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Estimating Water Quality Benefits: Theoretical and Methodological Issues

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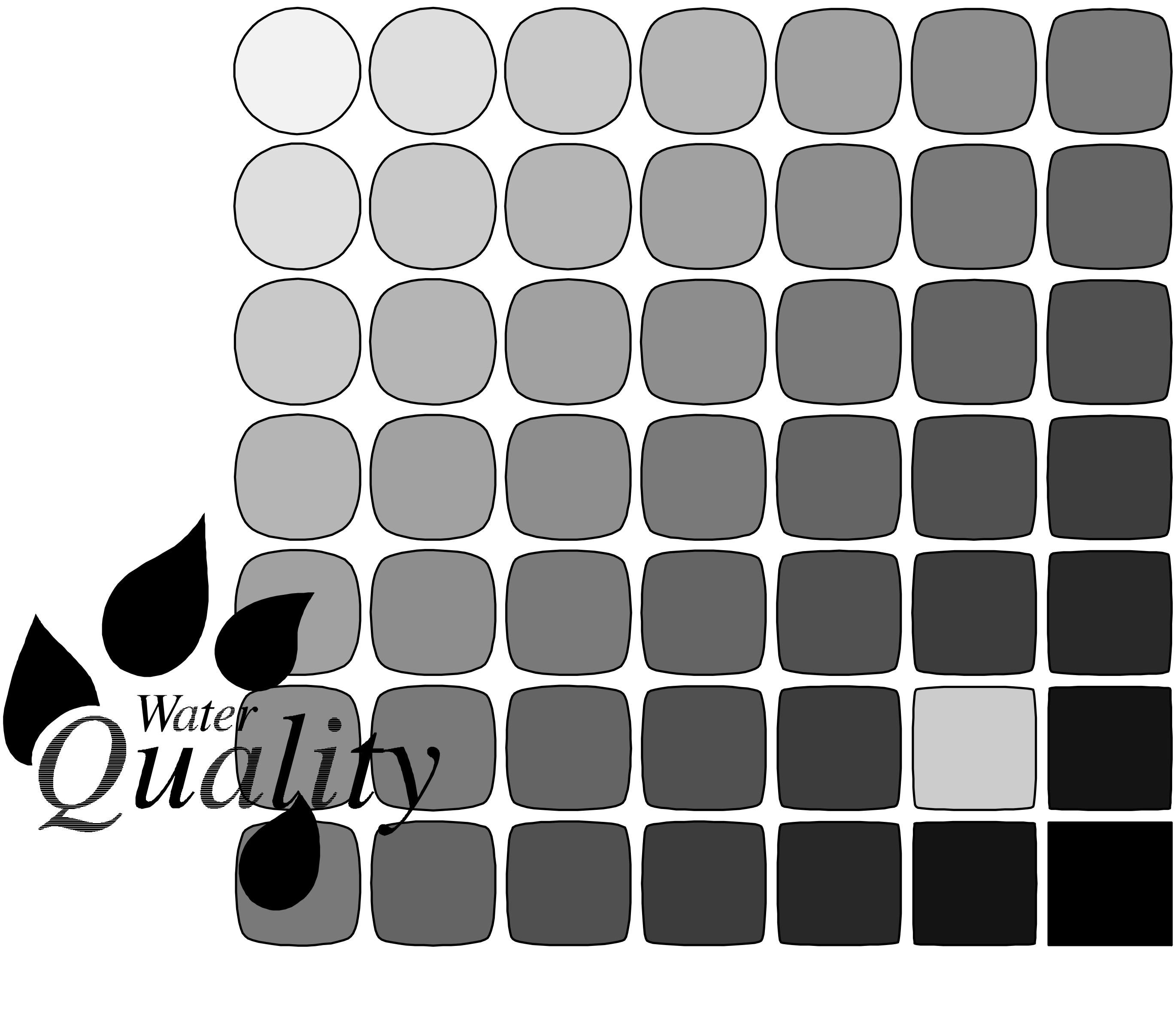

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Abstract

Knowledge of the benefits and costs to water users is required for a complete assessment of policies to create incentives for water quality improving changes in agricultural production. A number of benefit estimation methods are required to handle the varying nature of water quality effects. This report reviews practical approaches and theoretical foundations for estimating the economic value of changes in water quality to recreation, navigation, reservoirs, municipal water treatment and use, and roadside drainage ditches.

Keywords: Benefits, water quality, economic welfare, demand

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Introduction

Knowledge of benefits and costs to water users is required in any complete assessment of policies to create incentives for water quality improving changes in agricultural production practices. Estimating the economic effects of changes in water quality on water users is complicated by the lack of organized markets for environmental quality. There are no observed prices with which to measure value. Instead, economic effects are measured through observed changes in the behavior of water users. The types of water uses affected by changes in water quality include recreation, commercial fisheries, navigation, municipal water treatment and use, and reservoirs.

Benefits or costs of water quality changes are measured through changes in economic welfare, represented by consumer and producer surpluses. A number of methods exist for deriving these measures, including revealed preference, contingent valuation and averting behavior for consumer surplus, and changes in production costs for producer surplus. Each of these methods is reviewed, including theoretical framework, application, and data requirements.

Benefit Estimation Procedures

To gauge the importance of the effects of agriculture on water quality, one needs to measure the benefits and costs to society of programs and policies designed to improve environmental quality. These measures are properly expressed in terms of changes in social welfare, defined as net changes in consumer and producer surpluses. Different programs will affect these components of social welfare to different degrees. For example, pesticide regulation may increase consumer surplus (say, by permitting larger fish populations) at the expense of producer surplus, while cost-sharing of best management practices could enhance both producer and consumer surpluses, excluding consideration of government budgets.

In what follows, we consider both the components of social welfare and the techniques by which they can be measured.

Changes in Consumer Surplus

Environmental quality (EQ) can appear in an individual's utility function either directly or indirectly. Direct effects occur when an improvement in environmental quality increases utility (environmental quality appears separately in the utility function). Indirect effects occur when the personal production function for goods, such as recreation, is affected by environmental quality.

Regardless of how EQ influences utility, the economic benefit of improvements in EQ is best measured as the reduction in income that would keep utility at the original level before the environmental improvement. This measure is known as a compensating surplus, and can be found by computing the area beneath the compensated demand curve for environmental quality.[1] In cases where an environmental change is forgone, equivalent surplus is the relevant measure. This is the amount of income required to move an individual to a new utility level in lieu of achieving that level through an environmental improvement.

These measures are described in figure 1 through the use of an indifference map. The map is defined for environmental quality (X1) and a numeraire good representing all other consumption (X2). The individual is initially at point a on the indifference curve representing utility level U1, consuming X1' units of environmental quality. Suppose that through a policy change, environmental quality increases to X1". The individual is now at point b, consuming X1" of X1 and enjoying a higher level of utility. (The individual has no choice but to consume X1".) Compensating surplus is equal to bd, the amount of income (in the form of X2) that must be forgone to move back to the original utility level U1 while consuming X1". Equivalent surplus is equal to ca, or the amount of income that must be received to move to utility level U2 while still consuming X1'.[2]

In practice, benefits are often approximated by the area beneath the ordinary demand curve. This is the consumer surplus measure. Although formal economic theory dictates the use of a compensated (or equivalent) measure, in most cases the consumer surplus is an adequate

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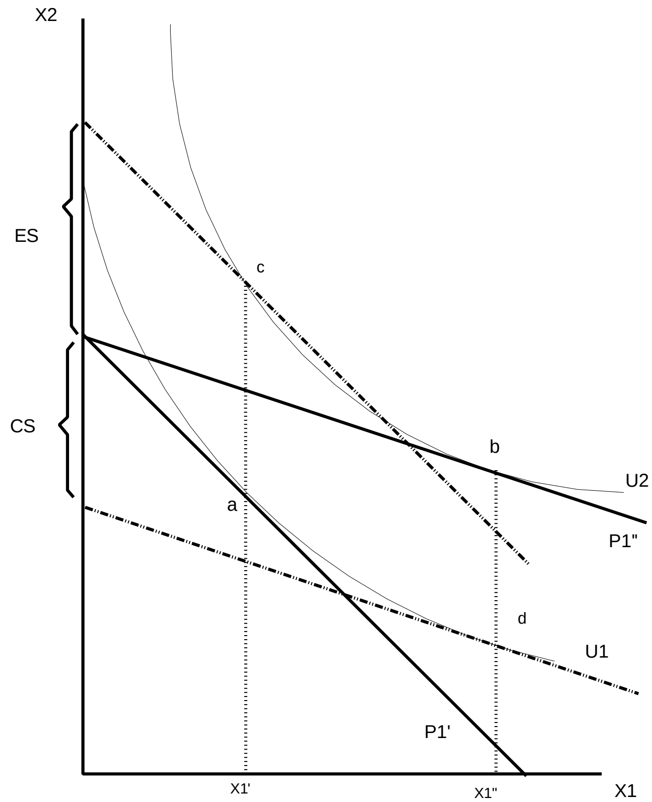
[1] Compensating surplus and compensating variation are two measures of welfare based on the compensated (Hicksian) demand curve. Compensating surplus is used when the quantity of a good is given or restricted (in this case, environmental quantity) and not the choice of the individual. Compensating variation is used when the price of a good changes, and the individual can choose any quantity of the good.

Akin to the compensating surplus and compensating variation are the equivalent surplus and equivalent variation. The difference between the two pairs of measures is due to the base utility level: compensating measures based on the level of utility obtainable prior to the change, and equivalent measures based on the level of utility obtainable after the change.

In actual practice, benefits are often approximated by the area beneath the ordinary demand curve (Willig). This is the consumer surplus measure. The consumer surplus measure is generally easier to estimate because an ordinary demand curve can be estimated with data on observed behavior.

[2] The ES and CS are formally expressed as changes in the expenditure function given a change in the price and level of environmental quality. For example, $CS = E(P_1,Q_1,U_1) - E(P_1,Q_1,U_o)$; where U_o is the base utility, $U_o = V(P_o,Q_o,Y_o)$, and P_o, Q_o, and Y_o are the original price, quality, and income.

Figure 1
Equivalent and compensating surplus



approximation (Willig). Since the consumer surplus measure can be estimated by using data on observed behavior, in this report we often use consumer surplus, or the change in consumer surplus, as a welfare measure.

The problem with estimating demand for environmental quality is that it is not a market good, and its demand cannot be estimated from direct observation of transactions for environmental quality. Instead, there are several general approaches for obtaining demand or benefit information on environmental quality. One approach is to study an individual's behavior in averting the consequences of poor environmental quality, such as expenditures made to prevent household damages from salinity. A second approach is to exploit the relationship between private goods and environmental quality (when it exists) to draw inferences about the demand for environmental quality. A third approach is to ask individuals to reveal directly their willingness to pay (compensating variation) for changes in environmental quality.

Defensive Expenditures

For many water quality problems associated with agriculture, a variety of averting or defensive expenditures can be made by individuals to reduce or completely negate the pollution damage. Purchasing water softeners and bottled water are two examples. Change in defensive expenditure has been shown to be a lower bound estimate of benefits from a reduction in pollution (Bartik; Courant and Porter). The theory from the standpoint of an individual consumer can be shown using the arguments presented by Bartik:

The problem for the consumer is to maximize

$$U = U(X,Q) \tag{1}$$

with respect to X and Q, such that:

$$X + D(Q,P) = Y,$$

where:

X = numeraire commodity,
Q = quality of personal environment,
P = pollution level,
D( ) = defensive expenditure function, and
Y = income.

An example of Q might be the level of cleanliness in a home, where P is particulate pollution and D( ) is the act of dusting. The first-order conditions reduce to:

$$U_Q/U_X = D_Q \tag{2}$$

The household chooses X and Q to equate marginal value of environmental quality to the marginal cost of maintaining that level of personal quality.

The benefits from a reduction in pollution are equal to the income required to keep the household at the original level of utility, given the change in pollution. The indirect utility function expresses utility as a function of income and pollution, the two exogenous variables in the model. As shown in Bartik, the household's maximum attainable utility, v, is equal to the household's utility maximization problem when X and Q are optimally chosen:

$$V = V(P,Y) = U(X^*,Q^*) + \lambda(Y - X^* - D(Q^*,P)), \tag{3}$$

where X^* and Q^* are the optimal quantities of X and Q (given pollution P and income Y).

The benefit of a change in P while V, Q, and X remain fixed is:

$$\frac{\partial Y}{\partial P}\Big|_V = -V_P/V_Y = D_P. \tag{4}$$

The benefit from a small reduction in pollution is D_P, the saving in defensive expenditures needed to maintain the original level of personal environmental quality Q* (and utility) (also shown by Courant and Porter; Harford). The results are similar for nonmarginal changes in P (Bartik).

Actually estimating D_p is not straightforward, since the data requirements for estimating the household demand for personal environmental quality are forbidding (Bartik). The observed change in defensive expenditure given an actual change in environmental quality is not equivalent to D_P. Actual change in defensive expenditure can be expressed as:

$$D(Q_0,P_0) - D(Q_1,P_1). \tag{5}$$

This measure is an underestimate of true benefits (Freeman). In the case of household dusting, the consumer's desired level of personal environmental quality is higher than previously because of the generally cleaner air, and changes in dusting activity partially reflect the new goal.

A lower bound estimate of D_P that requires information only on the defensive expenditure function (D(Q,P)) and household choices before and after the pollution reduction is expressed as:

$$D(Q_0,P_0) - D(Q_0,P_1) \equiv DS \tag{6}$$

DS is a measure of the change in costs to maintain the initial level of household cleanliness (not utility, as in the ideal measure D_P). Bartik shows that DS is analogous to a Laspeyres measure of the benefits of a price reduction and gives a better estimate than the actual change in defensive expenditures. (DS and D_P are exactly the same if the defensive expenditure function is linear.)

Revealed Preference

Water quality is an important factor in many water-based recreation activities. For these "goods," demand for quality (a nonmarket good) can be ascertained through differences in demand for recreation. It is possible to relate variations in quality to changes in demand by making use of the weak complementarity between recreation and site quality (Maler).

Suppose there exists a utility function where utility depends on the consumption of private market goods and environmental quality E:

$$U = U(X_1, ..., X_j, E). \tag{7}$$

If there exists a commodity X_i such that U is independent of E, and if that commodity is not consumed, then that commodity and E are said to be weak complements. This can be shown as

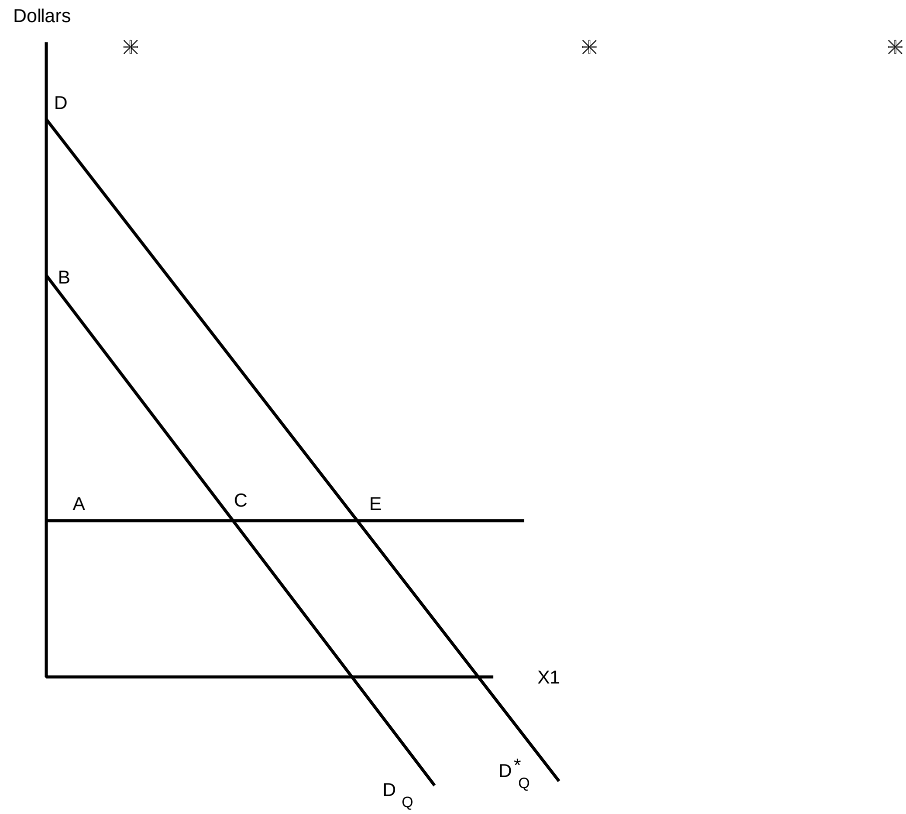
$$U_E(0,X_2, ..., X_j, E) = 0, \tag{8}$$

where U_E is marginal utility with respect to E. In this expression, X_1 and E are weak complements.

When weak complementarity exists, demand models can abstract from E, and assume that utility flows from the consumption of X_i. In a sense, X_i is a proxy for E. Even though human well-being may not be influenced by X_i, since the consumption of X_i is systematically related to the consumption of E, for estimating a willingness to pay for E, one can use the "demand" for X_i instead of the harder to measure "demand" for E.

In figure 2, X1 and environmental quality are weak complements. At the base level of quality, the demand curve for X1 is DQ, and X1' is consumed at price A. An improvement in environmentalquality to Q* shifts demand out to DQ*. X1" is now consumed at price A. The area between DQ and DQ* above the price line, defined by CBDE, is the willingness to pay for the increase in environmental quality, or, equivalently, the area beneath the demand curve for environmental quality.

Figure 2
Weak complementarity and revealed preference.



Transportation to a recreation site and the environmental quality at that site appear to fit this definition of weak complements. One is indifferent to the quality of the environment at that site unless a trip is made (assuming no option or existence value). The travel cost (TC) method is the best known revealed preference technique for recreation valuation. TC analysis correlates the cost of accessing an outdoor recreation site (the travel cost) with the decision to visit sites. A demand curve is typically generated by regressing the number of visits to the site on travel cost and other exogenous variables (presumably, higher travel costs lead to a diminished visitation, other things being equal). Consumer surplus values can be generated from this demand curve for "access to the recreation site."

Consider, for example, the linear travel cost model, where an individual (i) has a demand for trips (T_i) that is modeled as:

$$T_i = \alpha + P_i \beta, \tag{9}$$

with P_i the individual's travel cost, and β the price coefficient. Integrating between P_i and the cutoff price (the price where demand drops to zero) yields the consumer surplus, which in this case can be shown to equal $-Y_{i2}/2\beta$.[3]

While originally constructed to provide the full value of a single site, the travel cost model can be extended to recognize the contribution of particular characteristics of a site to individual welfare. A number of these extensions are considered later in this report, all based on the insight that if a number of sites exist in a region and environmental quality varies across sites, the demand for quality should be reflected in the relative intensity of use of the sites. Furthermore, with extensive data on the recreation activities of the population, this demand can be quantified and used to ascertain the marginal and infra-marginal contribution of improvements in environmental quality.

In addition to influencing consumers' decisions on recreational trips, environmental quality can also influence the decision on where to live. For example, houses adjacent to pristine waterways are probably more attractive than otherwise similar houses located next to polluted waterways. The "amenity value" of a locale's environmental quality can be measured using the property value of homes in the locale (Freeman). If environmental quality decreases, say from increased soil erosion, the value of living in this locale will also fall. This value can then be captured (Lind) by measuring net change in the value of properties in this locale.

Since there are many factors that affect property values, a hedonic property model is often used.[4] The hedonic property model ascribes the value of a house to its characteristics, such as number of bedrooms, size of kitchen, and ambient environmental quality. Formally, the value of a property, i, is: $V_i = f(C_i, E_i, \epsilon_i)$, where C_i are house-specific characteristics, E is environmental quality, and ϵ is a random term. The marginal value of E can then be determined as the derivative of f with respect to E, df/dE. This marginal value can then be used as a measure of the benefit of differences in environmental quality.

Contingent Valuation

Another approach is to ask individuals directly their willingness to pay for a general improvement in water quality. Contingent valuation methods can handle a wide variety of situations, such as cases where the emphasis is not on a particular site or groups of sites, making this approach a very flexible tool.

The goal of the contingent valuation method is very straightforward: to induce people to reveal directly their willingness to pay (WTP) for the provision of a nonmarket good such as environmental quality, or their willingness to accept payment (WTA) to sacrifice the nonmarket good. This involves asking people, in a survey or experimental setting, to reveal their personal valuations ("willingness to pay") for changes in the availability of nonmarket goods by using contingent markets (Randall and others).

The analyst is interested in evaluating the effect on welfare as a good q (say, environmental

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[3]See Bockstael and others (1990) or Hellerstein, for discussion of consumer surplus calculation in linear demand models.

[4]An "open city" assumption (Polinsky and Rubinfeld) is often made, which requires that environmental quality varies across the region, and that movement within the region is unconstrained (there are sufficient numbers of buyers).

quality) changes from a level of q_0 to q_1. A willingness-to-pay measure, in this case the compensating surplus, can be represented as:

$$WTP^* = e(p_0, q_0, U_0) - e(p_0, q_1, U_0), \tag{10}$$

where e is the expenditure function, p_0 is the vector of prices for market goods, q_0 and q_1 are the initial and final quantities of the nonmarket good (environmental quality), and U_0 is utility.[5] In contrast to revealed-preference techniques, which impute a value for WTP^* based on indirectly related measurement, the contingent valuation framework attempts to induce people to reveal WTP^* directly. Highly structured contingent (hypothetical) markets are proposed (in an ideal framework), so that all respondents are confronted with the same, clearly defined situation. There has been a long and continuing debate as to whether the contingent valuation method actually generates meaningful results, but over time and after much study, the method has received greater acceptance (Bishop and Heberlein; Brookshire, Eubanks, and Sorg; Mitchell and Carson).

Contingent valuation, revealed preference, and averting behavior are means by which changes in consumer surplus can be ascertained.

Changes in Production Costs

Environmental quality can be a factor in the production of a market good. In these situations, environmental quality affects the production and supply of a marketable good, and the benefits of changes in quality can be inferred from changes in variables associated with the production of the market good (Freeman). There are two avenues through which benefits can be obtained. The first is through changes in the price of the marketable good to consumers. The second is through changes in incomes received by owners of factor inputs.

Estimating the net effects to producers and consumers from a change in production requires knowledge of the effects of quality on the costs of production, the supply conditions for output, and the ordinary demand curve for the market good (Freeman). In the case of a single-product firm, calculating the welfare effects from a change in environmental quality is rather straightforward. The net welfare gain to society (change in consumer surplus plus change in quasi-rent) is the value of the marginal product of environmental quality (Q) in the aggregate production function for the industry (Freeman and Harrington). Alternatively, net welfare gains are equal to the change in cost with respect to the change in quality. This is the dual of the previous measure.

For the multiproduct firm, the gain in welfare for an increase in environmental quality is also measured by the change in consumer's surplus in the markets for all affected products plus the aggregate change in quasi-rents to the affected firms (Freeman and Harrington). However, measurement of changes in quasi-rents is complicated by jointness of production technology that characterizes multiproduct firms.

There are two special circumstances where extensive information on demand for, and supply of, a good are not required. The first is when environmental quality is a perfect substitute for a purchased input. In this case, improvement in quality results in a decrease of the purchased input. When the

[5] U_0 is the utility obtained at initial prices, income, and q: $V(P_0,Q_0,Y_0)$.

change in total cost does not affect marginal cost and output, the cost saving is a true measure of the benefits of the change in quality (Freeman). An example could be the reduction in chlorine needed to treat water for drinking as ambient bacterial levels are reduced.

When quality is not a perfect substitute, benefits can sometimes be measured by the change in net returns. If the firm is small relative to the output and factor markets, it can be assumed that product and variable factor prices will remain fixed after the change in quality. The increased productivity is expressed as increased profit calculated from farm budget analysis.

Applications to Water Quality

Measuring the economic benefits from agricultural policies related to water quality presents some daunting problems. Agricultural policies affect large geographic areas, presenting problems for any analysis. A substantial number of water bodies (streams, rivers, lakes, reservoirs, estuaries, etc.) can be simultaneously affected. As a result, the value of improved water quality is spread across a wide population of water users, both individuals and business. This spatial dispersion complicates analysis, since environmental data are not collected in a manner to represent large regions. The issue becomes one of how to estimate aggregate benefits: estimate benefits for a sample of water users using detailed quality and economic information about individual sites or water users (such as a travel cost demand equation for a site) and make inferences about the total user population; or use aggregate regional data to estimate regional demands for water quality.

Recreation

The national recreation benefit of a program to improve water quality is the sum of individuals' willingness to pay for the change in water quality. Measurement of these benefits will not be straightforward, since the effects are spread across a large range of sites. Specifically, how will the regions' lakes, reservoirs, streams, rivers, and wetlands be affected by such a policy. Furthermore, in the broadest sense, how are the affected resources used by individuals?[6] The combination of physical impact on water resources and society's uses of these resources determines what sorts of data need to be gathered. These data needs are enumerated in the following material.

Use of Unique Water Bodies

Sites recognized as having unique recreational features, such as State parks, should be examined separately. Either an onsite survey, a survey of a pre-identified user population, or a general population survey that inquires as to the use of these sites can be used to estimate willingness to pay. The choice between these approaches is in large part a function of cost, with relatively minor sites requiring large (hence, expensive) general population surveys before adequate visitation data can be obtained. However, general population surveys are more easily estimated, and are less prone to a number of econometric biases.[7]

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[6]A related issue is what new activities would become possible given an improvement in water quality.

[7]Specifically, onsite surveys are prone to bias from truncation, endogenous stratification, and other forms of sample selection error. While techniques exist to control for these biases (Shaw), they often impose statistical restrictions not required when general population data are available.

Water quality considerations can be incorporated in several manners, such as:

(1) Site choice models (often using discrete choice techniques) which include water quality as an argument in a utility function, with variations in water quality influencing the probability of which site a user selects (Bockstael and others, 1987),

(2) Hedonic travel cost analysis, which regresses the choice of water quality on the incremental costs of accessing sites with better water quality (for example Brown and Mendelsohn; Smith and Kaoru, 1987), and

(3) Generalized travel cost analysis (also known as the varying parameters models), a two-stage model that regresses travel cost coefficients on site characteristics (Vaughan and Russell).

<u>Discussion of site choice models</u>. Site choice models examine the choice of site made by an individual, where at each choice opportunity the individual is faced with many different sites and must choose which site to visit. The individual will presumably choose the site that yields the greatest utility, given the site's environmental quality and cost of access. Information on this "discrete choice," when combined with knowledge of the characteristics (including the environmental quality) of the site, can be used to infer the value of these site characteristics.

For example, suppose that on day t an individual faces a set of J sites, and chooses a site j (j=1,..,J) such that utility (V_t) is maximized:

$$V_t^* = \max_{j \epsilon J} V(P_j, EQ_j, v_{jt}), \tag{11}$$

where P_j is the cost of accessing site j (the travel cost), EQ_j is a vector of characteristics describing the environmental quality at site j, and v_{jt} is a random component that incorporates unobservable factors that influence the individual's enjoyment of site j on day t. Since v_{jt} varies across both site and time, the choice of site is not deterministic. Instead, it is a function of realization of all the v_{jt} (j=1,..,J).

If the distribution of v_j and the utility function are known, it is possible to compute the marginal value of each element of EQ. For example, it is often assumed that V has the following form:

$$V = \alpha\ P_j + \beta\ EQ_j + v_j, \tag{12}$$

where α and β are coefficient vectors to be estimated, and v_j is an independent and identically distributed type I extreme valued random variable. In this case, given information on the outcome of many choice opportunities, a multinomial logit model can be used to estimate α and β (Maddala). Marginal values of site characteristics can then be obtained by comparing β to α.

Variations of this approach, which generalize the above model, have been widely used. For example, Bockstael and others (1987) generalize the distribution of v and add a preliminary stage that predicts the number of choice opportunities. The question of "the number of choice opportunities" is critical when infra-marginal analysis is attempted, since the discrete choice analysis abstracts from the individual's decision about whether to "visit a site" or to engage in some other activity, such as staying home and watching TV (Morey and others).

Discussion of hedonic travel cost models. In hedonic travel cost models, the individual is presumed to derive utility directly from the site characteristics (including environmental quality). In this model, the actual sites are merely particular bundles of site characteristics and otherwise are not unique. If a sufficient number of these "bundles" exist, demand curves can be identified for each of these characteristics. In other words, for each characteristic, the individual can consume a quantity up to the point where the marginal cost of increasing consumption of the characteristics is greater than its marginal value.

A simple example of a hedonic travel cost model uses a two-stage zonal approach. In the first stage, total trip costs are regressed on several site characteristics to calculate the implicit price of each characteristic. Formally, consider an individual i who has M sites to choose from. For each available site m (m=1,..M), the price of access, P_{im} (for example, individual i's travel cost to site m), is regressed against a K x 1 vector of characteristics of the site, EQ_m: $P_{im} = f(EQ_m, \epsilon_m)$, where ϵ_m is a random error term.[8] An underlying presumption of this technical (nonbehavioral) model of implicit prices is that better sites (say, with increasing water clarity) can be obtained by traveling farther. The derivative of f with respect to EQ, dP_m/dEQ_k can then be used as an implicit hedonic price for an additional unit of each of the (K) components of EQ.[9]

The second stage requires that the first-stage regression be performed in many different geographic zones (such as counties). Under the likely case that the distribution of sites varies over space, with individuals living in some zones being close to good quality sites while individuals living in other zones are adjacent to lower quality sites, the hedonic price vector for each zone will also vary. An inverse demand curve for each characteristic (k) can then be formed by regressing, across all individuals (i) in all zones, the hedonic prices (dP_m/dEQ_k) of the characteristic against the quantity of the characteristic demanded (EQ_{ik}): $dP_m/dEQ_k = g(EQ_{ik})$. These demand curves can then be used for infra-marginal valuation (Englin and Mendelsohn).[10]

Discussion of generalized travel cost analysis. Generalized travel cost starts with simple travel cost models, and then correlates the estimated results with measurable site characteristics. Thus, environmental quality affects consumers by modifying the price, income, and other coefficients of the individual's demand curve for the site. In contrast to hedonic models, it is not postulated that consumers explicitly demand a known level of a site characteristic. In this sense, generalized travel cost models are similar to discrete choice models. However, unlike discrete choice models, the total quantity of trips is explicitly modeled, while the choice of "which site to visit, given we are going to take a trip" is not defined.

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[8] "Available" means that the site is not dominated by some other site, with domination occurring when one can obtain the same quantity of characteristics from a site which has a lower price. The (zone-specific) available sites are usually proxied by a list of all sites actually visited by individuals from the zone; nonvisited sites are assumed to be dominated.

[9] If f is linear, then the derivative for each characteristic equals the estimated coefficient. For nonlinear hedonic price equations, additional assumptions may be required (Mendelsohn).

[10] As with discrete choice models, hedonic models focus on a single choice occasion and do not explicitly model the number of times an individual chooses to "visit one of several sites." If the total number of site visits changes substantially when site quality improves (degrades), these infra-marginal values will be biased downward (upward).

Formally, the basic generalized travel cost starts with a set of individual demand curves for m=1,..,M sites:

$$
\begin{aligned}
Y_{i1} &= \alpha_1 + P_{i1}\beta_1 + \epsilon_{i1} \\
Y_{i2} &= \alpha_2 + P_{i2}\beta_2 + \epsilon_{i2} \\
&\cdots\cdot \\
Y_{iM} &= \alpha_M + P_{iM}\beta_M + \epsilon_{iM} \ ,
\end{aligned}
\tag{13}
$$

where Y_{im} is observed number of trips taken by individual i to site m, P_{im} is the price (travel cost) of accessing site m, ϵ_{im} is a random variable, and β_m is the price responsiveness for site m (assumed to be the same for all individuals). Each of these individual demand curves is estimated separately, and estimates of β_m, b_m are derived.

The next step is to regress the estimated price coefficient, b_m, against observed site characteristics:

$$
\begin{aligned}
b_1 &= \gamma_0 + Z_1\gamma_1 + EQ_1\gamma_2 + v_1 \\
b_2 &= \gamma_0 + Z_2\gamma_1 + EQ_2\gamma_2 + v_2 \\
&\cdots \\
b_m &= \gamma_0 + Z_m\gamma_1 + EQ_m\gamma_2 + v_m
\end{aligned}
\tag{14}
$$

where Z and EQ are measures of site characteristics.[11] γ_2 is interpreted as the extent to which price responsiveness changes as these EQ change. Valuation of changes in EQ can be generated by computing the price coefficient, b, at the before and after level of EQ. The difference of the consumer surplus values, with each value derived using a different value of the computed price coefficient (b), will yield an infra-marginal measure of the value of the change in EQ.[12]

While appealing in its simplicity, the generalized travel cost model suffers from a severe problem: the treatment of substitute sites is not internally consistent. The problem, as pointed out by Mendelsohn and Brown, can be seen by noting that the two stages can be collapsed into a single model with $Y_m = g(P_m, EQ_m)$, a model that does not include substitute price. For single-site demand curves, exclusion of substitute sites will lead to a missing-variables problem, resulting in a biased estimate of site demand at any given price (Caulkins and others). For the generalized travel cost models, the consequences are worsened since exclusion of substitute prices is tantamount to assuming that the characteristics of other sites do not affect the demand for a given site. However, if site characteristics are not important in the first stage, characteristics cannot suddenly be the crucial

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[11]Equation 14 differentiates between Z and EQ for heuristic reasons: where Z variables may consist of nonenvironmental variables (say, number of picnic tables in the campground) that may be of lesser interest to the analyst.

[12]This simple model can be expanded, such as by modeling the correlation between the error terms (permitting the joint estimation of both stages) or by using censored demand curves (Smith and Desvousges).

determinant in the second stage. The generalized travel cost technique is basically most useful when applied to a set of unique sites, where each site has no close substitutes, and where each site serves a unique market (no individual ever visits two of the sites).[13]

Use of Nonunique Water Bodies

The use of sites that provide recreation to individuals, but are not distinguished by unique features, should also be examined. For example, a given streamside may be used for a number of purposes (fishing, canoeing), but it has many substitutes; should it disappear, the users can readily find an alternative location that provides nearly the same services. Given the dispersed, low-level usage of these sites, gathering data on visitation would be difficult: any one site will yield only small numbers of visits even in an extensive population survey, while a complete set of onsite surveys would be expensive due to the sheer number of sites and the need for a lengthy survey period to collect sufficient observations. On the other hand, when a complete set of data exists, the sheer volume of information may encumber analysis.

Given these difficulties, some sort of generalization from a subset of sites is necessary. One approach is to use a general population survey to find the visits to all sites, without attempting to obtain thorough coverage of any one site. The sites listed by respondents can then be classified by site characteristics, such as degree of water quality change. Several analytic techniques can be applied to these data, including:

(1) Discrete choice models, which use assumptions about the consumer's utility function to permit discrete choice modeling with randomly drawn alternatives (Parsons and Kealy), and

(2) Multiple-site travel cost analysis, which estimates a demand system where each equation in the system represents a different category of site. Evaluation of changes in water quality can be accomplished by shifting sites between categories, recomputing prices, and using coefficients from the original data to predict changes in demand (Burt and Brewer).

A potential problem with using these approaches is obtaining information on site characteristics, such as water quality. Often, given the myriad of sites, this information will not be readily available. One could measure the characteristics of a subset of these sites, making sure to include sites mentioned by survey respondents, or one could ask survey respondents to describe the visited sites. Another way is to estimate the quality of the sites, given information on average environmental quality of the locale in which the site is located (and perhaps other information obtained from the user).

Description of the discrete choice model using randomly drawn alternatives. The estimation of discrete choice models when full information is not available is relatively simple. The standard discrete choice model is basically modified by using a subset of the sites, instead of using all possible sites. This subset includes the chosen site, and a number (M) of sites randomly drawn from the list of all sites. These M + 1 sites are then used in a discrete choice model, where the M + 1

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[13]An ad hoc method of dealing with this problem is to include a substitute price in a pooled (single) equation version of the generalized travel cost model (Vaughan and Russell). However, this does not address the fundamental problem of incorporating the characteristics of alternative sites in a demand model.

sites vary across observations. As long as the independence of irrelevant alternatives assumption holds,[14] the use of these subsets will not bias the results (Parsons and Kealy).

Note that knowledge of the characteristics of all sites is required, since inclusion of a site implies inclusion of its characteristics in a discrete choice model. In other words, if information is available on only a fraction of the sites that an individual might visit, this technique is inapplicable. Hence, the advantage of this technique is that it greatly facilitates estimation, especially when the full choice set consists of hundreds of sites (with each site represented by a vector of characteristics, and each characteristic akin to a single variable in a standard regression model).

<u>Description of multiple-site travel cost</u>. Multiple-site travel cost models extend the single-site model by classifying sites into a finite number of classes, with each class having similar site characteristics. For example, one class could consist of clear-water lakes, another class of murky-water lakes, and another of clear-water lakes that have periodic algal blooms. Each class of site is then viewed as a separate good. Instead of the demand for trips to a site, the choice variable is the demand for trips to sites of a specific class. The advantage to this approach is that the finite number of classes can be treated as a demand system, with each equation representing demand for trips to sites of a particular class.

Formally, individual i's demand for visits to M different types of sites ($Y_{i1},..,Y_{iM}$) is modeled as:

$$\begin{aligned} Y_{i1} &= f_1(P_{i1},P_{i2},...,P_{iM},\epsilon_{i1}) \\ Y_{i2} &= f_2(P_{i1},P_{i2},...,P_{iM},\epsilon_{i2}) \\ &\;\;\vdots \\ Y_{iM} &= f_M(P_{i1},P_{i2},...,P_{iM},\epsilon_{iM}), \end{aligned} \tag{15}$$

where $P_{i1},...,P_{iM}$ are the prices that individual i faces for each class of site m.[15] The vector of prices is often constructed by measuring the cost of traveling to each type of site for all sampled individuals. Thus, the price vector will vary across individuals, with those who live near (far from) a site of class m having a low (high) P_{im}.[16] After estimating the system, the value of changes in environmental quality can be derived by computing consumer surplus under the pre-existing condition, then reclassifying affected sites given the change in EQ, and recomputing consumer surplus for this new set of sites (Cicchetti and others).

The key to the multiple-site method is the ability to classify sites, permitting the inclusion of substitute prices in the demand system. If each site is unique, sharing little in common with any other site, this may not be possible. However, if there are many sites, and each site tends to have several close substitutes, then this technique can be valuable.

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[14] This assumption states that the ratio of the probabilities of choosing any two sites is independent of any third alternative; that the relative probability of choosing between two sites is not affected by the availability of a third site (McFadden).

[15] For example, the price of class j is the cost of accessing the closest site that falls into site j.

[16] Note that in contrast to the generalized travel cost model, the multiple site model explicitly controls for substitute sites.

Another important issue, especially germane when valuing infra-marginal changes using changes in consumer surplus, is the treatment of cross-price elasticities. If symmetry can be imposed on the demand system (which implies no income effects), then cases where there are multiple price changes (as when a number of sites are reclassified) can be readily valued (Hof and King). However, if this Hicksian condition cannot be imposed, then the value of the consumer surplus cannot be uniquely assigned in the case of multiple price changes.

Hedonic Property Analysis

Hedonic property analysis is similar to hedonic travel cost analysis in that the analyst regresses observed prices against observed characteristics including environmental quality. A linear model is typically used, such as $V_i = \beta_o + \beta_x Z_i + \beta_e E$, with V_i the price of the property, Z the house's characteristics, and E a measure of environmental quality. For example, Young and Teti regressed housing prices on Lake Champlain in Vermont against several house characteristics (such as lot size) and a water quality index. The derivative of V with respect to E (equal to β_e in this simple model) is then interpreted as the marginal value of E. Observed changes, or differences, in environmental quality (ΔE) can be approximated by multiplying this implicit marginal value (price) by ΔE.

Since it may be difficult to gather a complete set of data on house characteristics, one can use data on repeat sales of many different properties (if available), thereby avoiding the requirement for detailed (house-specific) information (Palmquist; Mendelsohn and others, 1992). If house characteristics are likely to have been constant over time, and if environmental quality has changed over time, panel estimation techniques can be used to control for the frequently unobservable house characteristics, thereby improving the quality of inference concerning the influence of environmental quality.

For example, consider a simple extension of the linear model. If Z is constant over time and E has changed, the model becomes: $V_{i,t} = \beta_o + \beta_z Z_i + \beta_e E_{i,t} + \epsilon_{i,t}$. Further assuming that two (or more) observations are available for a set of properties, the simple first-differencing panel model can be used, where:

$$V_{i,t} - V_{i,t-1} = (\beta_o - \beta_o) + \beta_z(Z_i - Z_i) + \beta_e(E_{it} - E_{i,t-1}) + (\epsilon_{i,t} - \epsilon_{i,t-1}). \tag{16}$$

Note that Z_i cancels out; it is treated as an unobservable "fixed effect." Thus, the change in V is seen to be strictly a function of the change in E.

While hedonic property models can be useful when measuring the value of recreational resources, it should be noted that hedonic property models and trip demand models often measure the same thing and that double counting may result if the results of a trip demand model and of a hedonic property model are added together (McConnell).

Contingent Valuation Apart from Water Bodies

As discussed earlier in this report, methods for estimating benefits from water quality changes that rely on site data present a number of difficulties. First, identifying a set of water bodies representative of all water resources affected by a policy is difficult. Second, unless the substitution possibilities between sites are adequately accounted for, independently derived site benefit estimates cannot be aggregated to obtain "true" national benefits (Hoehn and Randall). Contingent valuation is a way to get around these problems since it does not depend on a particular site or sites. Mitchell and Carson have demonstrated that individuals can value water quality in a general sense, given the proper experimental design. The public's willingness to pay for boatable, fishable, and swimmable water quality is an example. In addition, contingent valuation enables one to estimate non-use values, such as option value and existence value.

An advantage of the contingent valuation (CV) is that expansion of the results from a study to a larger population is easier than for methods that are dependent on individual sites. To evaluate the economic benefits from a national policy, a CV survey could be designed to cover the entire country. Such an approach would be very costly, however.

The results of a much smaller scale CV study could be expanded to the entire population in two basic ways: (1) apply an unadjusted willingness-to-pay (WTP) average to the entire population, and (2) apply an econometrically estimated WTP function to the entire population. The former is appropriate in the unlikely event that the sample population is representative of the national population in all socioeconomic categories. A better approach is to estimate an equation that expresses WTP as a function of the socioeconomic characteristics of the sample population (Loomis). Weighted least squares should be used if sample proportions for certain socioeconomic strata do not match population proportions of the same strata (Loomis).

Example: Valuing the Recreational Benefits from the CRP Program

An example of estimating changes in recreation behavior from a change in water quality is the sequential decision model used in the evaluation of the Conservation Reserve Program (Ribaudo and Piper). Recreational freshwater fishing was defined as a two-part decision by an individual: whether or not to fish in a given year (participation model); if yes, how much to fish (intensity model). Each decision was modeled at the national level as a function of socioeconomic variables, supply of surface water, and water quality.

This approach was possible because of the availability of a national survey on outdoor recreation, the U.S. Fish and Wildlife Service's 1980 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation. The survey consisted of two parts: a screening survey of the general population and a survey of those participating in outdoor recreation. The participation model was estimated with data from the screening survey, while the intensity model was estimated with data from those who actually fished. Water quality data from the U.S. Geological Survey's National Stream Quality Assessment Network system were used to calculate average "regional" levels of suspended sediment, nitrogen, and phosphorus.

The participation model was estimated as a logit, and used to estimate the change in the number of people that would participate in recreational fishing given a change in water quality in the regions closest to their homes. The intensity model was based on a travel cost model and used to estimate the change in the number of visits, given a change in water quality. Together, the two models are used to predict the change in regional and national recreational fishing visits given regional changes in water quality. This is accomplished without any information about particular sites.

Navigation

Sedimentation of river channels and harbors can cause delays in shipping and even the loss of vessels. State authorities and the U.S. Army Corps of Engineers share the cost of keeping channels open. Since agriculture is a major source of sediment in many parts of the country, especially in the Missouri and Mississippi watersheds, reductions in cropland erosion could reduce the need to dredge. For a particular harbor or stretch of river, the benefits from reduced erosion would be equal to the net increase in producers' and consumers' surpluses from reduced shipping costs and reduced dredging costs. Under the assumption that reduced sediment and dredging are perfect substitutes, and that dredging prior to the erosion reduction was optimal given the level of sedimentation and the demand for navigation, then the only effect of the reduced erosion is reduced dredging costs. Benefits are equal to the reduced dredging costs.

Estimating benefits from reduced dredging requires data on dredging and shipping activity in each watershed. While such data could be collected through a survey, a simpler approach is to use secondary data. Data on dredging costs and amounts of sediment removed from channels and harbors are available from the Corps of Engineers at the Corps Division level of aggregation. Ribaudo used those data to estimate the removal costs per ton of sediment for each farm production region in an evaluation of the Conservation Reserve Program. Assuming a linear cost function and optimal dredging, one can convert reduced sediment discharge to reduced dredging costs.

Sediment loadings to rivers can be calculated on a regional basis using data from a number of sources. Erosion on various types of land can be estimated with the National Resources Inventory at any number of levels of regional aggregation. There are advantages to using the aggregated sub-area (ASA) or production area in national policy analyses for two reasons. These regions approximate watersheds, and they follow county boundaries. It is therefore relatively easy to create ASA-level data from county-level databases, a number of which exist.

The amount of sediment reaching waterways in an ASA (loadings) can be estimated by multiplying erosion by a sediment delivery ratio (SDR). Resources for the Future (RFF) has calculated SDR's at the Major Land Resource Area level and used these to estimate annual loadings for ASA's. We estimated aggregated sub-area SDR's by dividing the RFF sediment loadings by total erosion.

Since ASA's are connected hydrologically, in that water flowing out of one watershed can enter another, some of the loadings will be carried downstream. The amount of sediment deposited in each ASA can be estimated from the Soil Conservation Service's estimates of inter-ASA delivery ratios.

Reservoirs

Reservoirs make excellent sediment traps. Flowing water can carry large loads of sediment in suspension. When a river's flow is checked by a reservoir, the carrying capacity of the river is greatly reduced, and sediment settles out into the storage basin. Without removal and with a continuous inflow of sediment-laden water, the reservoir will eventually fill with sediment.

A reservoir can provide one or more services such as flood control, drinking water supply, hydroelectric generation, and recreation. The economic costs from sedimentation take three forms: effects on the services provided by the reservoir (such as boating and irrigation), costs of remediation (dredging), and damage to the reservoir structure itself (turbines, pumps) (Southgate and Macke). Benefits from reduced sedimentation are the reverse of the above adverse effects.

Benefits can be expressed as:

$$B = \sum_{t=0}^{T} [P_{1t}\Delta Q_{1t} + P_{2t}\Delta Q_{2t} - \Delta C_{ot}](1+r)^{-t} + \sum_{t=T+1}^{T'} [P_{1t}\Delta Q_{1t} + P_{2t}\Delta Q_{2t} - \tag{17}$$

where:

P_i, Q_i = price and quantity of service flow i,
C_{ot} = operating cost,
r = the discount rate,
T = the economic lifespan of the reservoir given "before" sedimentation rate,
T'= the economic lifespan of the reservoir given reduced sedimentation rate.

The first summation expresses the change in net value of services produced during the without-treatment lifetime of the reservoir. The second summation indicates the net benefits of extending that lifetime. The greater the initial sedimentation rate, the greater the benefits from extending the lifespan of the structure, since T will be small and the effects of discounting will be reduced. Benefits should be estimated for each reservoir affected.

The economic life of a reservoir ends when the incremental benefits from continued use no longer exceed the incremental costs of operation. The economic lifespan of a reservoir can never exceed the physical life. Under ideal conditions (minimal sedimentation) the economic life of a reservoir can be assumed to be long enough so that its capital and operating costs can be amortized as a perpetuity (Lee and Guntermann). Sedimentation shortens the economic life of a reservoir to a finite period, shortening the time over which costs can be amortized and, therefore, increasing annual costs. The excess of annual costs above a perpetuity is a simple measure of sedimentation damages (Lee and Guntermann), assuming no reduction in service flows until the economic lifespan is reached. This definition of damages precludes the need to estimate specifically the demand functions for services supplied.

Most reservoirs have a sediment pool, constructed to trap sediment. No damage from sedimentation occurs until after the sediment pool is filled. Assume that the economic lifespan of a reservoir is equal to the years it takes for the sediment pool to become filled, plus the years for the remaining capacity to become half filled with sediment. Also assume that the flow of services will not be disrupted until the economic lifespan is reached, at which point all services cease. The lifespan of the sediment pool (N_1) can be estimated as follows:

$$N_1 = \frac{kC_{RS}}{G_e A_n DT} . \tag{18}$$

The physical lifespan of the reservoir can be estimated as:

$$N_2 = \frac{kC_{RT}}{G_e A_n DT} , \tag{19}$$

where:

C_{RS} = sediment pool capacity (acre-feet),
C_{RT} = total reservoir capacity, including sediment pool (acre-feet),
k = conversion factor for acre-feet to tons of sediment,
G_e = erosion rate in drainage area (tons/year),
A_n = net drainage area (total area - reservoir surface area),
D = sediment delivery ratio, and
T = reservoir trap efficiency.

Since economic lifespan is assumed to be equal to the years it takes to fill the sediment pool plus half the remaining capacity, economic lifespan can be expressed as the mean of equations 18 and 19:

$$\overline{N} = \frac{N_1 + N_2}{2} = \frac{k(C_{RS} + C_{RT})}{2G_e A_n DT} = \frac{\overline{C}}{G_e L} , \tag{20}$$

where:

C = average of sediment pool and total capacity (in tons of sediment), and
L = product of A_n, D, and T (a constant).

From Lee and Guntermann, the annualized damages from reservoir sedimentation can be expressed as:

$$DS = C_c\left(\frac{r}{1-(1+r)^{-\overline{N}}} - \frac{r}{1-(1+r)^{\infty}}\right) , \tag{21}$$

where:

DS = damages from sediment,
C_c = construction costs, and
r = discount rate.

This equation assumes that annual operation and maintenance costs are independent of the sedimentation rate. The marginal damage with respect to erosion can be defined as:

$$\frac{\partial DS}{\partial G_e} = \frac{\partial DS}{\partial \overline{N}} \frac{\partial \overline{N}}{\partial G_e} . \tag{22}$$

From equation 21, we get the following:

$$\frac{\partial DS}{\partial \overline{N}} = C_c\left(\frac{-r(1+r)^{-\overline{N}} \ln(1+r)}{(1-(1+r)^{-\overline{N}})^2}\right) . \tag{23}$$

From equation 20, we get the following:

$$\frac{\partial \overline{N}}{\partial G_e} = \frac{-\overline{C}}{G_e^2 L} . \tag{24}$$

The marginal benefits from a reduction in erosion in the drainage area of a reservoir are the product of equations 23 and 24.

To include the costs of reservoir sedimentation in an analysis of a national soil conservation program, data on individual reservoir location, size, sediment inflow, and operation and maintenance costs are required. Data on all except operation and maintenance costs are available from sedimentation surveys conducted for the Soil Conservation Service (Dendy and Champion). Operation and maintenance costs are available from a reservoir database maintained by the Soil Conservation Service.

Municipal Water Treatment

Rivers and reservoirs provide drinking water to over 112 million U.S. residents (Solley and others). Water treatment processes are affected by the quality of the source water. Conventional treatment can consist of flocculation, sedimentation, filtration, and disinfection. Intake water with low levels of suspended sediment may be treated by direct filtration, which eliminates the need for sedimentation and, sometimes, flocculation. Cost savings from the use of direct filtration include lower capital costs and lower costs associated with lower chemical coagulant doses and decreased sludge production and disposal. Low turbidity levels also simplify the disinfection process, thus making it less costly. Agriculture is a major source of sediment, thus turbidity, in many parts of the country.

The change in water production cost induced by changes in sediment load is a measure of the welfare effects from soil conservation. For a single-product firm (such as supplier of municipal drinking water), Freeman and Harrington show that the marginal welfare change from an improvement in environmental quality is:

$$W'(Q) = -C_Q(Y^*,Q), \tag{25}$$

where:

W = producer plus consumer surplus,
Y = output, and
$C(Y^*(Q),Q)$ = production cost as a function of equilibrium output Y^* and environmental quality.

If the supply and quality of the output (drinking water) remain unchanged given the change in quality of untreated input water, then consumer surplus remains unchanged and benefits are equal solely to the change in producer surplus (Freeman).

Holmes, using data from 600 of the largest water suppliers, estimated a hedonic model for water treatment costs. A hedonic cost function refers to a cost function that embodies attributes of the production process not typically considered by neoclassical models (Holmes). The quality of untreated intake water is such a variable. The form of his model was:

$$C_j = C_0 Y_j^{a1} Q_j^{a2} (\prod_i R_{ij}^{ai+2}) \ , \tag{26}$$

where:

C_j = expenditure of firm j
C_0 = intercept,
Y_j = water production of firm j,
Q_j = intake water quality of firm j,

R_{ij} = price of input i to firm j, and
ai = model coefficient.

Expenditures were defined as operating costs, including treatment plant costs, water acquisition costs, and distribution costs. Water quality was represented by the turbidity of intake water. Two input prices were included: pipefitter wage and electricity cost index.

With the estimated treatment cost function, estimating regional or national changes in treatment costs from specific changes in turbidity (suspended sediment) is rather straightforward. For a particular region, average change in treatment cost per gallon of water treated can be estimated from data on treatment plants in that region. Costs for all treated water can then be calculated with data from the U.S. Geological Survey on total treated water. This approach is appropriate given the assumption that the change in water quality does not affect price, quantity, or quality of treated water.

Drainage Ditches

When soil is eroded from a field, it can be deposited in roadside ditches, which line many rural roadways. Sedimentation in culverts and ditches reduces the capacity and the effectiveness of the structures, increasing the likelihood of road flooding during storms.

The costs from ditch sedimentation are the maintenance costs of removing sediment plus the damage from road flooding. Optimal cleaning can be defined as that level of maintenance that just prevents road flooding (Freeman). If ditch cleaning is optimal, then the benefits from reduced erosion are equal to the reduction in maintenance costs. If ditch cleaning is not successful in preventing flooding, then reductions in maintenance costs underestimate benefits, in that the damages from flooding will also be reduced by the reduced erosion.

Damages from ditch sedimentation were included in several local studies of the costs of soil erosion (Forster and Abrahim; Moore and McCarl; State of Indiana; Clark and others; Lee and Guntermann). We estimated a national model of ditch-cleaning costs with data from 33 States. Annual sediment removal costs were specified as a function of the cost of removing a cubic yard of sediment and the estimated total sediment discharge along the roads treated by the State. The latter variable provides the link between soil erosion and sediment removal costs. This variable is defined as percentage of rural roads treated multiplied by total discharge of sediment to streams in the State (Gianessi and others). The estimated equation was:

$$Y = 6974 DIS^{.49} UNIT^{.74} , \tag{27}$$

where:

Y = annual sediment removal costs for State roads,
DIS = sediment discharge along roads treated by State, and
UNIT = unit cost of removing a cubic yard of sediment.

All variables were significant at the 5-percent level.

Although the approach is rather crude, it enables a rough-and-ready method of estimating a lower bound of benefits from reduced ditch maintenance costs. The estimate is a lower bound because ditch maintenance usually occurs after the ditch becomes clogged with sediment, and flooding problems have occurred.

Municipal Water Use

The quality of water consumed by households has an influence on longrun costs of maintaining water-using appliances. High levels of total dissolved solids (TDS) or hardness can damage water-using appliances and pipes, increase the use of detergents, and deteriorate clothing and other textiles. Irrigated agriculture has been shown to be a factor in TDS levels in the Colorado River Basin, and possibly elsewhere. Agricultural policies or other policies that affect the TDS levels of large storage reservoirs, such as those in the West and Southwest, could affect the water supplies for major population areas.

Benefits from reduced TDS levels take two forms: increased appliance lifespan for those who take no defensive action and reduced defensive expenditures for those who do (purchase water softeners or bottled water, for example). Most studies have concentrated on the former benefits, including those by Black and Veatch, Metcalf and Eddy, DeBoer and Larson, Tihansky, d'Arge and Eubanks, Patterson and Banker, Lohman and others, Gardner and Young.

Directly estimating households' demand or expenditure function for water quality is not generally possible, since households cannot directly purchase water of varying quality. The approach used most often in the literature is to estimate physical damage in terms of expected appliance lifetimes, assuming that the household would be willing to pay up to the economic value of those physical damages to avoid them (d'Arge and Eubanks, p. 256).

A number of authors report regression equations relating appliance lifespan to TDS levels (Lohman and others, d'Arge and Eubanks, Tihansky). Any of these can be used to estimate household benefits from reductions in TDS levels. The lifespan X_a of each appliance in the study area can be computed for the initial TDS level. Assuming a lifespan X_h of the house, the number of times the appliance must be replaced over the lifespan of the house is X_h/X_a. Assuming that replacement costs are realized at the time of replacement, a cost stream over the lifespan of the house can be constructed, and a present value calculated. A decrease in TDS will increase the lifespan of the appliance and reduce the number of times replacement is necessary. The difference in present value between the two cost streams is the benefits from better quality (for example, less saline) water. Census data on the number of households and the types and numbers of appliances per household allow one to develop an aggregate estimate of benefits for a metropolitan area influenced by improved household water quality.

Conclusions

Evaluation of policies that address the effect of agricultural activities on environmental quality will require two types of information: information on the cost of implementing the required changes in agricultural practices and information on the benefit to society of the resulting improvements in environmental quality. This report reviews a variety of methodologies that can be used to compute these benefits. It particularly focuses on benefits related to improvements in water quality.

These benefits can be broadly classified into two categories: those that accrue to consumers, and those that accrue to producers. To quantify these benefits, we estimate changes in consumer and producer surplus. From a theoretical standpoint, the appropriate welfare measure for each type of water quality effect is clear. However, estimating benefits is complicated by several factors.

In particular, the nonmarket nature of improvements in environmental quality makes estimation of welfare effects difficult. Nonmarket goods, by definition, cannot be obtained through market transactions. Therefore, traditional demand and supply analysis will often be impossible since observations on market behavior are lacking. Given the lack of market data, computation of consumer and producer surplus measures is performed using a variety of indirect methods. For consumer surplus measures, we reviewed three broad classes of techniques that accomplish this indirect measurement: techniques that examine averting behavior, techniques based on consumption of complementary market goods, and techniques that use contingent valuation. Each of these classes has strengths and weaknesses. Averting behavior has limited applicability, but can be directly tied to environmental damage. Revealed preference is based on actual behavior, but requires a number of statistical assumptions that are often of a heuristic (as opposed to a strongly theoretic) nature. Contingent valuation obtains a direct measure of the benefit, but is based on intended rather than actual behavior.

For producer surplus methods, we review two broad indirect methods: defensive expenditures and changes in production costs. Defensive expenditures are relatively easy to measure, but the method depends on the assumption that defensive activities are performed in an economically efficient manner. This assumption is probably a poor one. In many cases, defensive services are provided by public authorities, and the level of activity reflects budget and bureaucratic realities more than actual needs.

Measures of change in production costs and/or economic profits may be the closest to the ideal measure of all the procedures reviewed. Inputs and outputs of the production process are generally priced in the market, and one can assume that the production process itself is efficient. For a given change in environmental quality, the measure of benefits should approach the ideal.

In addition to broad strengths and weaknesses, the data requirements of these techniques must be considered. For example, demand analysis is often best accomplished through a general population survey. However, when time and expense prevent the collection of such data, one is often forced to second-best solutions. These solutions include surveys of pre-identified users, or surveys conducted on-site. In such cases, zonal aggregates can be used to extend the range of the data, or sophisticated statistical techniques can be used to reduce the biases introduced by less than optimal sample design. However, these methods usually entail a worsening of model accuracy, and a reduction in the robustness of the model to misspecification. In other words, the poorer the quality of data, the greater the dependence on statistical fixes.

Benefit assessments face another complication besides the lack of market data, namely a lack of cause-effect information necessary for a priori evaluations. In particular, in order to model how changes in environmental quality affect consumer and producer well-being, knowledge of how consumers and producers use environmental services is required, as well as the ability to predict how a policy will affect the relevant environmental variables. The links between on-field agricultural activities and environmental quality are known in a qualitative sense, but fate and transport models that enable prediction of, say, the change in the quality of water entering a particular water treatment plant are lacking, especially in the context of a national policy. In such cases, a second-best solution is to use aggregate, zonal measures of environmental characteristics, or to predict relevant environmental characteristics from secondary data sources. While such techniques provide some insight into the relationships among agricultural production, the level of environmental characteristics, and human behavior, they risk serious misspecification, especially when the environment is very heterogeneous (when intra-zone variability is high).

A related issue is how to conduct a national analysis when the available data are of a very local nature. Even if a number of site studies have been conducted with the optimal data, the problem still exists of aggregating the results to a regional scale that is relevant to policymakers. In this area, research may contribute the greatest improvements in the quality of policy analyses. One benefit transfer approach is metamodeling. A metamodel is a statistical model of a simulation or other estimated model that enables the model results to be transferred from the situation where it was developed to a new application or population (Smith and Kaoru, 1990).

A prerequisite for any type of benefits transfer exercise is the availability of a national database of the supply of water resources (quantity and quality) and the demand for water resources by individuals and industry. A detailed database on a watershed basis (such as the aggregated sub-area) needs to be developed and maintained so that the basic environmental and demand data necessary for any national analysis are available.

In summary, the tools for estimating micro-level benefits have been developed and, for the most part, have received extensive professional application and review. Marginal improvements are likely to be made, but methods are available for most, if not all, types of benefits. The greatest challenge lies in the area of national-scale benefit estimates. An analyst has the choice of either collecting enough data at the national level to minimize the need for assumptions or statistical fixes, or to use less intensive data and find ways of extending micro-study results. Research is needed in both these areas.

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