountain area, developed or state park, national park with or without wilderness significance are the designations; variables are unity if satisfying designation, 0 otherwise
Substitute price	Qualitative variable = 1 if substitute price term was included in the demand specification, 0 otherwise
Opportunity cost type no. 1	Qualitative variable for measure used to estimate opportunity cost of travel time = 1 if an average wage rate was used
Opportunity cost type no. 2	Qualitative variable for the second type of opportunity costs of travel time measure = 1 if income per hour used (omitted category was predicted individual specific wage)
Fraction of wage	Fraction of wage rate used to estimate opportunity cost of travel time
Specific site	Qualitative variable for use of a state or regional travel cost model describing demand for a set of sites = 1, 0 otherwise
Demand specifications	Linear, log-linear and semilog (dep) are qualitative variables describing the specification of functional form for demand (semi-log in logs of independent variables was the omitted cateogory).
Estimators used	OLS, GLS, and ML-TRUNC are qualitative variables for estimators used, omitted categories correspond to estimators with limited representation in studies including the simultaneous equation estimators.

1. Use trips as the quantity measure where possible and attempt to segment the sample when it is known that the length of stay per trip is different.
2. Take account through sample segmentation of differences that might arise from use during different seasons or during different time periods when there may be different time or resource constraints.
3. Treat travel time as an element affecting the cost of a trip.
4. Include vehicle-related costs and the costs attributed to travel time as well as any entrance fees or site usage costs (i.e., parking costs, lift fees for skiing, etc.) in the unit cost estimated for a trip.
5. Use substitute prices to measure effects of substitute sites rather than an index of substitution; complete systems of demand functions are unnecessary if the objective is to measure demand for one of the sites.

6. Reflect quality features of the site in the demand models.
7. Recognize that heteroscedasticity is likely to be an issue with zone data and that selection effects can be important with individual data.
8. Avoid the problems posed by cost allocation issues that can arise with multiple destination trips by segmenting the sample according to the distance traveled to the site.
9. Substitute sensitivity analysis for strict adherence to one particular functional form of the demand function.

Equally important, areas exist for which there are either insufficient data or the absence of a clear consensus. These are:

1. the measurement of the opportunity cost of travel time; simple scaling of the wage rate was not found to be consistent with several of the demand studies based on individual data, yet explicit recognition of multiple prices for recreation time is generally beyond the information set available in most current studies; to date no compromise has been proposed to deal with this problem;
2. the treatment of the attributes of a site's services; and
3. the definition of a recreation site for modeling demand, especially where there are many comparable sites within a small geographic area or where there is one large "site" that extends over a wide area.

What has been missing in past assessments is some gauge of how important the decisions might be in influencing the valuation estimates that result.

From the perspective of being able to transfer valuation estimates, we would prefer that the empirical estimates of equation (11) be consistent with a maintained hypothesis that $\alpha_A = \alpha_D = \alpha_E = 0$. That is, judgmental modeling assumptions contribute to the variability in benefit estimates but do *not* impose systematic influences on the size of the benefits estimated. Of course, to the extent this is not our conclusion, then we believe the process has identified areas where further research, modeling, and data collection may be warranted.

Table 10.8 provides some descriptive statistics from the Smith-Kaoru data on the features of the studies, classified by the type of site involved. It reports the number of estimates for each type of resource, the mean and range in real consumer surplus (per unit of use) estimates, the proportion of the studies based on individual (as compared with origin zone) data, and the range of years represented in the studies. It is clear that there are exceptionally wide variations in the consumer surplus per unit of use—from under $1 to over $100 in five of the seven cases. Two of these have estimates over $200. These differences could represent dramatic differences in the character of the resources in each group, in the models used, or in the characteristics of the recreationists in each sample.

Table 10.9 reports the ordinary least squares (OLS) estimates for five models which consider whether the variations in real consumer surplus across

Table 10.8 A Comparison of Travel Cost Demand Results by Type of Resource

Type of Resource	No. of Estimates	Real Consumer Surplus[a] Mean	Range	PI[b]	Years[c]
River	257	$17.05	$.29–$120.70	.61	1966–83
Lake	483	16.85	.09–219.80	.55	1968–83
Forests	114	31.36	.80–129.90	.59	1968–84
National parks	12	44.01	23.48–120.70	.50	1980–83
Wetlands	9	45.86	17.45–120.70	.78	1980–83
State parks	107	42.49	.67–327.20	.07	1972–83
Coastal areas	28	35.49	.67–160.80	.61	1972–84

Source: Smith and Kaoru (1990).
[a]Real consumer surplus deflates the nominal estimates by the consumer price index (base 1967).
[b]This variable designates the proportion of the studies based on samples of individual recreationists' trip-taking decisions compared with origin zone aggregate rates of use.
[c]The range of years in which the data used in these studies were collected. Thus, this variable designates the range of years across the studies in each category in which behavior was observed.

Table 10.9 The Determinants of Real Consumer Surplus per Unit of Use

Independent Variables	Models 1	2	3	4	5
Intercept	23.72	16.07	20.30	27.03	18.75
	(5.62)	(2.08)	(6.19)	(3.68)	(0.58)
			[3.92]	[3.64]	[1.04]
Surtype	7.99	−4.13	−9.97	15.38	19.88
	(2.76)	(−1.45)	(−2.72)	(2.97)	(3.74)
			[−1.36]	[2.34]	[3.55]
Type of site (X_S):					
Lake	−11.70			−18.69	−20.32
	(−3.18)			(−3.24)	(−3.52)
				[−2.36]	[−2.48]
River	−5.57			−14.29	−19.03
	(−1.93)			(−2.99)	(−2.19)
				[−1.95]	[−1.75]
Forest	−.45			−18.45	−25.99
	(−.93)			(−2.36)	(−3.01)
				[−1.93]	[−2.49]
State park	19.93			24.95	22.37
	(4.44)			(3.47)	(3.44)
				[3.27]	[3.19]
National park	2.54			.56	−3.77
	(0.20)			(.04)	(−.23)
				[.08]	[−.13]
Model assumption (X_A):					
Substitute price			−18.73		−13.71
			(−3.27)		(−2.12)
			[−4.58]		[−1.80]

Table 10.9 ***(Continued)***

Independent Variables	Models 1	2	3	4	5
Opportunity cost of type no. 1			−14.97		−16.49
			(−2.10)		(−2.11)
			[−2.09]		[−2.48]
Opportunity cost of type no. 2			3.95		−15.86
			(1.02)		(−3.30)
			[.45]		[−2.87]
Fraction of wage			37.24		48.59
			(8.56)		(9.76)
			[3.83]		[6.94]
Specific site/regional TC model			22.23		24.21
			(4.10)		(3.85)
			[3.35]		[2.77]
Model specification (X_D)					
Linear		2.35			−2.87
		(.31)			(−.27)
					[−.31]
Log-linear		14.63			23.37
		(1.89)			(2.37)
					[2.88]
Semilog (dep)		11.26			16.89
		(1.52)			(1.86)
					[2.97]
Estimator (X_E):					
OLS					−14.45
					(−.48)
					[−.84]
GLS					−8.58
					(−.28)
					[−.54]
ML-TRUNC					−67.38
					(−2.15)
					[−3.43]
R^2	.11	.03	.25	.15	.42
n	722	722	399	399	399

Source: Smith and Kaoru (1990).
Note: The numbers in parentheses below the estimated parameters are the ratios of the coefficients to their estimated standard errors. The numbers in brackets are the Newey-West (1987) variant of the White (1980) consistent covariance estimates for the standard errors in calculating these ratios. Small sample properties of the White estimate are discussed by Chesher and Jewitt (1987) and MacKinnon and White (1985). While these studies raise questions with the approach for dealing with heteroscedasticity, it has not been evaluated in this more general case.

studies can be "explained" by the classes of variables hypothesized in equation (11). Models 1 and 2 in the table contain the least variables, with 1 considering only qualitative variables describing the types of recreation site and 2 variables describing the model specification. The remaining three models introduce groups of variables to illustrate the sensitivity of the estimates to the model specification, as well as to the reductions in sample size implied by these more detailed formulations. These reductions arise from the incomplete information available in the papers used to construct the Smith-Kaoru data base. Model 5 is their preferred specification.

The numbers in parentheses below the estimated coefficients are the t-ratios calculated with the OLS standard errors. Those in brackets below models 3–5 are the t-ratios using the standard errors estimated from the Newey-West (1987) proposed adaptation of the White (1980) consistent covariance matrix. They are reported to gauge whether the panel nature of these data might have influenced any judgments on the importance of variables describing the sites or the modeling decisions.

The Smith-Kaoru data set is a panel because there are a number of cases of multiple consumer surplus estimates reported from a single study. These can reflect different models estimated with data for a common recreation site, different sites and associated data, or both. Given this diversity in the source of multiple observations per study, the model does not readily conform to either a simple fixed or a random effects model. Newey and West's (1987) covariance estimator allows for a generalized form of autocorrelation and heteroscedasticity. As such, it provides a convenient gauge of the potential effects of the stochastic assumptions maintained in estimating the determinants of the real consumer surplus.

Several conclusions emerge from this statistical summary of the literature. The results clearly support the basic approach to reviewing empirical literature. The models' estimates indicate that the type of resource, the modeling assumptions, specification of the demand function, and estimator can influence the resulting real consumer surplus estimates.

For the most part, individual variables had effects consistent with a priori expectations. Nonetheless, there is at least one important aspect of the variable definitions that should be recognized. Our site classification variable is not a class of mutually exclusive categories. Some sites fall in multiple categories. For example, a state park with a lake would imply unitary values for both of these variables. The estimated coefficients must also be interpreted relative to an omitted category (coastal sites and wetlands), because all sites fell within at least one of these definitions. Thus the differential a state park with a lake would imply in per unit consumer surplus over coastal areas is about \$2.00. Nearly all the variables describing modeling decisions were found to be statistically significant factors in describing the variation in real consumer surplus.

Examples of these results, that are on the one hand consistent with intuition yet also disturbing from the perspective of developing benefit estimates that are readily transferred, include the effects of the treatment of substitute price measures; the value of the opportunity costs of time; the specifications used to capture the effects of multiple sites (e.g., the regional travel cost model); the demand specification (notably the double-log form); and estimator used to account for the truncation effects present with site-intercept surveys.

Overall these findings emphasize the sources of ambiguity in demand modeling described earlier. While the Smith-Kaoru findings represent a beginning and should be interpreted cautiously, some specific areas can be targeted despite this qualification. More careful consideration is warranted of why the treatment of time costs and the selection of an estimator are so important to these valuation estimates. In the first case, the sensitivity reflects the fact that we do not know how the constraints to an individual's time affect his recreation decisions or how an individual's implicit values on time vary with the nature of his choices. Data can be sought on both issues.

Similarly, the importance of the choice of estimator probably reflects the difficult subsidiary issues involved in deciding how to deal with the sampling (Shaw 1988) and selection (Smith 1988) effects associated with intercept surveys. An effort to improve the situation through data collection would involve returning to the early population surveys (i.e., samples designed to be representative of all households, not just users) that elicited information on households' recreation choices. These surveys originally were sponsored by the Bureau of Outdoor Recreation (see Cicchetti, Seneca, and Davidson 1969). However, any new surveys would require information on the sites individuals use and their patterns of use to overcome the problems that arise in the on-site surveys. (The early BOR surveys did *not* collect this type of information.) Understanding the "market" for a recreation site lies at the heart of evaluating why substitute prices and the qualitative variable for regional travel cost models were important.

We know very little of how individuals learn about and subsequently define (for choice purposes) the recreation opportunities available to them. Decisions on the use of "local" recreation sites versus more distant "national" sites will most certainly be made with different time horizons and constraints. How are these decisions to be distinguished and can they be modeled separately? Progress in modeling recreation decisions requires answers.

The empirical models also identify an important role for the functional form selected to describe demand. The recreational demand literature has seen increasing criticism of the use of arbitrary specifications selected largely for convenience or based on some fitting criteria. Several recent studies have argued that behavioral derivations of demand models would be preferable. That is, they suggest models should begin with specific utility functions and derive estimating equations by assuming optimizing behavior and by specifying the budget and time constraints assumed to face individuals. Of course, analytical tractability constrains how these efforts can proceed.

We believe that there is not an obvious answer to the question of imposing prior theory versus using approximations. In a genuine sense, all applications are approximations. What is important is whether the way they are undertaken affects the findings in important ways. The Smith-Kaoru results indicate that greater efforts are needed in developing more robust specifications. Both enhanced data and theory will be required to meet this need.

10.5 Recommendations for Data and Analytical Development

When compared with the effort and experience devoted to the conventional topics considered under the auspices of the Conference on Income and Wealth, the record of empirical analyses of public policies for the management of environmental resources is quite limited. While there has been rapid progress in the last two decades, our ability to deliver estimates of individuals' values for a wide array of environmental resources and, a fortiori, for changes in specific aspects of resource quality lags significantly behind the expectations of current environmental statutes and the projected needs for coming to grips with emerging policy issues. We have tried to describe the sources of these demands and the clear interaction between the needs for economic *and* noneconomic information.

In what follows we propose to use three themes to organize our proposals for new data developments in support of empirical research in environmental economics: learning about natural systems, learning what we know, and responding to emerging policies. As we noted at the outset, our objective is to consider first the generic problems extending over multiple problems that require data and, second, broad classes of environmental problems that seem likely to be important policy issues in the near future. The policy orientation is deliberate. Resources for addressing data and modeling needs are scarce, and we need to consider their net returns here just as in other allocation decisions.[16]

10.5.1 Learning about Natural Systems

As we have stressed at several points, analysis of the benefits (or damages) of proposed or actual changes in the use of natural resources inevitably depends on our abilities to trace the effect of the changed use through to a change in the valuation by consumers of a resource service. This implies that we must be able to (*a*) characterize the current state of the relevant system(s); (*b*) identify a mechanism by which the change in use affects the system; (*c*) model how the change has affected (for ex post damage assessment) or will affect (for ex ante regulatory analysis) the ambient quality of the system in terms relevant to consumer valuation.

In many cases, our knowledge is deficient in every one of these categories. For example, we have a great deal of data on water quality but are generally short of information that systematically covers all the water bodies that our activities affect and that our regulations are designed to protect or enhance. Further, the available information usually covers items relevant to scientists' search for understanding of aquatic biological or chemical processes rather than those that can be related to consumer valuation (David 1971). Even so, to a large extent our abilities to model aquatic processes are inadequate. The models often do not accept as inputs discharges or give as outputs indicators of use or of resulting ambient quality relevant to policy evaluation needs.

The great need here is for data-gathering and model-building efforts to reflect the demands of policy analysis. Identifying the need is a great deal easier than meeting it, for the required interaction has all the difficulties of interdisciplinary research plus those of interstate and interagency jurisdictional disputes. Leadership from U.S. EPA and the Council of Environmental Quality is clearly needed.

10.5.2 Learning What We Know

Over a decade ago, in closing his overview of the state of the art in benefit estimation, Freeman (1979) observed that economists could advise the EPA administrator how to measure benefits from a particular pollution control policy. All that was needed were the data and learning that accompany implementation. The intervening decade has seen some positive investments in both data collection and in empirical modeling. However, we cannot be overly sanguine about what has been accomplished. For the most part, the efforts have been very specialized—relying on existing data on consumer behavior or developing special purpose contingent valuation surveys to estimate how individuals would value (or respond to) changes in very specific resources. This process has made it clear that under currently shrinking budgets (or even with modestly expanding resources), we cannot possibly estimate the values for all the resources of current interest.

The notion of evaluating the conditions for transferring estimates from one resource to another is a relative new one. It has been an important part of the practice of developing the information benefit-cost evaluations involving nonmarketed resources. Freeman (1984) distinguished top-down and bottom-up transfers, where the former attempts to allocate an aggregate benefit for a change in all of one type of resource (e.g., the share of the national benefits from a water quality improvement attributed to one site), and the latter refers to using microestimates for the household and a specific resource in other contexts and aggregating. Naughton, Parsons, and Desvousges (1991) recently considered the generic issues in performing benefit transfers at the microlevel using the pulp and paper industry. Their results suggest that a transfer-based strategy for policy analyses is desirable but may require restructuring the design of future benefit estimation studies for environmental resources.

Another possibility proposed by Mitchell and Carson (1986) involves using survey methods to obtain estimates for national improvements in an environmental resource from individual households. These estimates would then be attributed to individual areas based on the amount of the resources present in the area. The example these authors used involved water quality improvements, and comparison of their approach with the results from a separate contingent valuation indicated a fairly close correspondence between the estimates derived from a specific survey and those from their national survey adjusted with their proposed proportioning method. At this stage, however, the literature is very preliminary. There has been no attempt to develop how the tasks involved in deriving transferable models are related to the factors (i.e., household and resource characteristics) affecting the variation in benefit estimates across resources and user groups.

First, we must learn what we know from experiences to date, and then we must proceed to identify what we need to learn. There is a long tradition in resource economics involving attempts to develop consensus practices in benefit-cost analysis and even specifying benchmark valuation estimates for resource services most closely aligned with water resource projects. These attempts were traditionally associated with the Water Resources Council. Our suggestion here is that we should extend these efforts to the valuation estimates for all environmental resources and thereby move beyond a judgment-based, single value for each type of resource service.

By treating the existing set of estimates for changes in the quantities or qualities of environmental resources as data, it is possible to develop a systematic appraisal of whether the state of the art has advanced to the point where we can associate variations in estimates with differences in the procedures used or with features of the resources (or consumers) involved. This process should identify the areas with greatest uncertainty.

The experience with the Smith-Kaoru pilot study of travel cost demand studies suggests that a more systematic approach, contacting authors to fill in missing details, is essential if a reasonably adequate data base is to be developed in areas in which there has been less research activity. Such efforts would also promote the development of statistical methods for dealing with the unique features of "panel" data sets composed from existing empirical studies.

10.5.3 Emerging Policy Needs

We have classified our views of the emerging policy needs into four categories and now consider each in turn.

1. Environmental Risk

This is one of the most difficult areas for current uses of economic analysis, especially because it appears that individuals' responses to a wide range of environmental risks do not conform to our conventional characterization of rational behavior. A recent EPA publication (see U.S. EPA 1987a) has highlighted just how dramatically inconsistent are public concerns and the rankings of environmental risks based on expert opinion (U.S. EPA 1987b). A comprehensive program of data acquisition and research is needed to determine how and why households value reductions in these types of risks more highly than other sources that often have greater likelihoods of serious effects.

This type of analysis will be important to the design of information programs associated with pollutants EPA does not currently regulate, such as radon, and to the development of labeling standards for products for which they do have responsibility. It is also likely to play a central role in defining "clean" for Superfund sites, in establishing priorities for policy initiatives involving monitoring the underground storage tanks, and in devising new policies associated with more stringent drinking water standards.

2. Air Quality

Acid deposition is hardly "emerging" as an issue; rather the reverse. But that is not because the scientific questions have been answered and the problems have been solved. Indeed, there is still debate in the scientific literature over the relative contribution of different compounds and source locations to observed low pH precipitation, fog, and dry acidic deposition. Under these circumstances, benefit estimation linked to a discharge-reduction policy cannot proceed to meaningful results. So a clear need is for further research into long-run atmospheric transport and chemical transformation processes, with the ultimate aim of allowing predictions of the form: If we reduce sulfur dioxide (SO_2) discharges in this region by this much, average pH of precipitation in this other region will increase by this much.

Even then we shall still be several steps from successful benefit estimation for a policy of SO_2 reduction. It must be possible to extend predictive natural system models to such issues as the link between average annual (or season-specific) precipitation, pH, and soil quality to vegetation health and growth, and to aquatic ecological system functions. For example, if we reduce SO_2 discharge in the Middle West, will New England and New York lakes and ponds have better fish populations (more and larger fish of more highly valued species)?

Only with those tools in hand will it be possible for economists to produce meaningful benefit estimates for the sorts of policies that are regularly debated in the Congress. To prepare for that day, the problems of benefit (or damage) function transfer must be addressed in this problem setting. In particular, it is necessary to consider how best to use the results of national studies on the one hand (e.g., Vaughan and Russell 1982) and local studies on the other (e.g., Smith and Desvousges 1986) to value *regional* effects.

A second air quality issue with even larger potential economic implications is ground-level ozone and particularly the value of trying to attain the currently mandated National Ambient Air Quality Standards for that secondary pollutant.[17] Here it is necessary to improve our knowledge of (a) the sources and actual levels of the precursor pollutants (especially volatile organic compounds [VOCS]), of ground-level ozone in urban and rural areas; (b) the morbidity effects of different levels of ozone; (c) the effects of ozone on vegetation and a variety of materials such as paints, plastics, and synthetic rubbers. Our estimates of the damages attributable to days of sickness of various types and severities must be refined. Moreover, theoretically consistent but practically implementable ways of measuring the value of damage to materials providing services to households, businesses, and governments must be developed.

3. Water Quality

One of the key policy initiatives in water quality will be associated with the national estuarine program. For point sources of waterborne pollution, the first round of effluent guidelines will be in place with over 30 regulations promulgated. All should be in place by the early 1990s. The future here is best characterized as one requiring extensions in the ability of economic valuation to realize greater degrees of resolution in valuing small changes in pollutants.

Present methods and data would not permit such evaluations. Clearly an improved understanding of the linkage between the technical dimensions of water quality and individuals' perceptions of and corresponding valuations for that quality will be necessary (David 1971).

Nonpoint sources, especially agricultural runoff of pesticides and fertilizers to surface waters, represent the largest unregulated source of water pollutants. Presently, EPA does not have authority to regulate these sources. However, recent opportunities to coordinate the selection of areas for the Department of Agriculture's Conservation Reserve Program, based on the effects of pollutants on water resources, expose a new area for economic valuation. Can we set priorities for the selection of lands for inclusion in this system based on their contributions to nonpoint source pollution? To answer this question we need both economic and noneconomic data. Agriculture has been wiling to pay premia over normal reserve payments for withholding lands that might otherwise contribute to impairing significant environmental resources.

4. Stock Pollutants and Global Climate Change

This last area is fundamentally different than the first three emerging issues we discussed in that the policy time horizon is long-term and extends over several decades. While not a new issue (Revelle 1985 suggested that it was identified over 100 years ago), it has achieved a more prominent role on the policy agenda with the Global Climate Protection Act of 1987. This legislation assigns to EPA the responsibility of summarizing the scientific understanding of the greenhouse effect (i.e., the role of the accumulation of carbon dioxide, chlorofluorocarbons, methane, and other trace gases in the upper atmosphere in increasing the average surface temperatures on earth) and in enumerating the policies available for stabilizing these concentrations.

As in our other examples, a key need in this area is for greater understanding of the natural system. In this case it is the link between these atmospheric gases and the extent and timing of any global warming, as well as of the implications of that global warming for regional weather patterns. This issue raises some distinct methodological needs because of the extent of scientific uncertainty over these questions, the time horizon for the potential climatic changes, and the irreversibility of the process.

The requirements for economic information depend, in part, on the progress made in improving our understanding of the natural system. As this proceeds, there is a clear need to understand the processes by which economic activities adapt and the institutions that facilitate such adjustment. Historical and cross-cultural analyses may well offer the only means for developing such insights. Equally important, there is a fundamental need to describe the inherent uncertainties in a way that is genuinely informative for policy. While not unique to this problem, this issue of communicating the inherent uncertainties remains one of the most significant problems facing economists involved in environmental policy.

Finally, in evaluating these data and modeling needs as compared with other data priorities, it is important to recognize that in contrast to positive uses of economic analysis where a lack of data may prevent decisions from being made, this is *not* the case in normative applications. *Decisions are made regardless of whether the economic information is available.* In some cases they are very bad ones. Consequently, here new data developments represent opportunities to improve the quality of decisions and the resource allocations affected by them.

Appendix

Calculating Surrogate Benefits Based on the Baltimore and Delaware River Environmental Quality Projections

Air Quality Surrogate Benefits

Oates, Portney, and McGartland (1989) reproduce their atmospheric model's projected patterns of total suspended particulate concentrations for 23 receptor locations in Baltimore for two alternative implementation approaches. We used and reproduce their table 2 here as table 10A.1. (We ignore their results for 83 micrograms/m^3, ug/m^3.) We follow them in taking the pattern associated with the 120 ug/m^3 standard as our base situation.

While Oates et al. (1989) describe the basis for their damage, and hence benefit estimates, they did not provide the functions they used. However, it turns out that a surrogate function that reproduces the pattern of their margi-

Table 10A.1 TSP Concentrations by Receptor: Least-Cost Case

Receptor Location	120	115	110	105	100	95	90	85
1	67.8	67.4	66.2	66.0	65.3	63.7	61.6	59.3
2	64.6	63.7	62.2	61.8	60.9	58.7	55.5	51.7
3	56.2	56.0	55.5	55.5	55.3	54.6	53.7	52.5
4	85.4	83.9	81.2	78.7	76.8	73.7	70.9	68.1[a]
5	94.3	92.5	89.0	86.2	83.8	80.5	76.9[a]	73.5[a]
6	107.2	102.6	99.7	97.9[a]	95.0[a]	90.7[a]	85.7[a]	80.8[a]
7	116.3	113.8[a]	107.8[a]	104.3[a]	100.0[a]	95.5[a]	90.0[a]	85.0[a]
8	93.3	88.7	86.1	84.4	81.6	75.6	69.9[a]	63.5[a]
9	119.7	115.3[a]	110.4[a]	105.5[a]	100.0[a]	95.2[a]	89.5[a]	84.7[a]
10	52.4	51.6	49.1	47.5	46.0	43.4	40.9	38.2
11	80.2	78.4	77.4	72.0	70.1	68.8	65.7	63.5
12	102.8	101.1	91.9	88.6	84.3[a]	79.7[a]	74.5[a]	69.2[a]
13	61.6	60.8	58.9	57.5	56.0	53.9	51.4	49.2
14	53.3	52.8	51.8	51.2	50.6	49.4	48.1	46.4
15	120.0	114.9[a]	110.4[a]	101.0[a]	99.6[a]	93.0[a]	79.5[a]	53.3[a]
16	56.4	56.4	55.3	55.1	54.3	52.9	52.2	50.9
17	72.4	69.9	66.5	65.1	63.5	59.4	53.1	43.3
18	84.9	84.0	74.9	74.2	73.0	66.4	62.5	55.9
19	51.6	51.4	50.8	50.5	50.1	49.3	48.3	47.3
20	67.3	66.1	64.4	63.3	62.1	60.0	57.5	54.4
21	64.0	63.6	61.2	60.8	60.0	57.1	55.0	52.0
22	64.6	64.3	62.0	61.8	59.7	56.5	55.4	53.1
23	105.3	102.8	98.9	97.7[a]	95.1[a]	90.4[a]	83.8[a]	74.1[a]
Unweighted averages of receptor TSP levels:								
	80.1	78.3	75.3	73.3	71.4	68.2	64.4	59.6
Population-weighted averages of receptor TSP levels:								
	77.4	75.7	72.9	70.9	69.0	66.2	62.9	59.3

Source: Oates, Portney, and McGartland (1989).

[a]Concentration reflected in the calculation of benefits using method (ii).

nal benefits is easy to find. We used a simple quadratic damage surrogate. That is:

(A1) $$G_i = \text{damage at receptor } i = [\text{TSP ppm}]^2 \times 10^3 \text{ (in millions)}.$$

Benefits of increasingly strict standards are then simply

(A2) $$B_i = G_i(120) - G_i(j) \quad \text{for } j < 120 \text{ ug/m}^3.$$

We reproduce here, as table 10A.2, a sample calculation of the damages and benefits for six receptor locations, one standard, and three methods. Inspection of table 10A.1 reveals immediately that method (i) yields an estimate

Table 10A.2 **Examples of Surrogate Damage and Benefit Calculations by Method**

		Method (i) (Modified)		Method (ii)		Method (iii′)	
Receptor	Damages at Base Level	Damages at 110	Benefits	Damages at 110	Benefits	Damages at 110	Benefits
2	4.2			4.2	0	3.9	0.3
7	13.5			12.1	1.4	11.6	1.9
10	2.7			2.7	0	2.4	0.3
12	10.6			10.6	0	8.4	2.2
15	14.4			12.1	2.3	12.1	2.3
18	7.2			7.2	0	5.6	1.6
For average level:	6.4	5.4	1.03×23				
Total			23.6		6.0		19.7
Marginal			11.4		3.4		12.0

Note: For modified method (i), base average surrogate damages = damages at the base average concentration, 80.1. Damages at the 110 standard = damages calculated for an average concentration of $80.1 \times 110/120 = 73.4$. Total damages for every standard are obtained by multiplying the damage associated with the average by 23 (receptors).

of zero benefits for all standards, since the initial average quality is already better than the strictest standard to be examined.

Water Quality Benefits

Water quality benefits are based on predicted water quality improvements in the Delaware estuary published in Spofford, Russell, and Kelley (1976). The quality indicator used is dissolved oxygen (DO) and the base levels are interpolated from their figure 2 reproduced here as figure 10A.1. Improvements associated with alternative standards are taken from table C-3 in the source. Their run, using a 3.0 ppm standard, is used here as a surrogate for a 3.5 ppm standard because in all but one reach, better than 3.5 ppm is attained under it. The predicted levels of DO for that standard and for a run with a 5.0 ppm standard are set out in table 10A.3. The implementation plan implicit in these runs is the least-cost arrangement of discharge reductions.

To calculate benefits, dissolved oxygen is translated into sustainable recreation activities using the table of equivalents developed by Vaughan (1981) and displayed here schematically as table 10A.4. Then the three alternative methods of benefit calculation were applied as summarized in table 10A.5, where the per capita per day values of the alternative sustainable activity measures of quality are drawn from (Smith and Desvousges 1986).

What we have not done is to associate numbers of people with particular receptor locations along the river. ("Receptor location" is usually called "reach" in the water pollution field. It means a stretch of river within which ambient quality is assumed the same.) This is difficult to do in any case without a study to measure the recreational suitability as determined by nonwater

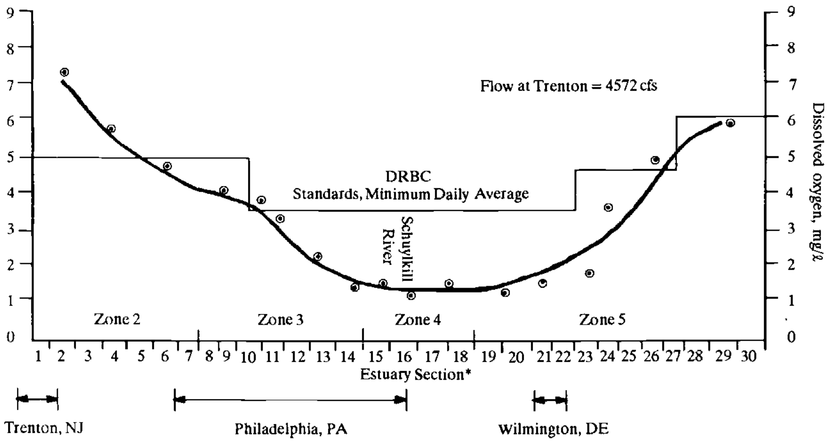

Fig. 10A.1 Delaware Estuary dissolved oxygen profile: July to September 1968
Source: Delaware River Basin Commission, "Final Progress Report: Delaware Estuary and Bay Water Quality Sampling and Mathematical Modeling Project," May 1970, figure 12.
Notes: *Delaware River Basin Commission sections. Note that these section designations differ from the ones in the RFF Study. "—": predicted by the DRBC's Delaware Estuary Model. " ⊙ ": mean of measurements for the three-month period (July, August, September 1968).

quality characteristics. But it is even more difficult to do within a massive urbanized agglomeration such as that which surrounds the Delaware estuary from Wilmington, Delaware, to Trenton, New Jersey. The figures in table 10.5 are therefore simply the sums of the relevant per capita benefits over all the reaches. These figures exaggerate the penalty for ignorance of the environment to the extent that more individuals could easily travel to and recreate on the middle reaches. They are the most heavily polluted, and therefore benefits associated with their cleanup show up in methods (i) and (ii), while any benefits associated with further cleanup of the most upstream and most downstream reaches tend to be ignored in those methods.

Note that the use of Vaughan's equivalence in essence begs an important question: Do we have an environmental quality indicator that is connectable both to discharges and to valued human uses of the environment? Dissolved oxygen is only one of the elements of a vector of water quality characteristics that determine how a body of water can be used. It may be the key element for fish populations but is certainly much less important in determining whether water is "boatable" (that is to say, pleasant to boat on) or swimable (where bacterial counts or turbidity are much more important).

Notes

1. To say that the analysis is difficult (and expensive) is not to say that it is of dubious value. The U.S. Environmental Protection Agency's (1987a) review of its use

Table 10A.3 **Base Case and Predicted Levels of Dissolved Oxygen: Two Alternative Standards Applied to the Delaware Estuary (ppm)**

Reach	Base Situation	3.5 ppm Standard	5.0 ppm Standard
1	8.3	8.6	8.6
2	7.0	7.7	7.7
3	5.6	6.6	6.9
4	4.9	6.0	6.3
5	4.6	5.7	5.9
6	4.4	5.9	6.0
7	3.8	5.9	5.9
8	2.7	5.8	5.9
9	1.8	6.1	6.4
10	1.3	5.3	6.8
11	1.2	3.6	5.3
12	1.2	3.7	6.1
13	1.3	3.6	5.7
14	1.5	4.0	5.7
15	1.8	4.5	6.1
16	2.3	5.2	6.4
17	2.8	3.0[a]	5.0
18	3.5	3.7	5.1
19	4.2	4.8	5.7
20	5.0	5.8	6.1
21	5.8	6.2	6.2
22	6.6	6.6	6.6
Average	3.7	Standard 3.5	Standard 5.0

Source: Spofford, Russell, and Kelley (1976).
[a]The standard actually imposed by Spofford, Russell, and Kelley (1976) was 3.0 ppm. But 3.5 is a lower bound for boatable quality water in the Vaughan scale, so we treat this run as though the standard were 3.5 for purposes of method (ii) calculations.

Table 10A.4 **Water Quality, Recreational Activities, and Associated Willingness to Pay**

DO ppm	Sustainable Activity[a]	Shorthand	Associated Annual Marginal Willingness to Pay per Person ($)[b]
7.0	Swimable (plus fishing and boating)	S	35.4
6.5			
6.0			
5.5	Game fishable (plus boating)	G	19.1
5.0			
4.5	Boatable	B	20.5
4.0			
3.5			
	Unacceptable for boating	U	0

[a]*Source*: Vaughan (1981).
[b]*Source*: Smith and Desvousges (1986).

Table 10A.5 Calculating Surrogate Benefits for Dissolved Oxygen Improvements in the Delaware Estuary by Method

A. Method (i)

Base Case Average:	3.7 ppm (B)	
3.5 ppm Standard:	3.5 ppm (B)	Benefit = 0 × 22 = 0
5.0 ppm Standard:	5.0 ppm (G)	Benefit = \$19.1 × 22 = 420.6

B.	Methods (ii) and (iii)	Marginal Benefits Method (ii)		Marginal Benefits Method (iii)	
Reach	Base Case Sustainable Use	3.5 Standard	5.0 Standard	3.5 Standard	5.0 Standard
1	S	—	—	—	—
2	S	—	—	—	—
3	G	—	—	S (35.4)	—
4	B	—	G (19.1)	G (19.1)	—
5	B	—	G (19.1)	G (19.1)	—
6	B	—	G (19.1)	G (19.1)	—
7	B	—	G (19.1)	G (19.1)	—
8	U	B (20.5)	G (19.1)	G (39.6)	—
9	U	B (20.5)	G (19.1)	G (39.6)	—
10	U	B (20.5)	G (19.1)	G (39.6)	S (35.4)
11	U	B (20.5)	G (19.1)	B (20.5)	G (19.1)
12	U	B (20.5)	G (19.1)	B (20.5)	G (19.1)
13	U	B (20.5)	G (19.1)	B (20.5)	G (19.1)
14	U	B (20.5)	G (19.1)	B (20.5)	G (19.1)
15	U	B (20.5)	G (19.1)	B (20.5)	G (19.1)
16	U	B (20.5)	G (19.1)	G (39.6)	—
17	U	U	G (39.6)	U	G (39.6)
18	B	—	G (19.1)	—	G (19.1)
19	B	—	G (19.1)	—	G (19.1)
20	G	—	—	—	—
21	G	—	—	—	—
22	S	—	—	—	—
Totals		184.5	326.1	372.7	208.7

Note: A dash (—) indicates no improvement in sustainable recreational use over the next lower standard or over the base case as appropriate. S = swimable; G = game fishable; B = boatable; U = unacceptable for boating.

of benefit-cost (B-C) analysis concludes that for three regulatory decisions, B-C analysis identified improvements with potential benefits of over \$10 billion (lead in automotive fuels, \$6.7 billion; used lubricating oil, \$3.6 billion; and premanufacturing review of toxic substances, \$.04 billion). Further, EPA estimates the costs of all regulatory impact analyses (RIAs) done under the terms of President Reagan's Executive Order 12291 as less than \$10 million. Therefore, the return to analytical investment appears to be over 1,000 to 1 in the aggregate.

Several cautions are in order in interpreting this conclusion. Most fundamentally, our argument in this paper, if one accepts it, must inevitably throw some doubt on these benefit estimates. Second, we cannot necessarily project such a return ratio in the future because it is likely that the biggest and easiest targets have already been attacked. And finally, we should include a grain of salt because the self-interest of those preparing the report was consistent with finding large returns.

2. This statutory requirement has not prevented benefit cost information from being included in the RIAs prepared for cases involving the primary standards. The proposed standard subjected to analysis is health based. It is too early to know whether the final standard that emerges after OMB review can be argued to have been affected by the benefit-cost findings.

3. Location is, of course, three dimensional, and altitude can make a big difference in some situations; but the points we make are only reinforced by considering a third dimension, while exposition is much simpler for two.

4. This last point is stressed by Oates, Portney, and McGartland (1989). We shall return to it below.

5. Our discussion assumes that producers will in fact comply with the regulations in question. Making sure this is even roughly the case requires investment in monitoring and enforcement. These costs should be counted as costs of the policy, and their amount and how they are used will help determine the realized level of benefits. It is also true that choices open in the design of implementation systems can affect monitoring and enforcement costs and thus also indirectly affect benefits by that route. We ignore these added complications, though they open up an entirely new and largely unexplored source of demand for data and analysis.

6. Reasonably straightforward theoretical expositions are available that include differential location. See, e.g., Førsund (1972), Tietenberg (1978), and Siebert (1985).

7. The matrix T may be thought of as representing the steady-state solution to a set of differential equations that reflects the transportation of pollution by average winds characterized by velocities and directions, and the diffusion of the pollution particles due to random motion in the plume. If the units of discharge are, say (average) tons per day, the units of the elements of T could be (average) micrograms per cubic meter.

8. For simplicity, it is assumed that the same damage relation applies at each receptor location, though as just stressed, we would expect the damages for a given pollutant concentration to differ across the various points in the regional space.

9. Here we calculate $\{Q\}$ on the basis of D_0 and T, but for the argument that follows, it is important to note that baseline ambient quality is actually realized and therefore can be measured. Thus, there is no inconsistency in assuming knowledge of $\{Q\}$ and ignorance of T. As a practical matter, however, we may very well be ignorant of $\{Q\}$ in any but the loosest, one might say anecdotal, sense. See, e.g., Russell, Vaughan, and Feng (1983). To be useful, our knowledge of ambient quality conditions must be reflected in measurements that are (1) meaningful in terms of their links with or effects on human valuation of environmental services, and (2) connectable to pollution discharges that will have to be altered to change ambient quality. We return to this matter of baseline quality in the final section.

10. The actual patterns of ambient quality produced by the rollback implementation method under the baseline and the alternative standards are:

Receptor	Base	S_8	S_4	S_u	S_2
I	10	8.0	6.0	4.0	2.0
II	8	6.4[a]	4.8	3.2	1.6
III	6	4.8[a]	3.6[a]	1.6	1.2

Superscript "a" indicates a quality level not reflected at all in benefit calculation (ii).

11. It should be emphasized that there is no reason to expect a mathematically desirable—or even smooth—pattern for marginal benefits. The complex relation among standard, discharge reduction amounts and location required under a given implemen-

tation method, and resulting pattern of ambient quality changes, can produce virtually any pattern of marginal benefits.

12. For a description of efforts to use natural world models in water quality benefit estimation, although some of the threshold aspects of the B^2 method are still used, see Vaughan and Russell (1982).

13. This section is based on research undertaken by Yoshiaki Kaoru and Smith and is reported in more detail in Smith and Kaoru (1990).

14. See Mitchell and Carson (1989) for an overview of the issues involved in using these methods.

15. This approach is not completely new to economics. Berndt's (1976) early attempt to reconcile the diverse estimates of elasticities of substitution between capital and labor is similar to our objectives. However, in his case, the focus was on the assumptions inherent in the estimation models and their likely implications for the estimates. Somewhat more closely aligned is the Hazilla-Kopp (1986) summary of their findings on the sensitivity of the characterization of substitution possibilities across different modeling decisions made with the 36 different manufacturing sectors they analyzed. In this case, the analysis parallels what we propose, but their objective was to summarize their own findings, rather than detect sources of differences across studies conducted by different individuals.

16. Thanks are due to Tom Tietenberg for suggesting that we make this point more explicit.

17. Ozone is "secondary" because it is formed in the atmosphere from chemical reactions involving sunlight and certain "primary" or discharged pollutants, especially volatile organic compounds such as gasoline, solvents, and oxides of nitrogen.

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Comment Thomas H. Tietenberg

This is a pioneering paper in the field. Very few authors have taken on the awesome responsibility of assessing the state of the art in data availability for supporting environmental policy. Pioneering papers written by top-notch scholars, such as Russell and Smith, are exciting for the new insights they offer and the possibilities for further research that they uncover. They are also a bit frustrating because they serve to open our eyes to how far we are from complete understanding of the most rational course of action for the future.

In investigating data needs for environmental policy, economists are confronted with problems above and beyond those faced by those collecting data for more conventional purposes. Not only are data on natural systems essential, a point made with appropriate force and clarity in the Russell-Smith paper, but also substantial and vocal opposition inevitably arises whenever the idea of putting a price tag on certain aspects of the environment is discussed.

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The very idea of monetizing our relationship with nature debases that relationship in the eyes of many activists in the area. The fact that much, if not most, of the environmental legislation specifically excludes benefit information from the standard-setting process reveals that this is no backwater view held by a few. In short, we continuously find it necessary to defend vigorously the objective of data collection, not merely to devote more time to thinking up better data to collect.

There is much to admire in this paper and I shall single out some areas for special attention. It represents an inductive approach to setting priorities; Russell and Smith review what has been done as a means of identifying the holes. By drawing upon their considerable experience in empirical environmental policy analysis, the authors are able to isolate some areas where the data needs are apparent. Their conclusions are reasonable and helpful.

Yet I could not help feeling that a deductive approach, perhaps as a complement to their analysis rather than as a substitute, would have been helpful. Such an approach would have set some broad goals for data collection and modeling and then sought to derive priorities from these goals. Motivated by this feeling that a deductive approach has merit and could generate some insights that would be overlooked by an inductive approach, let me briefly explore this idea.

A very simple economics of information model provides the framework for my critique. Information is a scarce commodity and increasing its supply is expensive. Efficient management of information is a corollary to the efficient design of environmental policy. Efficiency is achieved when the value of the marginal dollar spent on data collection and model building is equal to the marginal cost.

The first point suggested by this model is not likely to be a popular one with this audience. *It is not obvious that economists can in general be counted upon to recommend an amount of data collection that would conform to the efficiency condition.* At this kind of conference economists have an understandable bias toward calling for more and better data and being somewhat less sensitive to the costs of this commodity than would be normal for other commodities. Our ability to make unique contributions to policy debates frequently hinges crucially on our ability to back up our arguments with empirical results. Add the realization that we do not directly pay the bill for the requested data makes the urge for more data almost irresistable. To their credit Russell and Smith have not fallen into this trap. At least they have not completely fallen into this trap.

I would like to share with you a few other implications of applying this simple economics of information model to the problem at hand—deducing the needs for improving the data available to assist in creasing more efficient environmental policy. In many, but not all, cases these insights support the conclusions reached in the Russell-Smith paper.

Where should resources be committed first? Recognizing the existence of a budget constraint, what principles should be used to prioritize data collection expenditures? Russell and Smith focus most of their paper on the need for better data collection to support improved benefit estimates to be used in standard setting. In the beginning of their paper they do not really distinguish between data needs for setting *discharge* standards and data needs for setting *ambient* standards.

The economics of information model suggests that *data collected to support the setting of ambient air quality standards probably produces a higher value than data collected in support of setting discharge standards*. The value of additional data is largely determined by its contribution to improved policy. Since the EPA's Emissions Trading Program creates a pressure toward cost-effective air-pollutant discharge standards even when the decisions of the regulatory authority are based on limited or poor information, better data contributes little to this particular standard-setting process.

Why this particular regulatory program has such a remarkable capacity to produce more cost-effective outcomes in the face of limited information is not difficult to understand. Setting standards for each discharge point where air pollutants are emitted is a very difficult task for the regulator. Ideally control costs would play a role in setting these standards, but as a practical matter this is rarely done. The regulatory simply do not possess the requisite amount of information. To compensate for this lack of information, under the Emissions Trading Program the EPA allows various sources to trade control responsibility among themselves, as long as air quality is improved (or at least not degraded) by the trade. Mechanically this is accomplished by certifying any emission reduction that exceeds legal requirements as an "emission reduction credit." This credit then becomes completely transferable and can be used to satisfy the legal requirements at another, presumably more expensive to control, discharge point.

The power of this approach is derived from the fact that individual polluters have very good information on the menu of control options at their disposal, but the regulators do not. This system provides the incentive for those who have the information to use it in socially productive ways, eliminating the need to transfer that information from plant managers to regulators. As is well known, there is every reason to believe that any transferred information would be biased anyway, since regulatory outcomes would be based on it. In the Emissions Trading Program the responsibility for choosing the best outcome was transferred to those with the best knowledge as an alternative to generating more data for the regulators.

While this is a powerful argument against devoting large amounts of resources to better define *discharge* standards, it does not apply to the process for setting *ambient* standards. Since no corresponding market-type process exists for assuring the desirability of the ambient standards, data collection and modeling efforts aimed at improving ambient standards are likely to have a much higher payoff, in terms of improved outcomes, than efforts directed at improving discharge standards.

Not all ambient standards are equally deserving of enhanced data collection efforts. While it is not my intention to lay out before you which ones are the most deserving (the fact that I do not know probably has something to do with my reluctance), a few of the variables that would enter that analysis can be identified, and merely identifying them raises some interesting questions.

For ambient standards the benefits are a function of the magnitude of the individual damage inflicted upon each exposed human, tree, structure, and so on, multiplied by the number of those exposed. Given the sheer number of people and geographic areas exposed, does this imply that the global pollutants (such as those responsible for the destruction of the ozone shield or for global warming) should receive special attention in our data gathering efforts? Historically they have not.

One possible response might be that the effects of global pollutants are likely to occur so far into the future that the present value of more data collection is small at any reasonable discount rate. Is the use of the present value criterion ethically justified in this circumstance where current decisions are likely to have irreversible impacts on the earth's climate? I believe it is not.

The desire for better risk management does not always translate into large expenditures designed to provide better data availability to regulators, especially for employment and product-related risks. An alternative approach is to use the court system to generate information directly for consumers so that they can evaluate the risks they face as an alternative to direct regulation of that risk. For example, the courts have recently made clear to asbestos suppliers that corporate awareness of a risk associated with their products triggers a duty to warn those exposed to it. Failure to respect this duty usually results in the firm being forced to bear the liability for the resulting damage, whereas firms providing adequate warning can escape liability as long as they are not otherwise negligent.

Appropriately applied tort law remedies trigger new data for consumers without the unnecessary intermediate step of supplying more data for regulators. While analysts will still need to contribute to the development of a consistent research methodology for use by the courts in measuring damages, this is a rather different role than deriving national benefit estimates to be used in defining the level of "acceptable" risk. And it implies a rather different data-gathering strategy as well.

Two of the contributions of this paper that I found particularly stimulating dealt usefully and realistically with the problem of constructing reasonable policy in the face of limited data and very short deadlines for the analysis. The first was the rather extended analysis of the accuracy of traditional and widely used rules of thumb in benefit estimation. This is an important issue precisely because it recognizes that information is a scarce commodity. To the extent that simple rules of thumb, which require little informational input, are "in the ballpark," their use may be a preferred solution to gathering all of the costly information necessary to provide a full-blown evaluation. Unfortunately the Russell-Smith results are primarily negative; simple rules of thumb seem to do rather poorly.

The second noteworthy contribution reported on the Smith-Kaoru attempt to determine the degree to which existing estimates can be transferred to new valuation problems. The ability to transfer estimates would facilitate maximizing the value of the limited amount of data allowed by permitting its use for many purposes. Here the preliminary results are mixed. On the one hand their results suggest that estimates cannot be transferred from one recreation site to another by simply noting the differences in site attributes as isolated by hedonic price approaches. On the other hand, the fact that those areas were analytical judgments seem to exert a systematic influence can be isolated at least provides a point of departure for beginning to eliminate those influences.

Since the process that governs the effects of environmental policy is so complex (and therefore the data needed to completely document what is going on would be so expensive to collect), the Russell-Smith admonition that we should pay much more attention to getting the most out of any data that we do collect makes a great deal of sense. Studies designed as an integral piece in a larger research puzzle have rather different characteristics than studies designed to shed light on a single, geographically isolated policy concern; not all urgent policy issues have an equal claim on scarce data-collection dollars. The Russell-Smith recommendation for establishing protocols for research procedures to facilitate the transfer of estimates from one setting to another would be a very good start, providing we can be clear about what are the right protocols.

The large cost of these studies has implications not only for the standardization of research methodology, but also for public-sector research funding. Orchestrated research depends upon orchestrated funding. Since orchestrated finding is more difficult than independent funding, this will impose an additional public-sector burden, especially since the statutory responsibility for controlling environmental problems falls on so many different agencies. However, if the resulting information can be synthesized to produce insights that are more than the simple sum of the conclusions of the individual studies, the results would have a much wider applicability and the expenditures would be easier to justify.

In summary, the environmental and natural resource research community has its work cut out for it in the future. The data needed for "full-information" support of environmental policy is sufficiently costly that it is unreasonable for us to simply expect that all desired data will be forthcoming. Realizing this, it is incumbent upon us to set priorities for data collection, to use the available information more effectively, and, where appropriate, to use innovative means to manage risk, such as involving the court system or artificial markets, as an alternative to transferring a great deal of information to regulators. Russell and Smith have made some very useful contributions in this paper to this emerging field of inquiry. I have tried to complement their analysis by suggesting further considerations for setting priorities and constructing reasonable policy in a very limited information world. Much more remains to be done.

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