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The Architecture and Measurement of an Ecosystem Services Index

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RESOURCES FOR THE FUTURE

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Abstract

This paper describes the construction of an ecological services index (ESI). An ESI is meant to summarize and track over time the magnitude of beneficial services arising from the natural environment. A central task of this paper is to define rigorously ecosystem services so that services can be counted in an economically and ecologically defensible manner—a requirement if ecological contributions to welfare are to be incorporated into the national accounts. This paper advocates a particular economic structure and relates it to index theory and makes concrete recommendations for the measurement of such an index.

Key Words: ecosystem services, Green GDP, index numbers, ecological economics

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The Architecture and Measurement of an Ecosystem Services Index

Spencer Banzhaf and James Boyd*

1. Introduction

Joint models of ecosystems and of economic activity have played an important role in environmental policy since the seminal work of Kneese and Bower (1968). The interface between ecology and economy remains a priority for decisionmaking and a challenge for the sciences. Separately, ecology and economics have advanced much faster than the integrated methods necessary for real descriptive and predictive power. This paper confronts the challenge of "real descriptive and predictive power" by drawing on ecological principles, economic and biophysical data, and the economic theory of index numbers to propose indicators of ecologically derived well-being that can be used in policy analysis and interdisciplinary models.

In particular, the paper describes the construction of an ecological services index (ESI). An index, whether it is an ESI or something more familiar, such as gross domestic product (GDP), is meant to summarize a collection of disparate elements. The particular goal of an ESI is to create a measure of the contribution of ecological assets and functions to human well-being over time.[1] At the outset we distinguish an ESI, which we label an accounting measure of ecosystem services, from willingness-to-pay measures of nature's total value. An ESI is a measure of quantity that relates to, but does not measure directly, the total value of nature. In proposing an ESI, we hope to advance an agenda articulated by the National Academy of Sciences to develop a set of comprehensive non-market economic accounts to provide a fuller picture of the economic state of the nation.[2] A stand-alone index of ecosystem services also would serve as a useful government performance measure and a yardstick of gains and losses to environment-related well-being.

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[1] This goal is conceptually distinct from the "productivity indices" of Färe et al. (2004), which compare good outputs to pollutant outputs.

[2] This agenda is not confined to the environmental area. Environmental accounting is the focus, however, of the academies' call for an environmental satellite index (Nordhaus and Kokkelenberg, 1999; Nordhaus, 2005).

There are three fundamental challenges to the construction of an ecological index. First, ecological services must be measured and expressed in defined, standardized units. Markets tend to standardize units of consumption for conventional goods and services. Recognizable units aid marketing, bargaining, and the resolution of disputes. By contrast, ecological outputs are often public goods not associated with markets and thus tend to lack standardized measures of output. Second, given the lack of markets, we also lack prices or other convenient weights to reflect the relative value of ecosystem outputs. Third, ecosystem service flows depend on biophysical stocks. Think of these stocks as the biological and physical assets that produce ecosystem services. They include clean air, water, habitat, and biotic populations. Like the services they generate, they typically are not defined or valued by markets. Ecosystem stocks can be thought of as claims on future ecosystem service flows. Accordingly, an index should measure not only current flows but also discount the current value of flows to account for gains or losses in stock assets that affect future service flows.

The distinction between ecosystem services and ecological stocks is important not only to the formal construction of an index but also to the measurement of environmental outcomes and interactions between economic and ecological science. As will be clear, an ESI requires both economic and ecological content and analysis. Indeed, a goal of this paper is to assess the utility of existing ecological indices and indicators in the construction of an index such as ours. We argue that in economic terminology most theoretical and empirical ecology is geared toward the depiction of ecological asset qualities and interdependencies between ecological assets. Accordingly, ecological models and indicator systems play the central role in the assessment of stock effects. However, biophysical science is not geared toward the assessment of service flows arising from those assets. This has important implications for the use of ecological data and models in the construction of an ecosystem services index.

1.1 Why Construct an ESI?

There are three parts to the question of why construct an ESI. First, why is a measure of ecological conditions needed? Second, why measure ecological conditions as they relate to economic benefits or services? And third, why construct an index to measure these services? With regard to the first question, indicators of ecological condition may have several purposes. Most simply, they may help us to satisfy our natural desire to evaluate the state of things by comparing differences across space and tracking changing over time. In this way, indicators help us to answer the question "how are we doing?" Accordingly, they also may help identify problems on the horizon or help us evaluate the success of past policies, if only indirectly. Measurement of ecosystem services in a manner consistent with the national accounts also will ultimately help enrich national accounting measures such as GDP. Finally, in joint models of ecological and economic systems, indicators of ecological condition play a crucial role in the feedback between the two systems.

The second part of the question—why measure the economic characteristics of ecological conditions—is obvious to economists but may not be to others. We emphasize economic indicators of ecosystem services for two broad reasons: one philosophical, one pragmatic. As a matter of philosophy, economists believe that the objective of social policy is to maximize as best as possible human well-being. When we measure ecological conditions alone it is certainly far better than doing nothing, but it neglects deeper inquiry into what is socially beneficial about ecological systems. The second reason to assess human services provided by the natural environment is that it illuminates tradeoffs and the setting of priorities. Indicators of pure ecological condition do not help a decision-maker forced to choose between conflicting interventions. To say that better ecological conditions are better for society is true but unhelpful. In practice, decision-makers struggle with much more difficult questions, such as *which* ecological conditions are better than others? The economist's answer is that measures of social well-being—the services provided by nature—should guide the choice.

It should be emphasized, however, that any economic assessment of ecological benefits must be built on a foundation of biophysical analysis. The economist's role is to evaluate the consequences of an ecological change to social well-being. Economists rely on the biophysical sciences to describe those changes. Without biophysical assessment there can be no economic assessment. Accordingly, ecological benefit analysis is inherently integrative, demanding cross-disciplinary understanding, if not collaboration. The ESI architecture we advocate embodies that principle.

The third issue is why capture ecosystem services via an index? An index is designed to aggregate a broad range of services into a simple measure, with breadth referring both to the type of services and their spatial and temporal distribution. The issues we discuss apply to a national ESI as directly as they apply to an ESI at other meaningful scales, such as a watershed. Such aggregation and simplification can be useful to policymakers (and the public) by illuminating the big picture and by reducing a potentially vast array of signals into a smaller dimension that can be more easily processed by decision-makers. By the same token, by reducing the dimensionality of the problem in a theoretically consistent way, indices can be useful in models of the interaction between ecosystems and economic systems. For example, equilibrium or simulation models of an ecosystem and economy must constrain the dimensionality of the problem within computational limits. Creating one or a few aggregate flow(s) of services from the ecosystem to the economy is one way to do so.

1.2 Outline

This paper proceeds as follows. The next section formally derives the elements and structure of an ESI and advances a formal economic definition of ecosystem services. Section 3 illustrates the definition in practice and contrasts our accounting-based definition of services with other uses of the term in economics and ecology. Section 4 reviews indicator systems arising from ecology and relates them to our accounting framework. As we will argue, existing ecological indicators are a complement to—but no substitute for—an economically derived index approach. Section 5 turns to the weights given to ecosystem services in the index. We suggest one strategy of using welfare-based indicators to construct willingness to pay indices.

2. The Architecture of an Ecosystem Services Index

We begin by making a basic distinction between the ecosystem itself and the services it provides. We denote by Q an ecosystem, which may be partitioned into a set of ecosystem components $\{Q_1,\ldots,Q_j,\ldots,Q_J\}$, each with some physical or utility-based measure of its quality or health. The J parts of the ecosystem may be distinguished by media, spatial location, hydrological function, or biological function (e.g., place in the foodweb). While the ecosystem Q is an abstraction, $Q_1\ldots Q_J$ are in principle measurable entities.

2.1 Ecological Services, Ecological Assets

The ecosystem provides a set of services $\{q_1,\ldots,q_i,\ldots,q_I\}$ enjoyed by people. This flow of services is derived from the ecosystem according to an ecological production function[3] that is a mapping from R^J to R^I:

$$(1) \qquad \boldsymbol{q} = F(\boldsymbol{Q}).$$

These services are flows consumed directly by households or other economic agents, whereas the ecosystem components are stocks—or ecological assets—that are necessary inputs to the provision of services. In some cases, the relationship represented by F( ) is likely to be highly complex and non-linear. In other cases, however, it may be trivially simple. For example, if Q_j represents the stock of bald eagles, and if households have existence values for the total number of bald eagles, the stock yields an existence–value service flow $q_i = Q_j$.

In most modeling and policy applications it is necessary to reduce the dimensionality of $\boldsymbol{q}$ or $\boldsymbol{Q}$. If a representative preference aggregator existed and were known globally, it would serve as a perfect welfare aggregator. Linear indices, such as those used in the national accounts, are more feasible. A linear index is merely a weighted average of its components.

2.2 A Revenue Index

Consider first one particular linear aggregation: the sum of ecosystem services weighted by their marginal values or "virtual prices." Marginal values are not the only conceivable weights but in this case reflect our anthropocentric approach to measuring services. The sum of ecosystem services weighted by marginal willingness to pay (prices) can be thought of as a revenue index since it represents the revenue that could be obtained by hypothetically selling all services at marginal values. The revenue index at time or place *t* is

[3] In economics, we often think of a physical production possibility set from which choices can be made. Nature here is essentially a physical feasibility constraint plus a choice function.

(2) $R^t = \Sigma_i q_i^t p_i^t$.

Such an index may be a useful proxy to policymakers in a variety of settings. Suppose for example that we must choose between the protection of two wilderness areas designated by 0 and 1. Using the revenue index at each site, we would compare $\Sigma_i q_i^0 p_i^0$ to $\Sigma_i q_i^1 p_i^1$ and preserve the one with the greatest value.

However, as an aid to policymaking, the revenue index has two main problems. The first relates to the index's reliance on marginal values. Suppose both sites are in areas with equal populations of similar incomes and tastes and where there are similar opportunities for substitution. Suppose site 1 offers more services in the sense that $q_i^1 > q_i^0$ for all services i. This increased supply of services from site 1 would drive down their marginal values (because of diminishing marginal willingness to pay) so that $p_i^1 < p_i^0$ for all i. Depending on the elasticity of demand for the services, it could drive down marginal values so that the sum of products too is lower ($\Sigma_i q_i^1 p_i^1 < \Sigma_i q_i^0 p_i^0$). Thus, the index would give a lower value for site 1 even though starting from the same baseline it provides everything site 0 provides and more and moreover provides it to identical populations.

This distortion arises from the use of marginal, rather than total, willingness to pay. For measures of total value, consumer surplus is more desirable than measures based on marginal prices, as Pigou noted in his early discussions of GDP.[4] Prices are used for the simple reason that they are easier to measure.[5] Note that consumer surplus for the services at site 1 would indeed be higher in this example.

The second, related difficulty with this revenue index is that both the quantities of the services and the weights vary in the comparison. When both are allowed to vary, it is impossible to tell whether changes in the index arise from changes in the level of services or from changes in the weights applied to those services. Index numbers are more useful when comparisons are

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[4] The infra-marginal measure, consumers' surplus, rather than prices, is the appropriate neo-classical measure of welfare from consumption. That is, for each site, $\Sigma_i q_i^1 p_i^1$ is merely a (here, poor) proxy for $\int\ldots\int D(p_1,\ldots,p_I)dp_1\ldots dp_I$, where D(.) is the I-valued demand function.

5 According to Pigou (1932) in his seminal text on index construction, the data necessary to compute willingness to pay "are not, and are likely not, within any reasonable period of time, to become available to us…the only data which there is any serious hope of organizing on a scale adequate to yield a measure of dividend changes are the quantities and prices of various sorts of commodities" (p. 57).

made based on changes in service levels, holding weights constant across points of comparison, or on changes in the weights, holding service levels constant. The revenue index is analogous to nominal GDP, where both the quantity of output and prices are changing. Such an index is useful in some contexts but must be used and interpreted with care.

2.3 The Index Number Problem

A goal of the economic theory of index numbers is to factor, or separate, value into its two abstract, aggregate components: outputs of goods and services and the value of those goods and services.[6] Factoring in a manner that is logically consistent and that accords with economic theory is what Irving Fisher called "the index number problem." In our case, the index number problem is to consistently define ecosystem service indices S^t and value indices P^t so that for any two contexts 0 and 1

$$(3) \qquad R^1/R^0 = S^1P^1/S^0P^0 = (\mathrm{S}^1/\mathrm{S}^0)*(\mathrm{P}^1/\mathrm{P}^0).$$

The relative change in the revenue index is the product of the relative changes in the ESI and an index of the value of those services.

Taking the ESI first, the most conventional approach is to choose a constant set of weights and apply them to the changing bundle of services or other outputs. For comparisons of a site 0 and 1, candidates include p_i^0 and p_i^1 or any other set of constant weights (such as some average of p_i^0 and p_i^1). That is, we could compare $\Sigma_i q_i^0 p_i^0$ with $\Sigma_i q_i^1 p_i^0$ or, alternatively, $\Sigma_i q_i^0 p_i^1$ with $\Sigma_i q_i^1 p_i^1$.

Accordingly, we denote an ESI at time or place t by

$$(4) \qquad S^t = \Sigma_i q_i^t p_i$$

[6] Again, we acknowledge other types of index numbers, such as measures of efficiency and productivity (e.g., Färe et al. 2004).

where q_i^t is the level of service i at time or place t and p_i is its (constant) weight. A change in the index of the quantity of ecological services is well defined when the weights are constant. The relative change in the index, a portion of Equation (3), is

(5) $S^1/S^0 = \Sigma_i q_i^1 p_i / \Sigma_i q_i^0 p_i$

for two time periods or places 0 and 1.

Continuing our analogy to national income accounting, the ESI is like real GDP; it is a measure of quantity holding prices constant. That is, for a given constant set of weights, we can say which site or point in time provides a higher level of services. If the rank ordering is the same for any set of reasonable weights, we can be confident in that ordering. Otherwise, such as when weights change over time, the best we can say is that the comparison will be dependent on one's perspective as represented by the weights.

The ESI's parallel index in equation (3), and the second part of the index number problem, is an index of marginal value. A marginal value index at place or time t is computed using a fixed bundle of services and denoted by P^t, where

(6) $P^t = \Sigma_i q_i p_i^t$

Used alone, the value index allows us to compare $\Sigma_i q_i^0 p_i^0$ with $\Sigma_i q_i^0 p_i^1$ or $\Sigma_i q_i^1 p_i^0$ with $\Sigma_i q_i^1 p_i^1$.[7]

Again using our analogy to national income accounting, the marginal value index is like a measure of inflation. That is, it gives a measure of the relative marginal value of an additional

[7] For full consistency, such a value index, paired with its symmetric service index, in general will not consistently factor the revenue index (Fisher 1923). One of the two must be defined as an implicit index (i.e., as the revenue index divided by the other) or specific indices must be used, such as the geometric means of the two weights or the Fisher index.

increment of a fixed basket of ecosystem services at any time or place.[8] If for any set of constant q's the value is higher at one site or point in time, we can say that that location or point in time values an increase in ecological services more highly. Accordingly, a marginal value index has policy interest of its own. In particular, it can signal the relative value of marginal improvements to ecosystem services (taken as a whole) in various contexts. If the comparison is across time rather than space, it can signal the changing value of the ecosystem services.

2.4 Does an ESI Measure the Value of Nature?

We take pains to distinguish between an ESI, which we label an accounting measure of the quantity of ecosystem services, and measures of nature's total monetary value. An ESI relates to, but is not a direct measure of, the total value of nature. One way to illustrate the distinction between an accounting measure and a monetary valuation is to contrast our approach with that of the most famous (and controversial) attempt at comprehensive ecosystem valuation: "The Value of the World's Ecosystem Services and Natural Capital" (Costanza et al. 1997), in which the authors placed a value of $33 trillion on the world's ecosystems. The Costanza et al. paper arrived at the value of nature by multiplying a variety of measures of ecological services q by estimates of those services' marginal values p.[9]

As a measure of value, the Costanza et al. paper has been criticized for combining separately measured values for individual resources without accounting for the fact that the aggregate value is not necessarily the sum of the parts and must be limited by people's ability to pay (Bockstael et al. 2000). As a consequence, their measure of value has the strange feature of exceeding global income and does so by a wide margin.

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[8] This "fixed basket" interpretation of a price or marginal value index contrasts with the cost-of-living approach adopted in the "green price indices" suggested by Banzhaf (2004). That approach allowed increasing quantities of public goods to partially offset the effect of rising prices when computing the change in income required to maintain the standard of living.

[9] Costanza et al. prefer to use measures of average surplus to marginal values/prices, which generally is more appropriate. However, since all empirical estimates of surplus are in the context of small changes—relative to the global elimination of all ecological services—average consumer surplus over the relevant ranges still approximate marginal values.

Mechanically, our ESI bears a resemblance to the work of Costanza in that we too measure a kind of "revenue of all services" when we multiply services by a constant set of prices (i.e., marginal values).[10] However, as an accounting measure of ecosystem service quantity, in contrast to a measure of total value, our measure is not subject to the same criticism. In particular, there is no reason for a quantity measure to be limited by budget constraints. The quantity measure we derive employs virtual, not real, prices. Although still equal to marginal willingness to pay, in a general equilibrium system virtual prices are the prices that people would hypothetically pay for ecosystem services if their incomes were augmented to cover the necessary expense ("virtual income"). The only constraint in the accounting context, then, is that expenditures on market goods and hypothetical revenues to ecosystem services are less than actual income plus virtual income.[11]

All this said, there remains a connection between our measure of ecosystem services and welfare. If the quantity of some ecosystem services (which are assumed to be goods) increased, and none decreased, welfare must be increasing. Intuitively, even if some services decrease, if the aggregate measure of services, as indicated by the index, increases, welfare should still be increasing. In fact, changes in an ESI can be considered a first-order approximation to changes in welfare. In this sense, though not a welfare measure, an ESI is a welfare indicator.

2.5 Ecological Depletion

An accounting concept with particular relevance to the construction of an ESI is net GDP. Net GDP is the economist's preferred measure of economic output because it accounts for depreciation of capital.[12] Net GDP subtracts income based on an unsustainable depletion of the income-producing assets of the economy. If a factory produces extra automobiles one year, but

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[10] It should be noted that Costanza et al. were clear about some of the methodological liberties taken to arrive at their estimate of value.

[11] To put it in still other terms, if our ESI were combined with market outputs in a measure of "green GDP," there is no reason why the green component could not be greater than the conventional component.

[12] See, for example, Nordhaus and Kokkelenberg (1999). ("While the emphasis on GDP rather than net national product is understandable, the panel emphasizes that the latter is conceptually preferable as a measure of sustainable income," p. 27.)

only at the cost of increased wear-and-tear on its equipment, the value of that wear-and-tear is deducted from GDP to obtain net GDP.

Depreciation is related to what is sustainable in the longer term. If we "rob the future to pay for the present," this should not appear as an increase in current output because the increase is not sustainable. With depreciation adjustments, the expected loss of future income is booked today. If large quantities of capital could be traded for services at prevailing prices, net GDP hypothetically would represent the greatest level of income sustainable forever. As shown by Weitzman (1976), the present value of a constant stream of income evaluated at net GDP also can be interpreted as the welfare of the actual stream of income along an optimal path.[13]

Note the close connection between the concept of net GDP and what we referred to earlier as the need for depletion adjustments. The clear importance of ecological depletion to social well-being argues for similar depreciation adjustments in an ESI.[14] Instead of capital depreciation, an ESI should account for ecological depletion associated with either human activity (e.g., harvest) or external shocks to the quality of ecological assets.

To account for depletion effects on the sustainability of ecosystem flows, we must incorporate the ecosystem Q and its component assets Q_i into the ESI. First, note that an indicator of ecosystem quality should be higher for some state Q^1 than some state Q^0 if and only if over time t

$$(7) \qquad \Sigma_t S^{1t}\delta^t > \Sigma_t S^{0t}\delta^t,$$

where δ is a time discount factor. That is, the indicator should rank the relative quality of the current stocks with respect to their ability to provide future flows of services. For a service index with constant weights p_i over space and time, this is equivalent to

[13] The necessity of being on this path is only one of several limitations to the result. See also Nordhaus and Kokkelenberg (1999) (Appendix A) and Weitzman (2003) for theoretical discussion.

[14] The fact that we net out depletion for man-made assets like factory equipment suggests the principle of doing so for natural assets, like ecosystems, as well. Early attempts to green the national accounts have made just this point and have deducted unsustainable clear-cutting of forests in Indonesia (Repetto et al. 1989) and the deduction of depleted mineral assets in the United States (U.S. BEA 1994). Understandably, these early efforts focus on depreciating natural assets that yield market commodities. The principle applies equally to non-market commodities. See ENRAP (1996) for such an extension.

(8) $\Sigma_t(\Sigma_i q_i^{1t} p_i)\delta^t > \Sigma_t(\Sigma_i q_i^{0t} p_i)\delta^t$,

For a small, one-time change in Q_j at time 0, we can identify the impact on future services as

(9) $K^t = \Sigma_{i,t}((\partial F_i^t(Q)/\partial Q_j^0)dQ_j^0 p_i^t)\delta^t$.

K may represent a stand-alone index of ecosystem health or, analogous to net GDP, it may be subtracted from the index of services enjoyed at time t, yielding a net service measure

(10) $NetESI^t = S^t - K^t$.

The term $\partial F_i^t(Q)/\partial Q_j^0$ is the change in the level of ecosystem service i at time t as a result of the shock to ecological asset j at time 0. The present value of all such effects represents the required adjustment for a "net ESI" that accounts for the effect on future ecosystem services of current shocks to ecosystem assets. Confining ourselves to the effect on a single index of service, the adjustment is

(11) $\Sigma_t((\partial S^t/\partial Q_j^0)dQ_j^0 P^t)\delta^t$,

where the value index is used to adjust for the relative values of the bundle over time. Depending on the context, one may prefer a more physical measure unweighted by preferences and even without discounting.

In the case of marketed services and outputs, assets can be depreciated at their market prices. Unfortunately, there are no such prices associated with ecological assets. Indeed, the problem of missing prices and non-market valuation is particularly significant in this case. Missing prices for ecological services can be thought of as a problem of garden-variety non-market valuation. Missing prices for ecological assets must be derived from the values of these services coupled with the production of these services by the asset. That is, we must also predict changes in services caused by changes in ecosystem health. In any real ecosystem, these latter terms, written so innocuously above in the term $\partial F_i^t(Q)/\partial Q_j$, are bound to be complex. The underlying relationships are likely to be non-linear (or even non-monotonic) functions of the health of all ecosystem assets. Moreover, both this functional relationship and the path of $\{Q_j\}$ over time would need to be known. Within certain models, these relationships can be computed. For example, Brock and Xepapadeas (2003) model the dynamic relationship between genetic diversity and crop survival; Finnoff and Tschirhart (2005) and Smith and Crowder (2005) model interactions between predator/prey relationships and marine animal population adjustments over time. The predictive power of these models, however, is yet to be tested.

Our goal is not to create a standard that cannot be met. Rather, we emphasize these conceptual relationships as a guide for thinking about ecological indicators and index numbers and to set a course for future work. Our message to economists is that information from ecology must be used as a sustainability constraint on feasible service flows.[15] Our message to ecologists is that analysis of ecosystem asset quality is consistent with and central to an economic analysis of ecosystem services and social well-being derived from them. Thus, our goal is to measure both indicators of service flows and indicators of ecosystem health that will signal the future of those flows.

3. Ecology, Economics, and the Units of Account

This section discusses two empirical challenges to the construction of an ecosystem service index. First, what do we mean by q, the units of ecosystem service flows? Second, how are these units distinguished from the ecological assets Q that give rise to service flows?

3.1 What is an Ecosystem Service q*?*

[15] Again, see Weitzman (1976; 2003) on the link between such green accounts and the set of feasibility constraints from capital (here natural capital) in a dynamic optimization framework.

In the previous section we distinguished between indexes of ecosystem services and indexes of their value. In fact, the conceptual distinction between ecosystem services and their value often is surprisingly difficult to make, even for a single service. Because ecosystem services do not emerge from factories and are not sold in markets, defining and measuring their "unit of account" requires innovation on the part of both economists and ecologists.

Consider the following example, in which we first focus on a change in service rather than a measure of absolute levels. An orchard-grower grows apples A using a combination of human labor L and the pollination activity of bees, denoted B, provided by the local ecosystem. Labor is measured in man-hours of a given quality. Pollination in principle could be measured in "grains of pollen transported" or "tree-to-tree trips." More realistically, pollination might be proxied by the number of bees within a certain distance of the orchard. The production function is A=A(L,B). The respective prices (values) of each are P_A, P_L, and P_B, where the first two are observed market prices and P_B is a virtual price.

Production theory provides two perspectives on the role of bees. First,

(12) $P_B = (\partial A/\partial B)P_A$.

The value of bees can be derived from their productivity with respect to apple production, times the value of apples (see e.g., Freeman 2003, Ch. 9).

Repeating this tangency condition for labor and suitably arranging terms, we also have

(13) $P_B = \dfrac{\partial A / \partial B}{\partial A / \partial L} P_L$.

In other words, the value of bees can be derived from the substitutability of bees and labor in apple production and the value of labor.[16] This type of relationship often is used in non-market

[16] Normally, we would say the farmer positions himself on the production function to ensure $(\partial A / \partial B) / (\partial A / \partial L) = P_B / P_L$. In the theory of virtual prices and non-market valuation, the logic is reversed, so that P_B is the value associated with that point on the production function.

valuation studies of "home production" of health and other commodities (see e.g., Freeman 2003, Ch. 10).

For a change in the quantity of bees, d*B*, we have the total value

$$(14)\qquad P_B dB = (\partial A/\partial B) P_A dB = \frac{\partial A / \partial B}{\partial A / \partial L} P_L\, dB.$$

Which part of this expression should be considered the quantity measure of service q_i and which part should be considered the value of that service p_i? The environmental economics literature has been surprisingly vague about this issue. For example, Kopp and Smith (1993, Chs. 2, 7, and 14) variously refer to the change in total value (P_BdB in the notation of our example) the change in the final service ((∂A/∂B)dB), and the change in the environmental quality (dB) as changes in services.

Of these three possible definitions of services, our preference is for the third. The first is inappropriate as it includes value as well as quantity information. The second, (∂A/∂B)dB, does have some advantages. The service here is the contribution of the ecosystem to apple growing. This measure has three advantages: 1) it is defined in terms of the final good consumed by households; 2) it is easily measured in units such as number, pounds, or bushels; and 3) when the output is a final market good, as in this example, it has the readily observable value weight P_A.

However, difficulties arise with this second definition of service when we switch from the measurement of a change in services (the typical task in environmental economics) to the measurement of the total quantity of services (the task of an ESI). First, note that if we simply measure the final output—bushels of apples *A*—as the service level, such a measure would not capture the contribution of the ecosystem to production. For example, it could happen that the number of bees declines one year but the yield of apples increases due to other changes, including changes in labor or capital made by the farmer. This would not be a very good measure of ecosystem services. Instead, the measure requires all other inputs (labor in our example) to be held at some constant level (which we can call $\overline{L}$). Adapting the second this measure to this case we would have as a candidate

$$(15)\qquad q_i = A(\overline{L}, B) - A(\overline{L}, 0).$$

In our orchard example, where bees are non-essential, this measure might be useful. In cases where there is no substitute for natural capital, however, this output measure is identical to final output, since A($\bar{L}$, 0) = 0. (What would be the landings of a fleet of commercial fishing vessels if there were no fish in the sea?) Moreover, construction of the measure would require global knowledge of the production function. An alternative approach that captures the specific contribution of the ecosystem is to define the service at its marginal contribution

(16) $q_i = (\partial \mathrm{A}/\partial \mathrm{B})\mathrm{B}$

This is the linearized contribution of the entire stock of bees and bears an obvious similarity to the term $(\partial \mathrm{A}/\partial B)dB$ above. But this approach is a very superficial way to dodge the problems associated with equation (15), the difference being due only to the error in the linearization.[17]

Another difficulty with this second measure of services lies in the associated measure of value, P_A. While that price measure is readily available when the ecosystem contributes to a tangible good (apples in our case) that is traded in markets, it is not generally observable for cases where it contributes to an intangible service, such as in the household production framework. There is no clear value to recreation, for example, just a value for inputs to recreation (see e.g., Bockstael and McConnell 1983).

For all these reasons, we prefer to use the ecological factor that enters preferences or production as our measure of services, so that in our example

(17) $q_i = \mathrm{B}$.

In general, to identify the service, we suggest the following guiding principle: that the last link in the chain of ecosystem and economic service production that still involves ecological

[17] For linear functions F( ), of course, there would be no difference. And for non-convex production sets, the result could actually be more apples than are produced.

factors be identified as the ecosystem service. Or, to put it in other terms, wherever it is combined with final market goods and services in a model of (home) production, that is the point at which to define the ecosystem service. The value of that service can then be measured by its contribution to market outputs as in equation (12), or by its substitutability for market inputs as in equation (13).

This principle puts the ecosystem on an equal footing with market inputs and outputs by identifying the point at which they come together in production. This is important because it means our definition will allow for the eventual integration of an ESI into a more comprehensive set of national accounts. Our definition of services is entirely consistent with that used in conventional income accounting, so that our ESI could be combined with conventional GDP for a measure of green GDP (see e.g., Peskin 1989 and Hecht 2005). For interdisciplinary models of ecosystems and economic systems, the ecosystem services are the value of “exports” from the former to the latter.

Our definition may at first seem surprising to both economists and ecologists. For one thing, people tend to equate services with benefits. Recreation may be termed a service, or flood protection may be termed a service. These are ecosystem benefits, not ecosystem services. To illustrate, consider a second example: outdoor recreation. A household combines leisure hours, travel costs, a boat, fishing equipment, a lake, and a bass population in that lake to create a recreation experience. Abstracting from issues of water quality and views, the contribution of the ecosystem to the service is the bass population. Accordingly, the bass population is our indicator of the ecosystem service. The first reaction of some may be that the service is best measured by recreational benefits or perhaps fish caught. However, benefits correspond to the $P_B dB$ measure discussed above in the orchard example, which inappropriately combines quantity and value information. Fish caught, like the change in apples harvested, is inappropriate because it includes more than the contribution of the ecosystem; it includes the skill of the angler, the quality of his equipment, and the time he invests.[18]

Are we confusing ecosystem service flows with an asset, in this case the stock of fish? No, for three reasons. First, the goal of the ESI is to measure the contribution of the ecosystem to

[18] A subtler alternative corresponds to the definition of service explored in equation 15. According to that definition—and where there is no substitute for natural capital (i.e., fish)—the service is defined as the difference in the number of fish caught between the current scenario and a counterfactual, holding non-ecological inputs constant. But this leads an unsatisfying measure—total catch—because the counterfactual catch is zero.

outputs valued by people. That contribution is incorporated into the index through the weighting structure, so that services that make a bigger contribution to a final good like apples or angling are given greater weight. Second, note that our measures have a clear parallel in GDP accounting practices, where quantities of intangible services—economic consulting for example—are based on inputs, and, in some cases, entirely on capital inputs. Third, the identical measurement of services with some assets does not obviate the conceptual distinction between them. A change in the stock of bees or fish at time t shows up immediately in an ESI through the q terms and in the net ESI through reductions in the expectation of future services.[19] Moreover, additional ecosystem assets remain in other parts of the ecological production function, such as the habitats necessary to maintain bee and fish stocks.

There is an additional sense in which a given ecological component can be both an asset and a service. In the fishing example presented above, the chemical quality of the lake was not a service but rather an ecological asset supporting services such as water clarity and the bass population. However, chemical water quality is a service if the lake is used for drinking water. If the lake provides—or could potentially provide—drinking water, the chemical properties of the water are directly relevant to a consumption decision. Should a household boil their water, rely on municipal treatment, or choose to drill a well? These decisions depend directly on the chemical composition of the lake.[20] As we alter the analysis of the lake's benefits (from recreation to drinking water provision), an observable ecological element can change from an asset to service and vice versa.

3.2 Terminological Confusion: Ecosystem Services, Functions, and Assets

Given our definition, some ecologists might wonder why more species are not counted as services. For example, why aren't non-recreational fish species an ecosystem service? After all, recreational species depend on other species for their survival. Going further, why isn't the lake's

[19] Recall the bald eagle example where people valued only the existence of eagles as an increasing function of the stock. As a measure of existence-services the change in the number of eagles has an immediate effect on the current value of the ESI, and, as an asset, through population dynamics, on future services as well. We are in effect saying that many services can be best modeled in this way.

[20] Assets supporting the lake's chemical characteristics (now the service) include surrounding land uses.

ability to cycle nutrients a service? These things are important and valuable in our accounting framework, but they are not ecosystem services because they are not consumed as inputs in a (household) production function. To make the definition clear, we distinguish services from what we call ecosystem functions and assets.

- Ecosystem functions are the biological, chemical, and physical interactions associated with ecosystems. These functions are the things described by biology, atmospheric science, hydrology, and so on. Functions are socially valuable, but are not services.

- Ecosystem assets are intermediate physical components of nature. Assets are intermediate in that they are necessary to the production of services but are not services themselves. Assets are the inputs to an ecological production function that yields an ecosystem service.

In the fishing example, the lake's bass population—a measure of both an asset and a service—depends on other environmental assets. For example, the lake's chemical and biological water quality is an input to the production of bass and is therefore an asset associated with the bass population service.[21] These qualities of the lake are important and valuable but are not services in an economic accounting sense. Or again consider the bee example. As discussed above, the number of bees in an area is an ecosystem service to a nearby orchard. If edge habitat supports the bee population, it is valuable but not a service to the orchard. Its value to the orchard is as an asset supporting the bee population and is accounted for in measures of asset depletion and lost future bee populations. Measurement of ecosystem assets is important to ecosystem accounting, even though assets aren't services.

To further clarify these distinctions, consider Gretchen Daily's oft-quoted list of representative ecosystem services in following list:

[21] Note that economists will usually refer to recreation as the service because recreation is the end product sought by the household.

- •purification of air and water
- •mitigation of droughts and floods
- •generation and preservation of soils and renewal of their fertility
- •detoxification and decomposition of wastes
- •pollination of crops and natural vegetation
- •dispersal of seeds
- •cycling and movement of nutrients
- •control of the vast majority of potential agricultural pests
- •maintenance of biodiversity
- •protection of coastal shores from erosion by waves
- •protection from the sun's harmful ultraviolet rays
- •partial stabilization of climate
- •moderation of weather extremes and their impacts
- •provision of aesthetic beauty and intellectual stimulation that lift the human spirit (Gretchen Daily, *Nature's Services*, Island Press, 1997).

Many of these "services" are what we would call functions. Others are descriptive of assets, being inputs to services rather than services themselves. For example, is water purification an ecosystem service? We would argue no. Rather, purification is a function of certain land cover types that helps to produce clean water. In our terminology, purification is embodied in the production function of the service but is not the service itself. Rather, clean water is the service and is valued for its connections to health, recreation, and so forth. Similarly, the preservation and renewal of soils and the cycling of nutrients are processes. These processes yield, via a production function, soil characteristics that are services (e.g., a soils' nitrogen content). Or consider the detoxification of wastes. Detoxification is a process embodied in a set of production functions. These functions yield particular air, soil, and water characteristics. Air or water of a particular quality are the services because they can be combined to produce consumption that is valuable to the household.

Several of Daily's items are benefits, not services. Consider flood control. We term flood control a benefit, not a service. Rather, we call services components of the natural landscape (e.g., wetlands) that prevent flooding. Wetlands, after all, are an input, along with dikes and other man-made inputs, into the production of property protection. Similarly, "aesthetic beauty and intellectual stimulation that lift the human spirit" is a benefit of certain kinds of natural landscapes. The services used to create this benefit are more specific components of the landscape, such as undeveloped mountain terrain, unbroken vistas, or a large conifer forest.

3.3 The Challenge of a Comprehensive Services Account

Our ultimate goal is the characterization of a comprehensive list of ecosystem services, much like the national accounts seek a comprehensive list of economic outputs. The way to conduct this exercise is suggested by the examples we have discussed already. First, understand the ways in which nature contributes to human well-being as a substitute for built capital and via the provision of recreation, health, aesthetic, and intangible benefits. Second, think systematically about how natural capital is combined with conventional capital, human capital, and other inputs to produce those benefits. Third, determine the end-products of nature that are most closely related to household decision making.

Time and Location Dependence

Several prominent challenges are associated with this endeavor. The first is the need to make ecosystem service measures time- and location-dependent. The value of a car is not closely related to whether it is located in California or New Jersey. This is not the case with ecological services or the ecological assets on which they depend.[22] The value of ecosystem services is highly dependent upon their location in the landscape, the scale over which services are provided, and the time at which they are provided.

The benefits of damage mitigation, aesthetic enjoyment, and recreational and health improvements depend on where—and when—ecosystem services arise relative to complementary inputs and substitutes. Also, the ecological asset interactions that enhance or degrade service flows are highly landscape-dependent, an issue we turn to in more detail in Section 4. Accordingly, it is necessary to spatially define "service areas" and temporally define "service windows." An unfortunate reality is that these will be different for every identified ecosystem service. Boundaries are needed to define the likely users of a service, areas in which access to a service is possible, and the area over which services might be scarce or have substitutes. Nevertheless, this issue is well known in environmental economics and not confined to index-based evaluation tools (Smith and Kopp 1996). For example, a key methodological issue in any econometric recreational benefits study is the determination of the appropriate choice set facing recreators.

Quality Adjustment and the Definition of q

The practical imperfections of the national income accounts are legion. A prominent example is the treatment of changing product qualities over time. GDP counts computers, but because of technological innovation a computer in 1990 is clearly not the same as one in 2005. Unfortunately, adjustment for quality differences due to innovation or even some basic product characteristics creates measurement headaches. Currently, the national accounts rely on the selective use of hedonic adjustments in certain product categories. But imperfect quality adjustments are recognized as an important limitation of the accounts. Measurement of

[22] Location-, scale-, and time-specificity are core characteristics of modern ecology (O'Neill et al. 1999). For example, the quality of a habitat asset can be highly dependent on the quality and spatial configuration of surrounding land uses. The ability of areas to serve as migratory pathways and forage areas typically depends on landscape conditions over an area larger than habitats relied upon directly by the migratory species. The contiguity of natural landcover patches has been shown for many species to be an indicator of habitat quality and potential species resilience (Olson 1996). Hydrological analysis is yet another field that has long recognized the importance of relationships between landscape features (Boyle et al. 1998).

ecosystem services will suffer from the problem as well. Ecosystem services clearly differ in their quality. Unfortunately, measurement of quality differences based on easily observed data is even more difficult than in the conventional economy.

Consider one particular kind of benefit provided by nature: visual enjoyment. An ideal, but impractical, service measure is "units of enjoyment associated with visual appreciation of the surrounding environment." An index of services must instead choose a measurable proxy for this ideal measure of service. For example, we can define the service unit q as an "acre of natural land cover." Note immediately that a finer definition of q could be employed, such as an "acre of forested land cover," or an "acre of tulip cultivation." These are more precise in their depiction of visual quality associated with particular land cover types. Note also that there are alternatives to acres of land cover as units yielding visual well-being. An obvious alternative is "acres of visible natural land cover." Again, this speaks to the quality of the service.

For any type of ecosystem benefit the accounts will have to balance between the ideal and the most practical service indicator. Consider the benefit of flood damage avoidance. "Natural land cover" is the generic and easiest to measure ecosystem services providing this benefit. But more specific qualities of that land cover are relevant, including its location in the floodplain, the hydrology of the site, its vegetation and soil type. A reasonable compromise is to distinguish between "acres of wetland in the watershed" and "non-wetland natural areas." Wetlands already are measured as distinct land cover types and are associated with high-quality flood pulse mitigation qualities. In other words, it is relatively easy to measure wetlands, and their quality is different enough to make it worthwhile.

The weights given to services can help adjust for quality differences, since quality differences can be observed via prices or constructed indicators of willingness to pay (WTP). In fact, our strategy for ecosystem services is to build quality issues into the WTP weights via indicators. Nevertheless, note that when it comes to service quality, what is included in q versus what is included in p is a choice rather than a clear standard.[23]

[23] Again, this issue is mirrored in the national accounts. We count tires sold, rather than the vehicle miles over which the tires last. Prices at the time of purchase will reflect consumers' understanding of the quality difference, but the ideal output measure q embodies that quality difference.

Public Goods and Adjustment for Population Differences

Another important issue for quality adjustment relates to the treatment of public goods. Many, if not most, ecosystem services are public goods. If an ecosystem benefit is enjoyed by many, rather than a few, should we say that there are more services being provided? Clearly there are more benefits, but as we have noted often, benefits are not the same thing as an accounting measure of services.

Consider the role of population in measures of services and values. Continuing the notation of Section 3.1 for consistency, if B is a pure public good provided over some area x, the total value would be $B{*}P_B{*}POP_x$, where P_B is the average per capita value and POP_x is the population within the appropriate region x. This is the Samuelson condition for public goods. Now, taking up the question of Section 3.1, is population a part of the service or of its value? From one perspective, it seems natural to say that we have a given level of service, of which value is the sum of individual WTP. POP_x would then be part of the value. However, viewed another way, the service flows to more people, each of which has the value P_B. After all, a factory producing enough automobiles to be enjoyed by 1,000 people yields more services than one producing enough for 500 people. Likewise, doesn't a wilderness area producing ducks to be enjoyed by 1,000 people yield more services than one producing ducks to be enjoyed by 500 people? Do there really need to be more ducks to say there is more service?

Whether population goes in the *q* or the *p* side of the equation may depend on the specific context. In most cases, we probably would not want a simple increase in population to register as an increase in ecological services, interpreted as an indicator of the state of ecosystems. Consequently, we believe population usually should be on the value side of the coin but emphasize that these decisions are matter of judgment and interpretation, not economic theory, which specifies only a total condition.

4. Ecology and the Measurement of Ecosystem Services

In this section, we reflect as economists on the biophysical sciences, which we refer to generically as ecology. What is the role of ecology in the development and measurement of an ESI? Does theoretical and empirical ecology give us what is necessary to construct an ESI? To address these questions, we refer back to the basic architecture of an ESI. Current services are measured via the *q*'s and weighted by measures of WTP. In addition, we also argued that the value of current services in an ESI should be adjusted to account for the degradation or enhancement of the stocks necessary to the production of future services.[24] Theoretical and empirical ecology can contribute by suggesting good candidates for the q's by measuring them and by offering predictive relationships between changes in ecosystem assets and future services (i.e., the $\partial F/\partial Q$ terms in equation (9).

Functional relationships that describe the way that ecological stocks (e.g., marine water quality) affect future service flows (e.g., fish harvests) are clearly the province of the ecological and physical sciences. Accordingly, economists seek from ecology this kind of predictive capability. Not all ecology is predictive, but for our purposes predictive ecology is of primary importance.

4.1 Lost in Translation

Compared to neoclassical economics, which consists of a fairly rigid underlying structure, ecology encompasses a multiplicity of paradigms.[25] Consider the following types of ecology: physiological, which explores how individuals interact with their environment; ecosystems, which covers the dynamics of energy transformation and material transfers among organisms and the physical environment; community, which encompasses how biotic interactions such as predation and competition influence abundance and distribution of species; evolutionary, which depicts adaptive processes; landscape, which explores the effect of scale and habitat heterogeneity on ecological processes; and eco-toxicology, which depicts the fate and consequences of man-made substances in the natural world. The sheer variety of biophysical perspectives can thwart interactions between economics and ecology.[26] A recent National

[24] Recall our definition of ecological assets from Section 3.2: ecological components necessary to production of ecological services.

[25] It is dangerous for non-ecologists to depict the science of ecology. But mutual interpretation is necessary if the two disciplines are to interact more effectively. We welcome corrections to our interpretations of modern ecological science.

[26] For discussion of the lack of consensus and unifying structure in ecology, see Suter (1993) and O'Neill (2002).

Academy of Sciences' *Report on Ecological Valuation*, authored in part by ecologists, described the linkage between ecological structure and function and service valuation as "suffering somewhat from indistinct terminology, highly variable perspectives, and considerable divergent convictions" (National Research Council 2004, p. 51).

Another important source of difficulty is that economists seek an understanding of anthropocentric ecological value, while ecology is more concerned with the status, functions, and quality of ecosystems alone, rather than as producers of human benefit. An example of the difference between ecologically relevant endpoints and outputs amenable to valuation is habitat fragmentation (fragmentation's undesirability for most rare, threatened, or endangered species is a point of ecological consensus). Fragmentation clearly is important to the provision of ecological services but not in a way that can be valued easily in terms of effect on the consumption of recreational, existence, or other benefits.

Within ecology, particularly empirical ecology, ecosystems usually are assessed and managed with a goal toward one of two prevalent normative metaphors: health or integrity (Costanza et al. 1992). The concept of ecosystem health is flexible, with numerous competing health concepts debated (e.g., health as homeostasis, health as absence of disease, health as resilience). One definition of health is "the system's ability to maintain its structure (organization) and function (vigor) over time in the face of external stress (resilience)" (Costanza 1992). The definition of ecosystem integrity shares many of these same features. An oft-quoted definition of integrity is that it refers to the ability of an ecosystem:

> to support and maintain a balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats within a region … A system possessing integrity can withstand and recover from, most perturbations imposed by natural environmental processes, as well as many major disruptions caused by man. (Karr and Dudley 1981 p. 55)

Elements of structure, functions, and resilience can be seen in both definitions.

The seeds of definitional confusion can be seen in these alternative metaphors. A first observation is that the definitions are both holistic and complex, as is expected given their subject. But a corresponding lack of precision complicates empirical measurement within ecology and outside of it (Andreasen et al. 2001). Empirically, does a healthy ecosystem correspond to a natural, pre-development state? After all, ecosystems are characterized by constant change. So too, then, should the baselines by which health is measured. This is one reason that the normative definitions feature process- or function-oriented endpoints, such as resilience. Unfortunately, resilience itself then must be defined empirically. A statement such as "integrity suggests wholeness, completeness, and intactness" (Andreasen et al. 2001 p. 22) triggers the search for measures of completeness. Similarly, dissatisfaction with the health and integrity metaphors leads to alternative suggestions, such as the desirability of ecological sustainability or quality (Suter 1993). However, the empirical implications of these alternative metaphors are not clear.

What is noteworthy in all these definitions is their lack of emphasis on social benefits and what we refer to as services. This is not surprising given that the analysis of human well-being is the focus of economics, not ecology.[27] We should not expect ecology to give us measures of what is economically valuable. However, we can hope that ecology will provide understanding of the asset interdependencies that give rise to service flows. How do wetland losses translate into greater flood frequency? How does air quality degradation lead to lake acidification, and how does that acidification affect fish populations? How does patch size translate into population viability? These are relationships that are fundamental to knowledge of future service flows and are—at least in principle—empirically estimable, though the complexity of natural systems will always limit the predictability of such relationships.[28]

This raises two questions: How is the quantity and quality of ecological assets currently measured, and how predictive are state-of-the-art ecological models? Because of the complexity of ecological systems, natural variability, and the large spatial and time scales at play, we are limited in our ability to do this kind of normative ecology. If we are to capture the effect of stock losses and gains on future ecological benefits, we must be able to effectively measure ecological stock quality and predict the impact of stock quality changes on service flows. To address the first question, we describe existing ecological indicators systems. To answer the second, we review in detail two marine ecosystem models.

[27] See Shields et al. (2002) ("We are developing indicators that are meaningful to scientists, but not necessarily to policy makers and the public.").

[28] There are limits to how far we should push an ESI's ability to capture such asset interactions. First, the biophysical sciences themselves have yet to characterize fully these relationships empirically. That is a process likely to take decades, if not longer. Further, many cause and effect relationships may be largely unquantifiable given the complexity of interactions among ecosystem components (Costanza et al. 1997).

4.2 Ecological Indices

Ecological indices, like economic ones, are composed of individual indicators. Many individual indicators, such as those for air and water quality, are self-explanatory and are more or less directly related to the quality of an asset. Dissolved oxygen levels in water are an example. Here, we focus on indices designed to capture more complex ecological characteristics.

What is striking, but not surprising, is the multiplicity of indicators advocated by ecology. While numerous indices exist, few if any have achieved wide acceptance within ecology. Certainly there is no consensus within ecology on which indicators or indices are most important. This is due in part to the complexity and range of ecosystem characteristics. It is also a reflection of unresolved questions in ecological science.[29] A comprehensive depiction of existing indicators is beyond the scope of this paper. However, because of our interest in services and the measurement of well-being, we focus on indices envisioned as relevant to public decision-making: biodiversity indices, biotic integrity indices, the hydrogeomorphic wetland assessment method, and habitat suitability indices.

Biodiversity

Perhaps the most common ecological indices are diversity indices; biodiversity being one of the most studied subjects in ecology.[30] Classic diversity indices include Simpson's D and E and Shannon's H and E indexes that measure both the number of observed species and the evenness of species' distribution. It is common also for land and wildlife conservancies to advocate and deploy diversity indices. Such indices are used to identify biodiversity hotspots and are

[29] For example, the use of particular species as indicators of ecosystem quality, while still common, has been discredited within ecology (Andreasen et al.2001, p. 25).

[30] Chapin et al. (2000) and Naeem et al. (1999) provide comprehensive overviews of the state of knowledge regarding biodiversity and its relationship to ecosystem function.

frequently weighted according to the rarity of species.[31] Diversity indicators range from simple ones based on vegetation (e.g., Defenders of Wildlife 1988), to complex ones based on genetic diversity and polymorphism (Noss, 1990). Economists also have contributed to the literature on biodiversity. In particular, Weitzman (1992; 1993) has suggested a measure based on pairwise comparisons of the genetic distance between species or individuals. The measure also can be interpreted as the most likely taxonomic tree evolving from a common ancestor. Weitzman (1998) has extended this idea to policy rules for saving the greatest number of genes.

Biodiversity is a good example of how the interpretation of an ecological indicator as either an ecosystem service or an ecosystem asset can depend on context. If biodiversity has non-use value in itself, such measures may be a good measure of services. If biodiversity has value insofar as say, new pharmaceuticals are more likely to discovered if diversity is higher, diversity again may be considered a service input into pharmaceutical production, with a value equal to the marginal physical product times the value of drugs (Simpson et al. 1996). Finally, if biodiversity within a species is valued only insofar as it maximizes the likelihood of the survival of a species that is valued as a whole, it is not a measure of services at all. It may, however, still be a good measure of the future stream of services. In any of these cases, however, the ideal measure of diversity would weight the importance of various dimensions. None of the aforementioned indices account, however, for the fact that a more diverse population in terms of the presence of a useless internal appendage, for example, is of less consequence for any of these purposes than a population that is diverse in color, size, and eating habits.

Biotic Integrity

The most commonly employed indicator system in government decision-making is the Index of Biotic Integrity (IBI). Many states and federal agencies use the IBI to assess aquatic resources, and the method has spawned a variety of derivative approaches geared toward terrestrial systems (Canterbury et al. 2000). The system is composed of 12 indicators, six relating to the "composition and richness of species" and six relating to "ecological factors" (Karr 1991). The former include counts of particular species. The latter include measures of sampled fish health,

[31] The Nature Conservancy, for example, uses a rarity-weighted richness index to identify target conservation areas (Stein, Kutner, and Adams 2000).

including evidence of disease, damage, and anomalies. They also include so-called trophic factors, such as the prevalence of omnivores and top carnivores that help depict the flow of food and other energy through the system.

Biotic integrity measures are not well-suited to be service measures. Again, they are best thought of as indicators of asset quality. For the purpose of valuation, the trophic indicators are of scientific importance but are beyond the intuition of the public. Counts of individuals could be used as location-specific population indicators if they were related to species valued for recreation or their existence value, though the species counted in a typical IBI are neither. Similarly, a value potentially can be placed on deformity or disease in individuals, particularly if a connection to deformity in focal species can be made. The overall conclusion, however, is that the linkage between an IBI and outputs on which a value can be placed is weak. The point of an IBI, however, is to assess generally the quality of aquatic habitat. A good IBI score is not only good for the measured species but is presumed to be an indicator of the habitat's ability to support a larger community of species interactions.

Hydrogeomorphic Assessment

Another index system is the hydrogeomorphic (HGM) approach developed to support regulatory decision-making regarding wetlands.[32] HGM has several characteristics worthy of note. Like the IBI, HGM makes a comparison between the study site and a reference site used to represent the baseline ecological condition (i.e., baseline asset quality). Second, position in the landscape is important. Third, it focuses on depicting site capacity to perform functions, including nutrient cycling, surface and groundwater storage, floodplain connectivity, organic carbon generation, retention of organic and inorganic particles, and provision of vertebrate habitat, plant communities, and aquatic food webs. Indicators thought to affect the performance of the eight basic functions are measured at each site. Like the IBI, the function measures generated by the HGM method are measures of asset quality not flows of services.

[32] See Brinson (1993) and Brinson and Rheinhardt (1996).

Habitat Suitability

Another species-oriented index is the Fish and Wildlife Service's Habitat Suitability Index (HSI). The indices are meant to characterize the carrying capacity of a habitat for one particular species. On a species-by-species basis, HSI relates habitat quality to observable biophysical characteristics derived from the scientific literature. For example, tree density or canopy cover may be related to a habitat's species-specific carrying capacity. For aquatic species, indicators of things like dissolved oxygen, salinity, water depth, substrate type, and toxicity would be combined in an HSI score.

HSI scores could be used as proxies for intangible or recreational benefits but only for species that are perceived as important endpoints in themselves, such as endangered species. However, they are again measures of ecological asset quality not ecological service flows.

Other Ecological Indices

A range of other indices is worth noting in brief. The Environmental Protection Agency's Environmental Indicators Initiative, for example, is designed to track biophysical conditions over time. Indicators include air pollutant emissions, water quality measures, beach closings, and the prevalence of fish consumption advisories. Note that the goal of the indicators is not economic measurement but "scientific measurements that track environmental conditions over time."[33]

The U.S. Department of Agriculture uses indicators in a variety of programs. The Forest Service, for example, reports a set of forest health indicators, including measures of crown condition, ozone injury, damage from disease, mortality, lichen communities, woody debris, vegetation diversity, and soil condition (Fish and Wildlife Service 1996). The Conservation Reserve Program has used an Environmental Benefits Index (EBI) to target enrollments in that program. The EBI uses a variety of indicators to derive scores for six endpoints: wildlife support, water quality, erosion, air quality, and conservation benefits. Example indicators include

[33] More information about the Environmental Indicators Initiative is available at: www.epa.gov/indicators/abouteii.htm.

proximity to wetlands and conservation areas, soil erodibility measures, and distance-weighted population exposed to soil-related dust (Ribaudo et al. 2001).

In addition to operational indicator systems, several national-level indices are worthy of note. A recent National Academies report recommended the generation of indicators related to species diversity; land cover and land use, nutrient runoff, soil organic matter, ecosystems' capacity to capture energy,[34] amount of energy and carbon that has been captured, carbon storage, stream oxygen levels; and the trophic status of lakes (Ecological Indicators 2000). A heavily publicized report from the Heinz Center (2002), used primarily to demonstrate significant gaps in data, called for the collection of numerous indicators relating to land use, aquatic resources, forests, agriculture, and other ecological conditions. Finally, a multi-agency report geared toward coastal issues recommended collection of seven basic water quality indicators: measures of clarity, dissolved oxygen, coastal wetland loss, benthic conditions, contamination of fish tissues, eutrophic condition, and sediment contamination (U.S. EPA 2001).

Several conclusions should be drawn from this brief survey. First, while common elements do emerge, there is no consensus scientific or decision-oriented indicator system that should be the focus of a valuation-oriented ESI.[35] Second, ecological indices are not geared toward consumption-related endpoints. Ecological indices are thought of as a way to assess existing conditions, provide early warning of changes, and diagnose causes of environmental degradation (Dale and Beyeler 2001). In this respect—rather than as measures of service flows—that they are suited to application in an ESI.

4.3 Predictive Ecology: the Neuse Models

Economic analysis seeks tractable ecological production and damage functions to evaluate policy options and—in our case—to track economic welfare arising from ecological services. The importance of production functions, of course, is that they enable prediction of biophysical and economic outcomes. Unfortunately, ecology does not yield easily functions with sufficient

[34] For example, chlorophyll per unit area is an indicator of the capacity to capture solar energy.

[35] "No well-tested and direct index of biotic or ecological integrity exists for terrestrial ecosystems or for entire landscapes comprising terrestrial and aquatic habitats" (Andreasen et al. 2001, p. 22).

predictive power to be of direct use to economic assessment. In fact, contemporary ecology can be viewed as demonstrating that simple functional relationships do not exist—at least at a scale broad enough to be useful to economists.[36] Many functions are explored and depicted, but this tends to be the province of theoretical ecology rather than empirical ecology at a landscape level.

Increasingly, however, the problem of prediction is being addressed.[37] We consider in detail two innovative ecological production models, constructed by economists and ecologists, associated with a particular landscape context: the Neuse River watershed and estuary (Finnoff and Tschirhart 2005; Smith and Crowder 2005). The models are of particular interest for two reasons. First, they address a predictive policy question, namely, the impact of reduced nutrient loadings on ecological outputs. Second, these predictions arise from a model in which a variety of ecological interactions take place.

In the language of this paper, the models feature asset–asset interactions and use these to predict ecological service flows. As we argued above, the linkage of sophisticated ecological analysis to economic outcomes is rare in ecology.[38] The models are ideally suited to an illustration of what we mean by depletion and depreciation adjustments. Because they are predictive—and innovative—the models can be used to show the challenges associated with the calculation of depletion and depreciation adjustments in an ESI.

They are also a way to convey the differences and similarities between an ecological services index and bioeconomic modeling. To do this, we will describe the models in the language of an ESI. Both papers explore the effect of reduced nitrogen loads on an estuarine-marine environment. They do this by tracing nitrogen load changes through a system of ecological interactions. In Smith and Crowder (2005), nitrogen loads affect algal stocks that, along with assumptions about sediment oxygen demand, allow the estimation of dissolved oxygen (DO) levels. DO then interacts with species and habitat variables—including growth rates, mortality, and the habitat's carrying capacity—to produce estimates of crab and clam

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[36] This may be in part because after initially borrowing from nineteenth century mechanical metaphors in a spirit similar to neoclassical economics, some ecology has turned to simulation methods that are not based on maximization or equilibrium concepts (e.g., Mirowski 2002).

[37] See also Paul Murtaugh, "The Statistical Evaluation of Ecological Indicators," *Ecological Applications*, 6:1, 132–139, 1996.

[38] This is not meant as a criticism of ecology. Note that sophisticated ecological modeling is rare in economics.

populations. The most ecologically distinctive element of the model is that these interactions are location-specific and location effects are linked via the migration of predators between patches.

Smith and Crowder have a predictive model of spatially explicit, interacting ecological assets. Measurable indicators of these asset qualities include nitrogen concentrations, algal stocks, and DO. They also yield spatially distinct prediction of species abundance. Their model translates these, with the addition of harvest effort, into blue crab harvests. The ecosystem service may be alternatively defined as the harvest itself or as the value of blue crab stocks as a productive input into those harvests.

In Finnoff and Tschirhart (2005), a nitrogen load reduction leads to changes in algal populations of dinoflagellates and diatoms. Spatially explicit ecological interactions are also a feature of their model. The dinoflagellate and diatom populations are spatially distinct, being differentiated by water depth. These populations then interact with spatially distinct zooplankton, jellyfish, and clam communities. The populations of interest as outputs are two commercial species, croaker and blue crab. The spatially distinct populations are predicted based on several predator–prey interactions, including between croakers and blue crabs themselves. As in Smith and Crowder (2005), we are given a spatially explicit model with multiple ecological asset interactions that generates service flow estimates q. The model allows for services related to two spatially distinct blue crab harvests, as in Smith and Crowder (2005), and potentially croaker harvests as well. Finnoff and Tschirhart (2005) also raise the issue of accounting for croaker-related services even when these are exported outside of the geographic domain of the model. With the wider range of species included, one could imagine extending their model to clam harvests and perhaps to (negative) non-use values for nuisance species like jellyfish as well.

How would we use the elements of the Neuse models in calculation of an ESI for the Neuse watershed? The ecological variable contributing to the production of crab harvests is simply the stock of crabs. Assuming this stock is measurable in principle via some sort of census, measuring the service raises no difficulties. A more challenging task is identifying the appropriate measure of the decline in the ecosystem assets arising from nitrogen deposition and depletion of the fishery and the adjustment of other species to those shocks, insofar as they affect future service flows.

To illustrate such a measure, we use model output from Finnoff and Tschirhart (2005).[39] To first illustrate the construction of an index, we aggregate the crab services with croaker services. Although croaker is not actually harvested in the model, as the authors point out, this would be a natural extension. Our two measures of services s_i are thus blue crab and croaker populations, respectively. We construct an index of ecological services S^t for each of the 200 model-years by weighting each population in year t by a constant price. Following equation (13), the value weights are the marginal contribution of the stock, evaluated at the model parameters and at benchmark labor input (as measured by season length), times the wholesale price of blue crab and croaker, respectively.[40] Next, for the first 100 years of the model, we compute the present value of the next 200 years of ecosystem services, discounting at 3%.[41] Finally, we compute the change in this service flow, which is the depletion adjustment we require.

The role of ecological indicators must be to compute this future flow of services. We test the ability of three common types of indicators to predict these changes in future services. The first indicator is the Simpson index of biodiversity, computed from the populations of all other creatures in the model. The second is a measure of water quality, specifically deep-water DO, which has been used in the HSI. In the model, DO can be proxied with the populations of dinoflagellates (algae). The third is the populations of species on which crab and croaker prey, namely deep-water zooplankton and clams.

For each indicator, we regress the modeled change in the flow of future services on each of these indicators, for each case using four models consisting of 1-year lags, 1- to 3-year lags, 1- to 5-year lags, and 1- to 5-year lags plus 3rd order autoregressive terms. (Higher order autoregressive terms generally could be rejected from the models.) We also try an auto-regressive only model and a combined model of DO and prey. To illustrate the fit of these models, Table 1 reports the R^2 in each case as applied to the "business as usual" scenario and likewise Table 2 for the "30% nitrogen reduction" scenario.[42]

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[39] We thank the authors for sharing these data and for helping us to implement these illustrations.

[40] See equation 3.1 in Finnoff and Tschirhart (2005). The value of d_B is 0.00178; the value of α_B is 0.75; the value of β_B is 0.25; and the benchmark value of T is 48. The wholesale prices of blue crab and croaker are \$0.72 and \$0.67, respectively.

[41] With 200 years of model output, actual data only is available to compute this present value for the first year. Subsequent computations are made assuming the service flow for years 200 to 300 remain constant at year 200 levels.

[42] Clearly, future work along these lines must take account of the time dependency in the error structure. Durban-Watson statistics do indicate the presence of first-order autocorrelation.

In the business as usual scenario, all the models with autoregressive terms, including the autoregressive-only model, predict well. In this scenario, the simple correlation between adjacent changes is 0.82 and is high for other pairs as well. In contrast, in the 30% nitrogen reduction scenario, the autoregressive-only model does not perform as well for the case of the first order lag. The simple correlation between adjacent changes is only 0.12. However, the correlation is as high as 0.41 for lags over two periods, explaining the increased fit for the models with higher autoregressive terms, approaching the business as usual scenario.

Since actual future services are not observed, however, such models could not be used in any practical indicator of ecosystem condition. Instead, we must rely on measurable indicators, such as DO and species populations. Not surprisingly, given that biodiversity per se plays no role in the Finnoff-Tschirhart model, the Simpson index generally does not predict future service flows well, especially in the business as usual scenario. DO, which directly affects fish respiration, performs somewhat better, with R^2 values of 0.64 and 0.60 using 3rd order lags in the business as usual and 30% nitrogen reduction scenarios, respectively. However, in both scenarios the model using the population of blue crab and croaker prey—zooplankton and clams—performs best at each ordering of lags, with R^2 values as high as high as 0.92 and 0.95. Adding DO to these indicators improves the model further. Figure 1 illustrates the fit for the 5th-order model with zooplankton, clams, and DO.

A cynic might suggest that we have merely recovered the model used to generate the results. However, we believe that the model illustrates that a small set of static, theoretically measurable indicators can capture the dynamic dependency of future service flows, even using a reduced form relationship in a context where “reality” (here, the Finnoff-Tschirhart model) is structural and more complex. Even the single indicator DO, the easiest to measure indicator of these examples, can explain 60% to 65% of the variation in changes in future service flows. Models using zooplankton and clams, probably the next easiest to measure if uniformly distributed spatially, provide an excellent fit.

Nevertheless, we also take this exercise as a cautionary tale of the difficulty associated with calibrated, equilibrium bioeconomic modeling. The time, resources, and expertise necessary to develop these models yield a fairly limited output from a public decision-making standpoint. Two limitations are worth noting. First, the full range of service impacts arising from a nitrogen load reduction will be much more expansive than impacts on the harvests of two commercial species (a point the authors take pains to acknowledge). Second, the models did not need to confront the challenge of WTP estimation, since they were valuing ecosystem services with available market prices.

5. Indices of Willingness to Pay

In this section, we describe in more detail the dependence of benefits arising from services on the location, scale, quality, and timing of services. Benefits are the basis for the weights p_i assigned to particular services q_i in the ESI. In a consumption index of market goods, market prices are used as a proxy for WTP.[43] Absent market prices for ecosystem services, where are we to find reasonable proxies for WTP? One approach, which is highly arbitrary but appealing in its simplicity, is simply to use equal weights, for example $w_i=1 \; \forall \; i$. In ecology, uniform weights are common (e.g., the IBI). One prominent non-ecological example is the United Nations' Human Development Index (e.g., Anand and Sen 1994), which weights equally three indicators of welfare: life expectancy, adult literacy, and GDP. Equal weighting can be defensible for two reasons. First, the choice of indicators employed in an index may exclude factors thought to have less weight than those included. Second, in the absence of more detailed, empirically demonstrable ecological relationships, equal weights are as good an assumption as any.

5.1 Willingness to Pay-Based Weights

However, our goal is a broad service index and that captures an essential feature of ecological services: Their quality and landscape location strongly influence the social benefits generated. Accordingly, willingness to pay-based weights should be the aspiration of an ESI. First, WTP is inherently context-dependent. In particular, for those times and places where people value one service particularly highly, because of tastes, or a scarcity of that service, or scarcity of substitutes, the service in question would be given greater weight in the index. Second, with WTP weights, the value of the ecological services can be compared with the value of market-based substitutes, which are valued in monetary terms. Just as monetizing benefits facilitates

[43] Market prices reflect society's marginal, not total, WTP. Measures of total WTP, while desirable, are impractical because they require the calculation of consumer surplus (see discussion in Section 1.2). Marginal WTP can be thought of as the virtual price that would induce consumers to consume the exact quantity of ecological services, if they were free to choose such quantities, as they in fact do receive from nature (e.g., Neary and Roberts 1980).

comparisons to economic costs in benefit-cost analysis, weighting services by WTP facilitates comparisons of ecological service flows to other service flows.

Of course, WTP for ecological services raises several issues that are only marginally present in market-based accounts. Fundamentally, prices for ecological services are unobserved. Virtual prices would have to be estimated using non-market valuation methods (see Freeman 2003 for an overview). Ideally, such methods would be applied to all the major services within the spatial scale of any particular ESI. Yet such an exercise would still raise a number of new implementation issues. For example, a spatial sample of virtual prices would have to be constructed differently than a sample of market prices as used in existing accounts. While market prices can be assumed to be largely constant within a single market, there is no arbitrage to ensure this condition for the implicit prices of environmental resources. Also, many ecological services are best thought of as differentiated goods with important place-based quality differences. As noted earlier, the biophysical characteristics of ecosystems are highly landscape-dependent.[44] The same is true of ecological services' social benefits.[45] Accordingly, WTP for ecological services is best represented by a hedonic price function, not a single price.

As a general concern, we highlight one additional issue: The potential instability of preferences for ecological services over time versus an index's need for constant relative preferences across services.[46] Markets implicitly inform the development of preferences by facilitating exchange and educating consumers. Lacking the information provision and exchange properties of marketed goods, the average consumer of ecological services should not be expected to know as much about the factors that give rise to them or to their relative value. As a consequence, preferences for ecological services are likely to be less stable than preferences for conventional goods. This is a reason to pursue independent development of a value index (as described in Section 2) as a complement to a services index.

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44 See Bockstael and Irwin (2000). Also, Noss (1990), Gardner et al. (1993), Gustafson (1998), Richards et al. (1996).

45 See Heal et al. (2001) and Bockstael (1996).

46 About this problem, Pigou remarked that "the utmost we can hope for is a measure which will be independent of what the state of tastes and distribution actually is in either of the periods to be compared" (1932, p. 58). For example, the Consumer Price Index is an adjustment used to value past output at current prices. In effect, GDP, when deflated this way, is recalculated for all past years and then compared to current-year GDP. Alternatively, GDP can value current output at past prices. The two calculations closely track each other but are not equivalent.

5.2 Building Weights Via Benefits Transfer

When it is not feasible to conduct original non-market valuation for all the important services of an ESI in its specific spatial and temporal context, the transfer of virtual price information from other contexts is an alternative (Desvousges et al. 1998). To increase their applicability, such benefits transfers should adjust the estimates for differences in households and resources across contexts (Loomis 1992). One recent approach to such adjustments is the structural meta analysis of Smith and Pattanayak (2002). Their approach calibrates the parameters of a preference function to benefits estimates in the literature, each viewed as a draw (with error) from different points on the function. This approach is particularly attractive when differences in incomes and other population differences are a priority. In the context of an ESI used in joint models of ecosystems and economic systems, it also would have the advantage of yielding index numbers for the ecosystem consistent with the structure of preferences on the economic side of the model. However, this method may not be feasible when there are a large number of quality factors that affect WTP.

An alternative approach, which we introduce here, is a reduced-form regression of WTP on various factors. While this is an old approach for benefits transfer econometrically speaking (e.g., Walsh et al. 1992), our innovation is to introduce landscape-dependent indicators of the contribution of ecosystems to final goods and services (i.e., $\partial A/\partial B$ in equation 1) and landscape-dependent indicators of substitutes and complements, as in Boyd and Wainger (2002; 2003), as well as indicators of population differences.

WTP, while not directly observable, is a function of various characteristics that are observable. Let WTP weights p_i be denoted as a function of indicators I, so that

$$(18) \quad p_i = F(\boldsymbol{I}).$$

In principle, this function, on a service-by-service basis, can be calibrated by relating observable indicators I to existing WTP estimates of service value. Were this possible in practice, location and ecosystem-specific indicators I could be used to transfer monetary WTP estimates to locations where they are not available.

Currently, service-specific WTP estimates are too few in number to serve as a basis for such transfers across the national landscape. This particular empirical strategy is best thought of as an aspiration for an ESI. Nevertheless, we can lay out the basic strategy in more detail. We first derive a set of examples that connect the analysis of specific service flows to concrete data. The strategy in these examples is to first define a unit of account q. As discussed in section 3, the choice of units is driven largely by practical considerations.

With the unit of account defined, we identify a set of WTP proxies and organize them into three basic groups. First, because we are interested in social WTP, we look for population measures of the number of households or individuals likely to benefit from the service to the extent that population is not captured in the service measure q.[47] For example, the number of households within the viewshed of a natural area, the number of recreators with access to a stream or lake, or the number of homes being provided clean drinking water can be used to weight the service provided by a particular ecological feature (the service).

Second, we look for quality factors likely to affect WTP that are not captured in the definition of q. As discussed earlier, we might measure wetland-specific services via q. However, we may be able to rank the quality of particular wetlands by their proximity to sources of polluted runoff or likely flood pulses. These factors can be included to provide location-specific quality weights p_i. Another aspect of this service's quality is the value of homes, businesses, and infrastructure protected from flooding. For a given unit of service (e.g., a wetland acre), the more value protected by that service unit, the higher the quality of service being provided and the higher the WTP for flood mitigation services.

Third, we look for substitution and complementarity factors. Included in this category are measures of the scarcity of the service. Close substitutes for the service in question will depress willingness to pay for it, and thus the appropriate weight p_i. For example, the presence of abundant wetland resources in a region will tend to suppress any given wetland site's value. Where visual amenities are concerned, WTP also will be related to the scarcity of the amenity within the viewshed. A measure of this is the availability of other acres of visible natural land cover, surface water, or other visually desirable land uses to residents of or visitors to the site's viewshed.

For recreational services, the importance of substitute recreation to WTP requires the definition of service areas or zones.[48] In general, the relevant service area will differ depending

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[47] See discussion in Section 3.3.

[48] This is also relevant to the analysis of visual amenities.

on the service in question. A key methodological issue in any recreational benefits study is the determination of the appropriate "choice set" facing anglers, hunters, hikers, and birders.[49] For a given recreational benefit, analysis of other forms of recreation are relevant as weak substitutes. We suggest substitution indicators at a variety of scales, local to regional. These indicators can then be empirically evaluated as part of the reduced form meta analysis. For certain types of recreation, angling included, the relevant population may include transient vacation populations that can be assessed via tourism-related indicators.

WTP for ecosystem services will in some cases also be dependent upon the presence of complementary assets or services. This is particularly true in the case of recreational services, where access to natural areas is important. Access tends to be provided by complements, such as boat ramps, docks, and public parks and beaches.

5.3 Reduced Form Benefit Transfer

The first step in our proposed benefit transfer approach is to assess the predictive power of these population, quality, substitution, and complementarity indicators on existing dollar-based WTP estimates. Several things should be noted about this step. First, there is only a small set of available benefit estimates for use in calibration relative to the scale of activity necessary for national or regional analysis. Second, there are reasons to believe that existing benefit estimates are unlikely to be representative of benefits arising across the broader landscape. Third, there are reasons to question the accuracy of any dollar-based ecological benefit estimate given the inherent challenge of benefit estimation in the absence of market prices.

In principle, however, WTP indicators—such as those presented above—can be used to leverage the results of relatively costly, time-consuming, and expert dollar-based WTP studies. Indicators, if properly validated, are the bridge between relatively isolated econometric studies and regional or national mappings of ecological benefits. The corollary, of course, is that the way we derive the weights p_i for use in an ESI is by calibration based on estimates arising from non-market valuation studies.

[49] For a good collection of studies that address this issue, see the special issue of *Marine Resource Economics*, Vol. 14, no. 4, 1999.

In general, note that the WTP indicators we propose can be thought of as the location-specific independent variables necessary to shift WTP estimates or functions in any benefit transfer study. Moreover, the landscape data that we argue is necessary to an ESI would be invaluable to any econometric analysis of environmental benefits.

We also note that weights can be derived in ways other than econometric analysis of revealed behavior. Expert elicitation, citizen juries, mediated modeling involving stakeholders, and even political referenda are all processes whereby weights are either explicitly or implicitly derived. Some of these methods may better respond to demands for collective learning and decision-making as an alternative or complement to more technical analysis (Van den Belt 2004). In contrast to econometric estimation of weights, these alternatives involve a closer marriage of choices and learning. Typically, these methods involve a collective process whereby alternatives, relationships, and values are actively debated as part of the weighting process.

6. Conclusion

This paper draws upon ecological principles and the economic theory of index numbers to inform the design of an ESI. The design of an ESI first involves a sound theoretical underpinning—what we have termed the index's architecture. The architecture highlights the factors necessary to a consistent treatment of preferences and ecological interactions and disciplines the choice of observable indicators relevant to the valuation of specific services. For an ESI to be valid both economically and ecologically, a precise definition of ecosystem services is required. We have suggested a formal definition, illustrated the definition, and contrasted it with alternative definitions found in both economics and ecology.

An ESI generates ambitious implications for both the economic and ecological sciences. First, we identified the ways in which economics demands greater predictive power from theoretical and empirical ecology. This is no small challenge. Ecological prediction is extremely difficult given the complexity of the biophysical phenomena in play and the lack of historical data amenable to predictive experiments. Moreover, compared to economics, prediction is less of a preoccupation within the biophysical sciences. In the near term, economists should try to accept (and treat sympathetically) the lack of predictive "production function"-based ecology.

The paper also proposes an econometric agenda for the validation of WTP estimates. This is a significant challenge in its own right. The challenge to this validation is not in the econometric methods necessary to the validation. Rather, the challenge is associated with the inherent difficulty of original dollar-based estimations and the conduct of a large enough number to validate the broad range of estimates an ESI requires.

In terms of the data required, much of it already exists. Importantly, many of the service measures and factors influencing WTP are available in a spatial format. As we have argued, this is particularly important because services, their quality, and their importance to welfare are strongly dependent on location. The federal government, states, counties, and localities currently collect geographic information systems data in accessible forms covering a large array of data types, including data on the presence of rare and endangered species, recreational opportunities, land cover, future land use, watershed land cover, floodplain characteristics, and roads and trails. Vast amounts of socioeconomic data, including population and other demographic data, are centrally distributed through the U.S. Census Bureau and are aggregated by census block, or block group, city, census tract, county, state, and for the nation as a whole. Federal and state databases, for example, cover census data by census tract, road networks, parks, housing and commercial and industrial buildings, public and private water supplies, historical sites, riparian and coastal characteristics, aquifers, and topography, among other things. Regional economic databases are available from several sources, such as the U.S. Department of Agriculture National Agricultural Statistics Service and the U.S. Bureau of Economic Analysis, for evaluating economic characteristics of regions of various sizes (counties, congressional districts, etc.).[50]

An index of the ecosystem services enjoyed by society could play an important role in public policy. First, it could allow for a more comprehensive measure of well-being by acting as a non-market complement to existing national accounts.[51] Second, an index of ecological services, many of them public goods, could provide the public with a marker of the ways in which government and private sector economic activity affect ecosystem-based well-being over time. Like the existing national accounts, an ESI would condense a set of complex information

[50] Boyd and Wainger (2002) provide a more complete inventory of data types and an application to habitats in Florida.

[51] Indeed, adding ecosystem services and other non-market goods and services is a leading item on the wish-list of GDP reformers. Other items on the list include measures of other public goods such as education and safety, and the distribution of income (see Pigou 1932 for early discussion). Nordhaus and Tobin (1973) provide one attempt to address these shortcomings within a GDP context, while quality of life indicators (e.g., Anand and Sen 1994) provide an attempt to do so outside of it.

into a simple performance measure. Also like the national accounts, this measure would prompt analysis, interpretation, and revision by a range of experts over time.

An ESI might also be used in academic models of ecological and economic interactions. Such models must simplify reality by reducing the dimensionality of various services, and the theory of index numbers can provide a consistent way to do so. We have highlighted the most important factors required in making indexes consistent with preferences and with ecological interactions so that they might reflect the feedbacks between the two systems. Finally, insights from economic index numbers also can be used in models of pure ecology where reducing the dimensionality of the problem remains important.

By standardizing measurement and consistently aggregating ecological outputs, an ESI would be a large improvement over current practices, where information about ecosystem conditions remains haphazard and where aggregate measures of services omit the role of ecosystems.

Tables and Figures

Table 1. R^2 of Regression Using Various Lagged Indicators to Predict Changes in Future Ecosystem Services in a "Business as Usual" model of the Neuse River Estuary (Finnoff and Tschirhart 2005)

	Lags			
Indicators	1	1 to 3	1 to 5	1 to 5 plus 3rd-order auto-reg
Autoregressive only	0.65	0.67	0.68	N/A
Simpson Diversity Index	0.00	0.13	0.15	0.83
Dinoflaggelate population	0.00	0.58	0.64	0.80
Zooplankton and Clam Populations	0.70	0.88	0.92	0.99
Dinoflaggelate, Zooplankton, and Clam Populations	0.72	0.93	0.96	0.99

Table 2. R^2 of Regression Using Various Lagged Indicators to Predict Changes in Future Ecosystem Services in a "30% Nitrogen Reduction" Model of the Neuse River Estuary (Finnoff and Tschirhart 2005)

	Lags			
Indicators	1	1 to 3	1 to 5	1 to 5 plus 3rd-order auto-reg
Autoregressive only	.01	.18	.18	N/A
Simpson Diversity Index	.13	.59	.68	.88
Dinoflaggelate population	.18	.44	.60	.73
Zooplankton and Clam Populations	.13	.86	.95	.99
Dinoflaggelate, Zooplankton, and Clam Populations	.24	.87	.96	.99

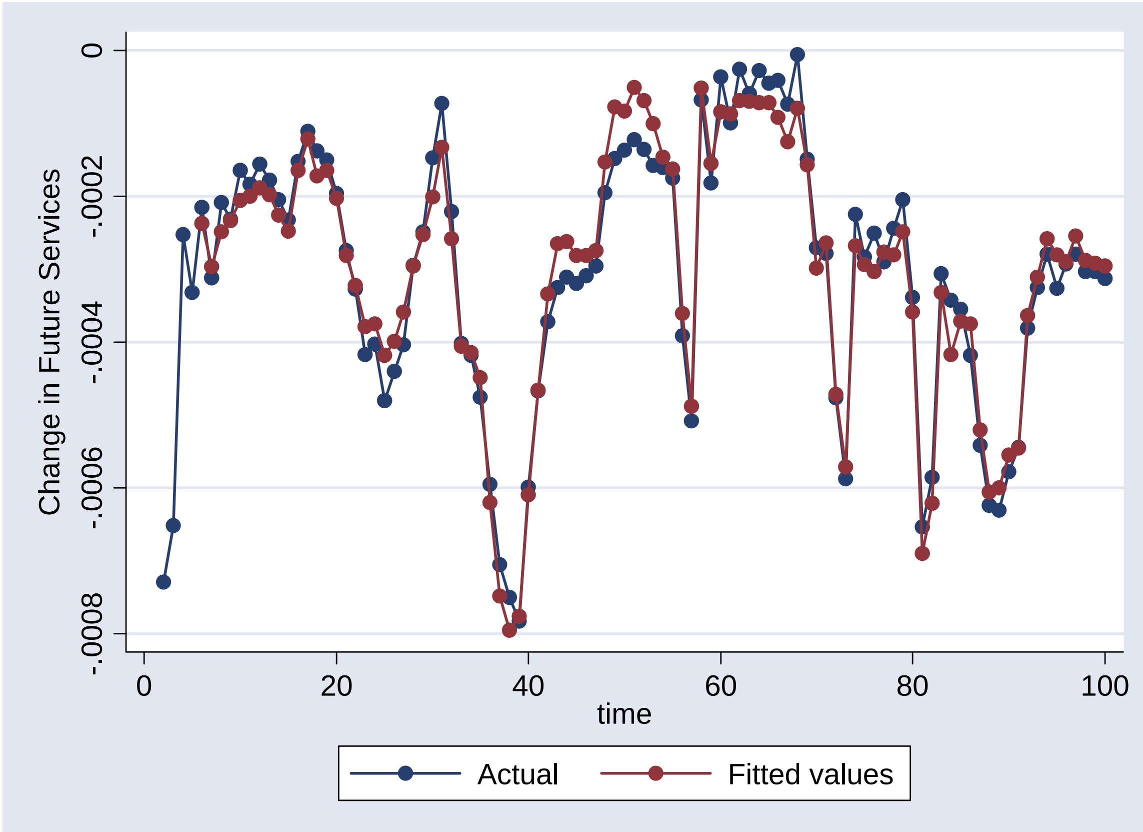

Figure 1. Predicted vs. Actual Changes in Future Services Using 5th Order Lags of Zooplankton, Clam, and Dissolved Oxygen Indicators in a "Business as Usual" Scenario

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