PAPER # 1

COMPARING ECOSYSTEM SERVICES AND VALUES

With illustrations for performing Habitat Equivalency Analysis

by

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ABSTRACT

An ecosystem can be "any spatially explicit unit of the earth that includes all of the organisms, along with all of the components of their abiotic environment within its boundaries" (Likens, 1992). Ecosystems exist and interact at many different geographic scales. The services and products they generate and the benefits they provide depend on these interactions and, as a result, are very site-specific. To compare ecosystems on the basis of their services or values, it is necessary to consider their landscape context as well as their biophysical characteristics.

Unfortunately, conventional ecosystem *assessment* methods are based primarily on ecosystem morphology (studies of their biological form and structure). They are useful for assessing and comparing on-site features of ecosystems and their capacity to provide certain functions. However, they usually do not take account of the interdependencies between ecosystems at higher and lower scales or the effects of landscape context on whether ecosystem functions will take shape or will benefit people.

Ecosystem *valuation* methods are limited in scope at the other extreme. They attempt to assign values to ecosystem services, usually in absolute (dollar) terms, but rarely give consideration to the specific biophysical or landscape features that generate them. Because many of the values of ecosystems are not traded in markets, these methods rely heavily on applications of recently developed "nonmarket valuation techniques." These techniques can be extremely expensive to apply and are reliable only when applied to well-defined products and services in specific contexts. The cost of applying these methods to the full range of services and products provided by even a single ecosystem is usually prohibitive. Moreover, since these services and products and their values are site-specific, estimated values for one site usually cannot be transferred to another site without additional research.

At least for now, the results from ecosystem assessment and valuation methods provide only part of the information needed to compare ecosystems in terms of their services and values. However, recent research related to wetlands suggests that practical and reliable indicators of relative ecosystem values can be developed based on the fact that the functions, services, and values provided by ecosystems depend in predictable ways on *on-site biophysical characteristics* (e.g., soil, vegetative cover, hydrology)and *landscape context* (e.g., proximity to certain features of natural and human landscapes). On-site characteristics determine the *capacity* of an ecosystem to provide various functions (e.g., support waterfowl). Landscape context determines: 1) if the ecosystem will have the *opportunity* to provide these functions (e.g., attract waterfowl), and strongly influences 2) what *services* will flow from the functions (e.g., hunting and birding opportunities), 3) the *values* that will flow from those services (e.g., how much people are willing to pay), and 4) the *distribution* of benefits to various segments of society (e.g., urban or rural, rich or poor).

This paper describes how these two sets of factors associated with site-specific characteristics and landscape context can provide the basis for indicators of the relative economic value of ecosystems. Such indicators would be useful for two reasons. First, they would provide a basis for prioritizing ecosystem protection and restoration efforts, and for establishing requirements and trading rules to govern compensatory restoration and mitigation programs. Second, they would provide a basis for applying "benefit transfer methodologies" whereby economic values estimated for one ecosystem might be adjusted to reflect those provided by ecosystems with different characteristics or different landscape contexts.

The paper has three sections. Section 1 identifies some typical ecosystem functions and related services and presents some concepts and terms that are useful for comparing ecosystem values. Section 2 identifies two sets of criteria for developing indicators of ecosystem value: one based on capacity, opportunity, payoff, and equity considerations under fixed landscape conditions, and another based on factors such as scarcity, vulnerability, sensitivity, and reversibility under changing landscape conditions. Section 3 illustrates why such indicators are important by showing their essential role in performing habitat equivalency analysis (HEA) and in "scaling" primary and compensatory restoration projects as required under the 1996 amendments to the Oil Pollution Act of 1990.

SECTION 1

ECOSYSTEM BASICS

What Is an Ecosystem?

Strolling the beach you encounter a pile of putrefying organic matter. Whether it is a week-old dung heap or the remainder of a month-old whale carcass, it is an ecosystem. So is the beach you are on (including the pile), the larger estuary of which it is a part, the reef that is breaking the waves you can hear in the distance, and the ocean that you know is just beyond the fog bank. The term *ecosystem* refers to "any spatially explicit unit of the earth that includes all of the organisms, along with all of the components of their abiotic environment within its boundaries."[1] Ecosystems exist and can be characterized and analyzed at microscopic or global scales; however, they interact in important ways across many different geographic scales. This fact makes it difficult to assess or compare the services or values of ecosystems without considering their specific landscape contexts.

Sources of Ecosystem Value

The natural world consists of approximately 250,000 plant species and, excluding insects, about 1.1 million animal species. If insects are included, the number of species ranges from 5 million to 30 million.[2] Each species and its respective habitat exist in hierarchical organizations that combine at various scales to form ecosystems. A few of these individual components of ecosystems contribute in direct and measurable ways to economic welfare (e.g., timber, crops, and edible fish); a few more contribute to the quality of life in other noticeable ways (e.g., dolphins, songbirds, and wildflowers). However, most of the millions of species that exist in nature contribute in obscure and roundabout ways to human welfare (e.g., pollinators and decomposers on land; benthic organisms, plankton, coral, and forage fish at sea). Their lives and functions are so intertwined with each other and with surrounding ecological landscapes that their individual contributions to human welfare, as a practical matter, cannot be isolated. Nevertheless, their contributions can be inferred from the *values* that people attach to the *functions* and *services* provided by the ecosystems of which they are a part. This is why assessing the socioeconomic value of ecosystems, despite all the inherent difficulties, is so important. It is the only way to show how most of the natural world contributes to human welfare. Therefore, it is the only way to use conventional (anthropocentric) concepts of benefits and costs to justify protecting or restoring natural systems.[3]

Limiting the Scale of Comparison

In scientific literature, ecosystems are often compared on the basis of the "complexity" and "richness" of their internal biological hierarchies and the "interconnectedness" of these hierarchies with each other and with ecosystems at other scales. The focus of this paper, however, is criteria for comparing the beneficial effects of ecosystems on people. As a result, it will be necessary to gloss over many specific differences in the "interconnectedness" and "richness" of ecosystem processes and to maintain a relatively narrow focus on the outcomes of

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[1] This widely used general definition of an ecosystem is from Likens (1992).

[2] There are disagreements over the exact number of plant and animal species. The numbers used here are from a standard reference on biodiversity by E. O. Wilson (1988).

[3] In this paper the terms *benefits* and *values* are used interchangeably and are anthropogenic in the sense that they refer strictly to the beneficial effects of ecosystem functions on people. In strict economic terms this would be measured as the aggregate "willingness to pay" by all individuals for all of the products and services generated by all of the functions of an ecosystem.

ecosystem processes. However, practical frameworks for comparing ecosystems in terms of their value to humans can be consistent with prevailing ecological theories and models without getting too deeply involved with them. Differences in the complexity of biological hierarchies within ecosystems, for example, are reflected in factors related to their biophysical characteristics, and linkages between ecosystems at different scales are reflected in factors related to their landscape context. Observations about these factors, therefore, reflect some important underlying ecological linkages that are too complex to deal with directly. Fortunately, these same factors also determine the capacities of ecosystems to provide certain functions and the outcomes of those functions, and strongly influences the services and values that will flow from them. Later in this paper they are proposed as the basis of an indicator system that, in the absence of conventional (dollar) measures of value, can be used to compare ecosystems on the basis of their expected values.[4]

Uncertainty of Ecosystem Functions and Values

Ecosystems are constantly changing (e.g., through succession) and adapting to change (e.g., hurricanes, wildfires, oil spills). As they reach certain thresholds, they can also make unexpected shifts from one successional trajectory or evolutionary pattern to another. One important new finding receiving attention in the ecological literature is that ecosystem changes are faster at relatively small scales (e.g., ant pile or pond) than at larger spatial scales (e.g., wetlands or watersheds).[5] Since perturbations affecting populations and communities at different scales ripple through ecosystem hierarchies at different speeds, there are always trends operating on various ecosystems scales that may be difficult to notice at that particular scale. This probably accounts for the high failure rates and wide variability of outcomes from ecosystem restoration projects.[6] It also suggests that comparing expected ecosystem values by considering the mix of functions and services that would be provided by ecosystems in a static landscape context may not be adequate. Changes in landscapes due to changing land use patterns, weather, and other factors will not influence all ecosystems locations in the same way. As a result, the risks and uncertainties associated with the expected streams of ecosystem services and values from similar ecosystems (even in currently similar landscape settings) may not be the same.[7]

Differences in the susceptibility of ecosystems to landscape changes, and in the ability of people to respond to or adapt to those changes, affect the risks and uncertainties associated with the expected services and values of different ecosystems. These differences are the reason why the next section outlines two distinct sets of criteria for comparing ecosystem services and

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[4] The difficulties of estimating the use and non-use values of on-site and off-site services provided directly and indirectly by ecosystem functions are summarized later in this paper and described in more detail in Kopp and Smith (1993) and Smith (1996).

[5] Scale, as it is used here, refers to the spatial or geographic range of an ecosystem. The relationship between ecosystem scale and ecosystem management was addressed recently in Gunderson, Holling, and Light (1995) and in a special issue of *Ecological Applications* (Pitelka, 1996b).

[6] The high failure rate in historical wetland restoration projects and the reasons for them were described recently in National Research Council (1992) and Shabman, Scodari, and King (1994).

[7] The important point here is that the risks associated with streams of services and values expected from ecosystems at different sites are different not only because of differences in observable site and landscape conditions, but because of differences in the exposure and vulnerability of different sites to changing landscape conditions.

values. The first addresses expected values under current landscape conditions; the second addresses differences in risk and uncertainty related to the impacts of changes in natural and human landscapes on different ecosystems and on different people. Recent evidence shows the influence of uncontrollable environmental factors such as extreme weather and the spread of noxious weeds is far more powerful in young (recently restored or created) ecosystems than in mature (undisturbed) ecosystems.[8] This has significant implications for the way risk and uncertainty should be factored into comparisons of natural and restored ecosystems.

Focusing on Ecosystem Comparisons

Scientists who compare ecosystems to learn more about them usually focus their attention on differences in ecosystem processes and ecosystem dynamics. However, because this paper is concerned with the contributions of ecosystems to human welfare, the focus is on factors that reflect differences in the expected outcomes of those processes. There are four attributes of ecosystems that represent building blocks between what is known about ecosystems and the benefits they will provide. They are:

- Ecosystem **Features**—the site-specific characteristics of an ecosystem (e.g., soil, ground cover, hydrology). These establish its capacity to support various forms of life and perform various biophysical processes.

- Ecosystem **Functions**—the biophysical processes that actually take place within an ecosystem. These can be characterized apart from any human context (e.g., fish and waterfowl habitat, cycling carbon, trapping nutrients).

- Ecosystem **Services**—the beneficial outcomes that result from ecosystem functions (e.g., better fishing and hunting, cleaner water, better views). These require some interaction with, or at least some appreciation by, humans. However, they can be measured in physical terms (e.g., catch rates, water quality, aesthetics).

- Ecosystem **Values**—defined in conventional economic terms as the aggregate "willingness-to-pay" by all individuals for all of the services associated with all of the functions of an ecosystem. These are usually measured in absolute (dollar) terms associated with specific services. However, they can be expressed in relative terms (e.g., using indicators) for purposes of comparing ecosystems.

These terms describe attributes of ecosystems that are obviously related to one another and are sometimes used to represent one another. (Improvements in fish habitat or in the abundance of fish, for example, are sometimes used to represent improvements in fishery-related values.) However, there are three reasons why it is important to maintain clear distinctions between them. First, the information needed to evaluate each of them and the criteria used to assess ecosystems with respect to each of them are significantly different. The features of an ecosystem that might give it a high capacity to provide a particular function (e.g.,

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[8] The evidence that the functions of restored ecosystems are less predictable than those of naturally occuring ecosystems was reported by Kentula and cited in a special issue of *Ecological Applications* on wetland mitigation (Pitelka, 1996a). This national study was based primarily on comparisons of vegetative ground cover at natural and restored wetland sites and yielded results that are consistent with papers presented at symposiums of the Association of State Wetland Managers, Inc. (Kusler, Quammen, and Brooks 1986).

suitability as waterfowl habitat) do not guarantee that it will actually provide a high level of *function* (e.g., attract or support the greatest number of waterfowl). Ecosystems that provide the highest level of function may not provide the highest level of *service* (e.g., birding, hunting, educational, scientific opportunities). Those that provide the highest levels of service may not provide the greatest *value* (e.g., aggregate "willingness-to-pay" for birding, hunting, educational opportunities). Lastly, the ecosystem that generates the greatest value may not result in a distribution of benefits considered *equitable* (e.g., opportunities for rich vs. poor or urban vs. rural).

Second, there are significant differences in the attributes of ecosystems that allow them to provide different types of ecosystem functions and services. Figure 1 provides a list of the most often cited functions provided by ecosystems and identifies a few of the associated services. Some of the functions and services listed are provided best by ecosystems that are located away from people and are surrounded by undisturbed natural landscapes (e.g., endangered species habitats); others require that the ecosystem be relatively close to people (e.g., educational and recreational opportunities, flood damage prevention, aesthetics). Similarly, some services, such as those associated with sediment, nutrient, or contaminant trapping, are provided only if the ecosystem is located near disturbed landscapes where runoff is a problem; others, such as breeding habitat for migratory waterfowl, are provided more effectively at sites in undisturbed landscapes. Unless the factors affecting various functions, services, and values of an ecosystem are considered separately, it is impossible to identify important environmental and socioeconomic tradeoffs, or to compare ecosystems with similar features in terms of the functions, services, and values they can be expected to provide.

The third and most important reason for distinguishing between these terms is that the generally available analytical methods for assessing and comparing ecosystems usually do not strive to make clear distinctions. As a result, many of them tend to mask, rather than clarify, critical linkages between ecosystem features, functions, services, and values, which can differ significantly from site to site. A review of specific methods is beyond the scope of this paper, but a few brief observations can be offered about the two most widely used approaches: ecosystem *assessment* methods and ecosystem *valuation* methods.[9]

Ecosystem *assessment* methods were developed primarily by scientists and evolved as extensions of morphological studies—studies of the form and structure of biological systems.[10] These studies focus on ecosystem features and often employ indicators that refer to an ecosystem's biophysical capacity to provide various functions. Although they may refer to "functional values" or "value indices" when describing the "functional capacity" of ecosystems, and may use these methods to compare ecosystems, they rarely address ecosystem values as the

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[9] The brief comments offered here are needed to justify the proposed development of "leading indicators" of ecosystem values in the following section. These indicators are only worthwhile because useful direct measures of ecosystem value are not available and are not forthcoming using other methods. Refer to Amman, Allen, et al. (1991, 1993) for reviews of specific ecosystem assessment methods and Freeman (1993), Kopp and Smith (1993) and Smith (1996) for reviews of specific ecosystem valuation methods.

[10] The most modern and fully developed of these methods is the "hydrogeomorphic" or HGM method which was released in late 1995. (See Brinson 1993). HGM, like previously designed morphological methods, provides a basis for comparing wetland features; however it is being used as the basis for guiding wetland mitigation trades under Section 404 of the Clean Water Act, and for other purposes that imply that it represents a useful measure of wetland value.

term is defined above. Ecosystem assessment methods provide only the front-end part of the analysis required to compare ecosystems on the basis of the services and values they provide.

By contrast, ecosystem *valuation* methods attempt to assign values to ecosystem services, usually in absolute (dollar) terms, but usually without much regard for the specific ecosystem features or functions that generated them. Because many services of ecosystems generate "off-site" and "non-market" benefits, these ecosystem valuation methods rely extensively on the application of recently developed "non-market" valuation techniques to assign values to specific ecosystem services. However, these techniques are far too expensive to be applied to the full range of services listed in Figure 1.[11] As a result, they have usually been applied to only a sub-set of them, and therefore serve more to illustrate ecosystem values than to provide a comprehensive accounting of them. The fact that ecosystem values are so site-specific also limits the usefulness of these methods. To be credible, they must be applied to estimate the value of services provided by a specific ecosystem in a particular landscape context. Using the estimates of value developed for the services of one ecosystem at another site requires additional research.[12] Ecosystem valuation methods provide only the back-end part of the analysis that is required to compare ecosystems on the basis of the services and values they provide and are usually too expensive to be used for this purpose.

Landscape Context of Wetlands

Although exact distances differ from region to region and site to site, Figure 2 illustrates two important sets of spatial relationships that help establish the effects of location on wetland functions and values. In this figure distance from a wetland site is measured along both the horizontal and the vertical axis. Along the horizontal axis distance measures the geographic range over which various types of wetland values accrue. Most of the value associated with timber, hay, and cranberry production, for example, require proprietary rights (own or rent); the values associated with on-site hunting and fishing, most scientific and educational uses, and most aesthetic and spiritual benefits require access, or at least near proximity. Other important wetland values, such as spawning, feeding, and nursery habitats for migratory waterfowl and fish extend across much greater regional scales. Still other functions, such as those associated with carbon cycling and biodiversity support, generate values that are national or global in scale.

Similarly, distance measured along the vertical axis in Figure 2 reflects the extent to which various wetland functions and values depend on attributes of the surrounding ecological landscape. For example, even though they generate benefits across much different scales, the capacity of a forested wetland to provide timber or sequester carbon are both shown to depend very little on conditions outside the wetland site itself. A wetland's capacity to provide fishery habitat, on the other hand, may depend on water quality in lakes and streams several miles away and on physical connections with them and with the ocean. At the extreme a wetland's capacity to provide waterfowl habitat might depend on conditions along migratory routes and in breeding areas thousands of miles away.

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[11] So far, a technique called "contingent valuation" is the only way to explicitly attach "passive-use values" to ecosystem services. A description of the procedures that are required to apply this method to acceptable standards under OPA is provided in NOAA (1992).

[12] Studies aimed at transfering estimates of value developed for one site to another site are generally referred to as "benefit transfer" studies. No generally accepted methodology exists for conducting benefit transfer studies. The approach developed in the following section for comparing ecosystems in terms of their relative values may provide a reasonable basis for conducting benefit transfer studies.

The relationships illustrated in Figure 2 with regard to wetlands reveal two facts that are important when comparing the services and values of ecosystems. First, the location of an ecosystem can have a significant effect on the mix of functions and values provided by an ecosystem, and the levels of services and values associated with them. Second, the location of ecosystem can have significant effects on the distribution of services and values.

The Critical Science–Policy Gap

The analytical problems outlined above are the focus of long-term research. Scientists are giving more attention to landscape context in the development of assessment methods, and economists continue to look for ways to reduce the cost of estimating ecosystem values and improve their reliability. However, these two separate areas of research, even if they are successful, may not be of much practical value to policy makers and jurists making decisions that involve comparing ecosystems in terms of their relative values. These decisions cannot be made on the basis of "value-free" scientific assessments of ecosystems, even if they are improved and expanded to consider landscape linkages. Nor can they be made on the basis of improved estimates of the value of specific ecosystem services, even if they could be provided more reliably and more affordably. The fact is, in most legal and regulatory contexts where decision-makers are asked to compare ecosystems on the basis of their values, they are asked to compare their *expected future services and values* on the basis of information about their *current biophysical features and landscape contexts.*

To be useful under these circumstances, there must be some way for the assessment of observable ecosystem features to be linked forward to expected ecosystem functions and services, and for the valuation of ecosystem services to be linked backwards to observable or measurable ecosystem features. Most of the ongoing research to improve the scientific basis of ecosystem assessment methods and the credibility of ecosystem valuation methods will not fill this gap.[13] Arguably, improving methods for linking current ecosystem features with future flows of ecosystem services is more important for evaluating the socioeconomic tradeoffs associated with ecosystems than improving methods of attaching values to services.[14]

Ecosystems as Natural Capital

Taken together, ecosystems encompass most of what is important to life, so there is no need to justify their overall value. However, all ecosystems are not equal in terms of their environmental or socioeconomic importance and as a result, they do not all deserve the same level of protection or the same level of restoration spending. "Natural capital" is a term used to

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[13] In order to keep methods of ecosystem assessment "objective," scientists continue to focus most of their attention on features and functions, not services and values. Economists are bound by professional standards that require clearly identifying what is being valued using non-market techniques; this means that their work will continue to focus on estimating values for specific ecosystem services.

[14] There are two methods of trying to make this ecological–economic link: deterministic models and indicator systems. Deterministic models attempt to describe the link using mathematical equations; these have not advanced to the point that they can be used for this purpose. For a review of these models and their strengths and limitations consult the special issue of the journal *Ecological Applications* (1996b). Criteria for developing indicator systems that might be used to forecast ecosystem values on the basis of ecosystem features are developed later in this paper.

refer to ecosystems and components of ecosystems as specific assets that contribute in specific ways to economic welfare. Natural capital is comparable to manufactured capital or human capital in the sense that these terms focus on values (measurable or not) stemming from the streams of products and services they are expected to provide over time. This perspective of ecosystems, not as life-support systems but as collections of assets with specific values, is useful for many reasons. It provides a basis for comparing ecosystems in terms of characteristics that affect their capacity and opportunity to provide services and values; it allows investments in ecosystem protection and rehabilitation to be justified on more than purely environmental or emotional grounds; it allows ecosystems to enter social and economic accounting on a par with manufactured capital and human capital; and, most importantly, it permits more realistic assessments of the future costs and risks of economic development, industrial accidents, and policy decisions that degrade or destroy ecosystems today.

The Asset Value of Ecosystems

There are some important similarities between natural capital and manufactured capital that help illustrate the pathways by which ecosystems generate value. For example, with little or no out-of-pocket costs, ecosystems provide all of the basic services to the economy that are provided separately and at considerable expense by each of the three conventional forms of manufactured capital: *inventories*, *plant and equipment,* and *infrastructure*. Ecosystems have value as *inventories* because they are enormous storehouses of raw materials (e.g., timber, fish, beautiful views). They are valuable as *plant and equipment* because they include all the necessary support systems to replenish these inventories (e.g., wetlands and coral reefs). And ecosystems interact at various scales to provide the basic *infrastructure* that sustains natural and economic systems and supports all other forms of biological and industrial productivity (e.g., biodiversity and water, nutrient, energy, and carbon cycling). The contributions of ecosystems to human welfare can be compared on the basis of how they contribute in each of these basic categories.

When evaluating ecosystems on this basis, it is important to note most decisions involving ecosystems do not involve total losses of ecosystems or the elimination of ecosystem functions without perfect or near-perfect substitutes elsewhere in nature. In some cases, an injured ecosystem may lose only its capacity to provide some functions and services, and it may have the capacity to adapt or recover over time. For purposes of assessing an ecosystem's value as natural capital, therefore, some important and often overlooked questions involve the scarcity of the services it provides, the availability of perfect or near-perfect substitutes for them, the ability of the ecosystem to recover or to be restored or replaced, and the capacity of humans to adjust and adapt to temporary or even permanent ecosystem changes. The answers to such questions provide another basis for what might be referred to as "leading indicators" of ecosystem value.

Critical Questions about Ecosystem Value

The questions about ecosystem features and functions addressed by conventional assessment methods, and the questions about the values of specific services addressed by conventional valuation methods, are only a few of the important questions that need to be addressed when comparing the asset value (natural capital value) of ecosystems. The important questions that need to be addressed for this purpose (including some that are at least partly addressed by assessment and valuation methods) are:

1. What functions are provided by this ecosystem?
2. What services, products, and amenities do these ecosystem functions generate?
3. How much value, at least in relative terms, do people place on them?
4. Could they be provided just as well by other nearby or distant ecosystems?
5. Are there man-made substitutes that exist or could be developed?

6. What determines an ecosystem's ability to generate certain services and values?
7. With what reliability, precision, and frequency should ecosystem changes be measured?
8. Do changes in characteristics at one level in an ecological hierarchy reflect changes at other levels (e.g., forage or food fish)?
9. Do changes in an ecosystem at one location (e.g., a single wetland within a watershed) reflect changes at other locations (e.g., all similar wetlands in a watershed)?
10. How can normal fluctuations and cycles in the mix of ecosystem features and resulting services and values be distinguished from significant trends?
11. How reversible are ecosystem changes naturally or through technology?
12. Are bio-physical relationships within ecosystems linear or are there important threshold points beyond which there are abrupt shifts in the mix of services and values provided?
13. How can and do people adapt to not having certain ecosystem services?
14. And, most importantly, if waiting to measure actual ecosystems services is impractical or dangerous, what are useful "leading indicators" of them?

Lessons from Wall Street

Providing specific answers to most of these questions would be difficult and in some cases impossible. However, they are still the right questions to be asking. For purposes of this paper the important issue is whether there are practical ways to address them, and to organize information about them for purposes of comparing ecosystems. Practical ways of dealing with these kinds of questions, however, are not likely to be found in "scientific" literature aimed at reducing uncertainty about ecosystems and their values. It is more likely to be found in the results of research designed to help people make informed decisions in the face of uncertainty. Clues about how to develop practical methods for comparing ecosystems on the basis of their future services and values comes from what may seem to be an unlikely place: Wall Street.

Ecological systems may be more complex and more difficult to assess than market systems, and the expected values of natural capital may be more difficult to compare than those of manufactured capital. However, the two sets of tasks are very similar. Individuals whose livelihoods depend on forecasting the values (earnings streams and future prices) from various mixes of manufactured capital (corporate stocks) consult scientific research; but in the final analysis, the most successful rely on systems of indicators for making comparisons. There is simply too much relevant information about firms, industries, and markets and about financial, legal, and regulatory issues to make sense of investment alternatives in any other way.

The most widely used Wall Street indicator systems employ a variety of composite indicators employing terms such as diversity, resistance, resilience, vulnerability, and volatility. These are terms familiar to anyone exposed to modern literature related to landscape ecology or ecosystem health.[15] Wall Street indicator systems also employ concepts of "scale" including links with lower scale labor and input markets and with higher scale intermediate and final product markets. They give attention to "hierarchies" and consider "nested interdependencies" involving firms within industries within industrial sectors within national markets within global trading networks, and so on. However, there is one noteworthy difference between the way these terms are used on Wall Street and the way they are used in modern ecological literature. On Wall Street, they are used almost exclusively to provide clues about potential changes in the expected streams of economic returns from manufactured assets. In the ecosystem assessment literature, they are used almost exclusively to describe ecosystem processes or draw inferences about the health or capacities of ecosystems themselves with no specific focus on expected service flows or values.

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[15] See Karr (1992).

Some Practical Differences

The information needs of policy makers and jurists involved in comparing and trading natural capital are more similar to those of Wall Street investors than to designers of ecosystem assessment or valuation methods. They need practical information to help them deal with unavoidable uncertainty now, more than they need better ways to nibble away at questions that might help them avoid uncertainty some time in the future. As a result there are bound to be differences between the information needs of decision makers who focus on ecosystem outcomes and those of scientists who are concerned about ecosystem processes. In some cases such differences may merely be a matter of how data are organized and interpreted. Information about vegetative cover at a site, for example, can be used to analyze primary biological productivity and to determine how much the site contributes to fishing opportunities or fishing success in nearby waters. However, some differences are more significant. Information about on-site food web linkages that may be important for scientific studies, for example, may be far less important for purposes of comparing ecosystem values than information about the use of the site by migrating birds or fish or the accessibility of the site to schoolchildren.

There are two reasons why differences between the information needs of researchers and decision-makers are far more important in the case of natural capital than in the case of manufactured capital. First, the indicators used by Wall Street decision makers to develop indicators usually employ the same types of financial and market data that economic researchers use to analyze markets and industries. Second, the financial and market data that they use are collected routinely for other purposes. Unlike the biological components of ecosystems, each individual and corporation that makes up each industrial and market sector of the U.S. economy reports data about their economic and financial health routinely to the IRS, and to countless other local, state, and federal agencies. These institutions then sort and aggregate this information in various ways and published it at very little cost.[16] The cost of developing and testing indicators related to manufactured capital, therefore, is very low and usually does not require any dedicated research.

Comparable data about the components of ecosystems, on the other hand, need to be generated from the ground up through directed research and monitoring undertaken at considerable expense. Collecting information about one aspect of an ecosystem usually means that other important information will not be collected. Collecting indicators of how coastal wetlands contribute to fishery values, or what their capacity is to protect property from storms or floods, or what their vulnerability is to future development, or how restorable they are, for example, only marginally improves scientific understanding of wetland ecosystems. There are significant tradeoffs involved in designing research to support scientific studies of ecosystems and to support value-based ecosystem indicators.

The Need for Careful Indicator Specification

The fact that there are differences between the types of data needed to support scientific studies of ecosystems and the types of data needed to develop practical indicators of ecosystem services and values is very important. It means that every choice about the focus and scale of an indicator, and the frequency and precision with which the indicator should be measured, has implications beyond just how reliable a predictor it will be. It also determines how much money will need to be spent collecting data about specific ecosystems and, as a practical matter, what data about ecosystems will not be collected. It also means that although general criteria may be

[16] Industry and market indicators rely primarily on data collected and published by the U.S. Department of Commerce, U.S. Bureau of Labor Statistics, and the U.S. Security and Exchange Commision. By contrast, environmental indicators rely primarily on primary data collected specifically for the purpose of developing indicators.

developed for designing indicators of ecosystem values, the criteria used to specify indicators-how detailed and site-specific they should be—will not the same under all circumstances.

For purposes of assessing mitigation requirements for wetland permitting, for example, government agencies may make dozens of decisions each month involving small wetland trades within a single watershed. In such cases the stakes involved in each case may be relatively low, the locations and types of ecosystems may be quite similar, and a general-purpose set of indicators based on limited on-site testing may be adequate. On the other hand, for purposes of settling natural resource damage claims for environmental injuries caused by oil spills the stakes may be quite high and quite widespread and any basis of ecosystem comparison will need to be capable of withstanding strong technical and legal challenges. In such cases it may be appropriate to develop detailed sets of indicators to evaluate each type of injury, base them on documented statistical relationships, and apply them carefully using biophysical profiles of specific sites.

The following section develops general criteria for designing leading indicators of ecosystem values. These criteria help identify which attributes of ecosystems will be useful in predicting the services and values it will provide, and help identify information needs for developing indicators related to them. The criteria should also be useful in selecting indicators that provide an appropriate level of predictive value at an appropriate level of cost. However, they are not used here, and probably cannot be used, to develop a general set of leading indicators of ecosystem values that can be used in all circumstances.

SECTION 2

LEADING INDICATORS OF ECOSYSTEM VALUES

Background

An indicator can be defined as "something that provides a clue to a matter of larger significance or makes perceptible a trend or phenomenon that is not immediately detectable."[17] The significance of an indicator extends beyond what is actually measured to larger, more important phenomena. Indicators should serve two general purposes: 1) they should *quantify* information so significant changes or differences in the larger phenomena are more readily apparent and can be compared, and 2) they should *simplify* information about complex phenomena to improve understanding and communication.[18]

Indicators can be classified in many different ways on the basis of what they measure and how they are linked to the genuine focus of interest. One important distinction is between *current, leading,* and *lagging* indicators.[19] *Current* indicators refer to measures reflecting phenomena happening now, or more typically, phenomena that happened recently. In the absence of actual estimates of changes in recreational fishing values, for example, it is possible to use changes in the number of anglers, fishing days, catch rates, money spent per trip, distance traveled, or fishing gear purchased as indicators of changes in fishing values.[20] These are current indicators because changes in each of them can be linked with changes in aggregate "willingness to pay" for recreational fishing in the same period. Current indicators, such as these indicators of recreational fishing value, are useful primarily for what might be called "scorekeeping."

In economics, the term "leading indicators" is used to refer to variables that are measured not because they are good proxies for changes taking place, but because they provide clues as to what important changes are likely to take place in the future. Leading indicators are more useful than current indicators for making investment and management decisions. Typical leading indicators of future conditions in national and regional economies include housing starts, industrial equipment orders, producer price indices, and winter yields of feed corn. The widely used "U.S. composite of leading economic indicators" is a highly reliable indicator of changes in economic income six to nine months in advance. More specialized leading indicators based on trends in demographics, technology, and trade are used routinely on Wall Street to make more long-term forecasts for specific industrial sectors and markets. In ecosystems, leading indicators may be associated with changes in certain "keystone" or indicator species or with

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[17] This defintion of an indicator is taken from Hammond, et. al., (1995).

[18] When pondering questions about the future, such as the services and values of natural or manufactured capital, it is not an exaggeration to say the value of measuring anything depends on its usefulness as an indicator.

[19] A discussion of leading, lagging, and coincidental (current) indicators is provided in *The Economist* (1992).

[20] Because there are so many reliable current indicators of recreational fishing values they are often used as examples of how dependable non-market valuation can be. In the context of this paper it is important to note: 1) recreational fishing provides <u>use</u> value which is much easier to measure than the <u>non-use</u> values associated with most ecosystem functions; and 2) the problem being addressed here is how ecosystem functions can be compared in terms of their ability to provide services such as improved recreational fishing, not the problem of measuring the dollar values of these services once they occur.

changes in political, economic, or land and water use patterns known to result in ecosystem change. Declines in forage fish populations, for instance, may be leading indicators of future declines in fish stocks that feed on them. The relaxing of fishery regulations in already overcapitalized fisheries or the backlog of orders for new boats that will enter such fisheries may also be useful leading indicators that a fish population will decline.[21] The range of *potential* leading indicators of changes in the derived value of ecosystem services is limited only by imagination. The range of *practical* leading indicators of ecosystem services and values is limited by cost, reliability, statistical evidence, and other factors discussed later in this section.

Lagging indicators are important primarily as a way to confirm an outcome was the result of the phenomena on which the leading indicators are based. In economics, for example, it may not be clear for many months when increases in gross economic production peaked. However, utilization rates for manufactured capital peak about one month after gross economic production, job vacancies peak about one month later, growth in net business earnings about three months later, and unit labor costs about five months later.[22] These same variables lag declines in economic production with a similarly predictable pattern. If an economic upturn or downturn occurred, it could be a critical leading indicator of other important phenomena; the first signs of it may be observed changes in these lagging indicators. In the case of ecosystems, observed changes in invertebrate communities or the size of non-commercial fish populations may be useful *lagging indicators* that the decline in the previous year's fish abundance index was a result of habitat losses or water quality problems, not overfishing.[23]

Site-specific Criteria for Developing Indicators

Previous sections established that two sets of observable characteristics determine whether an ecosystem will generate certain streams of values. *Site-specific features* (e.g., soil type, vegetative cover, topographical features) determine the capacity to provide various functions); and *landscape context* (e.g., proximity to other features of the natural and human landscape) determines if the ecosystem will have the opportunity to provide these functions and strongly influences what services will flow from them, the benefits that will result, and the distribution of benefits.

To focus the development of indicators consider the factors related to capacity, opportunity, payoff, and equity associated with each specific ecosystem function listed in Figure 1. Each of these factors and general information needs can be summarized as follows:

Capacity: Does the ecosystem have the biophysical conditions necessary to provide this function?

[21] The term leading indicator, as used here, includes factors related to driving forces and pressures that can be expected to result in changes in the state of a resource system. For a review of these factors and their use in other types of indicator systems see OECD (1993a, 1993b) and Bakkes et al. (1994).

[22] The specific lags mentioned here are those used in The Economist magazine to report on the status of various national economies and are explained in a book published by the publishers of that magazine; see *The Economist* (1992).

[23] It may not be possible for many years to establish whether changes in catches or sightings of certain populations in a given year are a result of changes in abundance or changes in availability. Lagging indicators of catches and sightings elsewhere sometimes help establish which it was, and, if it was a result of a decline in abundance, whether fishing-related, habitat-related, or other factors were responsible.

Example: Can it support migratory waterfowl or filter nutrients?
Information needs: site specific biophysical characteristics

Opportunity: Is it located in the ecological landscape where it will serve this function?
Example: Is it situated along a flyway or adjacent to a farming area?
Information needs: location and landscape context

Payoff: How will providing this function at this location result in benefits to people ?
Example: How will attracting waterfowl to this site rather than another or improving the quality of adjacent water here rather than elsewhere affect people?
Information needs: location and landscape context

Equity: Who gains and who loses as a result of the ecosystem providing the function at this location, not elsewhere?
Example: Will attracting waterfowl to this site make them more valuable to some and less valuable to others (intragenerational equity) or more vulnerable to hunting and other pressures (intergenerational equity)?
Information needs: location and landscape context

Building Blocks of Ecosystem Value[24]

Working through an illustration related to one of the many functions provided by wetlands—nutrient trapping—will show how the four factors defined above reflect conditions that contribute to services and values of wetlands with specific features at specific sites.

First, consider the *capacity* of a wetland at a given location to filter nutrients and prevent them from having adverse impacts on adjacent water bodies. A wetland's capacity to provide this function depends primarily on its slope and vegetative cover and other site-specific biophysical features; these can be considered independently of its watershed context and are the focus of most ecosystem assessment methods.

However, the *opportunity* for the wetland to trap nutrients depends on the expected flow of nutrients from adjacent land which is determined almost exclusively by its location within the watershed, in particular its proximity to certain upland land uses (e.g., farms and construction sites vs. forests and grasslands). Opportunity determines the "rate of capacity utilization," which is a critical factor in establishing the level of function provided and resulting service flows and values.

Similarly, the *payoff* from the nutrient trapping function depends on the wetland's location, in particular the characteristics of the adjacent (receiving) water body and the resources that exist in it or depend on it. At one location, for example, there may be important water quality payoffs and improvements in nearby shellfish beds and finfish spawning areas because they are protected from overnutrification. At another location, a wetland with identical capacity to filter nutrients and the same opportunity to filter nutrients might be adjacent to a fast-moving, highly polluted river which empties directly into the open sea, resulting in no watershed-level

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[24] Some widely used wetland assessment methods include "social significance" components that employ terms similar to those used here. However, the socioeconomic components of those methods have never been fully developed or peer reviewed, and have never been used to develop relative indicators of wetland values. For a review of how twenty-eight of the most widely used ecosystem assessment methods deal with issues related to services and values see King (1997).

payoff at all. Differences in the residence time of receiving water adjacent to two wetlands, therefore, may be a reasonable index of differences in payoff. Note that these kinds of indices don't require ethical judgments.

And finally, there are *equity* considerations. Society may concern itself with the difference between identical wetlands that generate identical values at two different locations if there are significant differences in who gains from them (e.g., rich or poor, urban or rural). Equity considerations involve policy choices, not scientific or economic criteria, but the nature of the policy choices depend in critical ways on location. Who has access to the wetland or to the different fisheries or waterfowl populations that depend on them? Whose property is protected from flooding? Whose scientific or educational opportunities or aesthetic values are enhanced and diminished? Indicators of who gains and who loses from wetland mitigation trades, for example, can be developed quite reliably without making any ethical decisions or value judgments about whether the group that gains is more or less "deserving" than the one that loses.

Same Factors/Different Criteria

The influence of site and landscape factors on an ecosystem's capacity or opportunity to provide a function, and on the resulting services and values will be vastly different for different functions. However, this does not mean that the relevant factors themselves are different or that vastly different data are needed to estimate indicators related to each function. Preliminary field testing of this type of indicator system to evaluate the size, characteristics, and siting of vegetative riparian buffers in the Chesapeake Bay watershed, for example, suggests that a limited number of site and landscape factors related to soil, hydrology, shape, vegetative cover, distance to open water, upslope land use, accessibility, and so on can be used in different ways to develop indicators for many different ecosystem functions, services, and values.[25] Developing indicators for different ecosystems functions, in other words, does not require collecting different kinds of data for each function.

An Illustration

The situation depicted in Figure 3 illustrates how the criteria listed above affect the services and values of ecosystems. In the situation depicted two wetland areas are located on either side of a highway.[26] They are being compared on the basis of the values they generate by providing three specific functions: nutrient trapping, wildlife habitat, and fishery support. The two wetland areas are the same size, the same shape, and have identical biophysical characteristics. However, because of slight differences in their landscape contexts, Site A is shown to generate significantly more benefits in each of the three categories than Site B.

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[25] These initial field trials using the system of indicators described here (but without considering equity impacts) involved prioritizing riparian forest buffer restoration projects in three sub-watersheds of the Chesapeake Bay. The application was aimed at ranking each stream reach in terms of its capacity, opportunity, and payoff potential with respect to eight specific functions that were eventually aggregated into indicators in three areas: water quality, in-stream habitats, and terrestrial habitats. For details refer to King. Hagan, and Bohlen (1996) and King and Bohlen (1996).

[26] The term "on-site" is often used to refer to environmental mitigation projects that are undertaken at sites very near the site of the environmental injury. Suitable sites for "on-site" mitigation may be difficult or impossible to find, requiring "off-site" mitigation. Figure 3 illustrates, however, that , differences in location can be important even if suitable compensatory restoration sites can be found very close the injured site.

In this illustration there are no differences in the capacity of the two sites to trap nutrients, support wildlife, or protect and nourish coastal fish habitat. However, Site A has more opportunity to provide all three of these functions because of its proximity to upland land uses that generate nutrients, its closeness to the coast and adjacent fish habitat, and its accessibility to wildlife from the upland wildlife refuge area. The payoff from providing functions at Site A is also greater than at site B because of its accessibility to people and the fact that the fish habitat it protects is larger and less contaminated than the one adjacent to Site B. For sake of argument, Site A, and the fish and wildlife resources it supports, are also assumed to be located where they provide scarce aesthetic and educational opportunities to a large urban disadvantaged population. Site B, on the other hand, is surrounded by large tracts of private land and forest areas that benefit only a few relatively wealthy families.

Without judging the merits of protecting and restoring wetland capacity at *both* Site A and Site B, the evidence presented in Figure 3 leaves little doubt that the expected services and values from Site A exceed those from Site B. Investments in protecting and restoring wetlands at Site A, therefore, would yield greater environmental and economic benefits than protecting and restoring wetlands at Site B. In the situation depicted in Figure 3 the two sites are located very near one another, yet are shown to have significant enough landscape contexts to result in different functions, services, and values. This illustrates that the effects of location may be important even when making "on-site" comparisons.[27]

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[27] Since the two sites are adjacent, it might be logical to assume that the same people would gain from investments at either site. However, to illustrate that equity issues can be important even when comparing nearby sites, a few assumptions are made here about the access and proximity of the two sites to urban/poor and suburban/rich populations.

Criteria for Leading Indicators of Ecosystem Values

CRITERIA	TYPICAL QUESTION	TYPICAL INDICATOR
CAPACITY	Does this wetland have the **biophysical characteristics** to trap nutrients ?	soil type, vegetative cover hydrology *(What's the expected nutrient uptake capacity?)*
OPPORTUNITY	Is this wetland in a **location** where it can and will trap nutrients?	slope and land use of adjacent land *(Does it have a source of nutrients to trap?)*
PAYOFF	Will environmental and natural resources in adjacent water bodies be affected by the reduction of nutrient loading at this **location?**	residency time of nutrients delivered to receiving water *(What changes are expected if nutrients are trapped here?)*
	How important are the resources being protected at this **location** compared to those that would be protected elsewhere?	proximity of wetland to water important for: fishing, swimming, and drinking; finfish, or shellfish spawning, and feeding; and waterfowl spawning, feeding, and nursing
EQUITY	Whose recreational, educational, aesthetic opportunities are enhanced?	proximity of wetland to various populations; demographic and land use characteristics

Landscape-level Criteria for Developing Indicators

To make the comparison of Site A and Site B in the illustration even easier, assume that in the county's 10-year land use plan the area around Site A is designated "environmentally sensitive–recreation" and the area around Site B is designated " fast-track–industrial." The environmental and economic values provided by Site A are not only higher than those provided by Site B, they are less likely to be lost due to the future development of adjacent land. The same type of differences in risks might be associated with differences in exposure and vulnerability to risks from water diversion, sea level rise, and other factors. This suggests that the criteria listed above may only be suitable for forecasting ecosystem services and values in the relatively short term, as long as site conditions and landscape context are expected to persist. For some purposes it may be important to take a broader and longer perspective that accounts for: 1) the likelihood that landscape conditions may change; 2) the fact that different ecosystems may be more or less vulnerable (exposed to change), resistant (able to withstand change), and resilient (able to recover from change); and 3) the fact that the people who benefit from different ecosystems have different capacities to adapt and to respond to change.

The following list of criteria reflect one important set of factors: an ecosystem's ability to generate future values under current landscape conditions. To introduce a more dynamic perspective, they are placed alongside other criteria that reflect differences in the costs and risks associated with the differing effects of change on different ecosystems.

QUALITY

- Capacity (Do the features of this wetland allow it to trap nutrients?)
- Opportunity (Is it in a location where nutrients will be available to trap?)
- Payoff (What resources are protected by trapping nutrients at this site?)
- Equity (Who gains by having nutrients trapped here rather than elsewhere?)

SCARCITY—The relationship between current supply and expected demand for services associated with this type of resource.

Critical factors: *Abundance* and *availability* of the resource
Critical questions:
- What is the quantity and quality of existing assets?
- What are the trends in supply and demand for services of these assets?

VULNERABILITY—The extent to which this type of resource is susceptible to being lost to various types of controllable risks (e.g., land use patterns) and uncontrollable risks (e.g., storm surge, sea level rise).

Critical factors: *Exposure* and *sensitivity*
Critical questions:
- Is resource at this site likely to be lost anyway making it *less* valuable?
- Are other similar sites likely to be lost making this one *more* valuable?

REVERSIBILITY—The scientific, technical, and economic feasibility of restoring and rehabilitating this type of resource.

Critical factors: Where we are on the learning curve for restoration technologies that can be applied to this resource.
Critical questions:
- Do we know how to restore this resource if we misjudge threats or reconsider priorities later?
- Can restoration depend on natural processes or does it require engineered solutions?
- What would restoration cost , how long would it take, and what are the risks?

REPLACEABILITY—The capacity to produce perfect or near-perfect substitutes for this resource or the services of this resource.

Critical factors:
- Scarcity of resource at broader ecological scale (e.g., outside the watershed).
- Where we are on the "learning curve" for restoration technologies that can be applied to this resource (e.g., fish spawning habitat) or artificial substitutes (e.g., hatcheries).

Critical questions:
- Do we know how to replace this resource if we misjudge threats or reconsider priorities later?
- Can replacement depend on natural processes or require engineered solutions?
- What would replacement cost, how long would it take, and what are the risks?

SUBSTITUTABILITY—The capacity to find imperfect but acceptable substitutes for the services of this resource. (e.g., lake fishing vs. ocean fishing opportunities)

Critical factors: The uniqueness of this resource and the abundance and availability of similar resources or services of comparable value at a reasonable cost

Critical questions:
- What substitutes are available for the services of this resource?
- What are the differences in quality, costs, and who has access to them?

Developing Actual Indicators

This section develops general criteria for specifying leading indicators of ecosystem values, but stops short of recommending specific indicators. This is because the appropriate focus, scale, precision, frequency, and reliability of indicators depends on circumstances. The following section illustrates why an indicator system based on the criteria outlined in this section, or something very similar, will be essential to settle natural resource damage claims related to oil spills under a 1996 ruling related to the settlement of damage claims under the Oil Pollution Act of 1990 (OPA). This provides a useful context for exploring applications of the types of indicators being considered here for two reasons. First, the 1996 OPA ruling requires comparisons of injured and restored ecosystems based on services and values, not just "functional equivalency." Second, the economic stakes involved in oil spills litigation are high enough, and the frequency of spills is low enough, that the development and testing of site-specific indicators to form a basis for comparing losses from injuries with gains from restoration should be worthwhile.

A separate paper based on this one provides a preliminary profile of twenty-eight of the most widely used ecosystem assessment methods and evaluates their potential usefulness as a basis for developing indicators of capacity, opportunity, payoff, and equity with respect to various ecosystem functions.[28]

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[28] This preliminary evaluation of ecosystem assessment methods is presented in King (1997). More detailed evaluations dealing with the development specific indicators and with respect to specific applications under OPA are forthcoming.

SECTION 3

APPLICATIONS UNDER THE OIL POLLUTION ACT OF 1990

Background

The Oil Pollution Act of 1990 (OPA) authorizes public trustees—federal and state governments and some Native American tribes—to seek recovery of damages for injuries to natural resources. The goal of OPA is "to make the environment and the public whole for injuries to natural resources and natural resource services resulting from an incident involving a discharge or substantial threat of discharge of oil."[29] Under a final rule issued in January 1996, this goal is to be achieved "through returning injured natural resources and services to baseline and compensating for interim losses of such natural resources and services through the restoration, rehabilitation, replacement or acquisition of equivalent natural resources and/or services."[30] Efforts to restore the injured resource to baseline conditions are referred to in the 1996 rule as "primary" restoration. Restoration undertaken to compensate the public for interim lost services is referred to as "compensatory" restoration.

The 1996 final rule specifies that the adequacy of restoration should be determined by the following process:

1) *Compare* the type and quality of lost resources and services with those that result from primary and compensatory restoration,

2) *Determine* if they are of *comparable value*; and, if they are not, and

3) *Implement* scaling procedures that adjust the level of restoration to a size that will compensate the public for the injury.

Guidance for Ecosystem Comparisons

The rule does not provide specific guidance regarding how the type, quality, and comparable value of lost and replacement resources or services should be compared, or how "scaling" should be accomplished. However, under certain circumstances, it does recommend using specific types of analyses. According to the rule, "when the injured resources and/or services are primarily of indirect human use (e.g., species habitat or biological natural resources for which human uses are primarily off-site) the appropriate basis for evaluating and scaling the restoration is *Habitat Equivalency Analysis* (HEA)."

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[29] "Section 1006 (e)(1) of the Oil Pollution Act of 1990 requires the President, through the Under Secretary of Commerce for Oceans and Atmosphere, to promulgate regulations for the assessment of natural resource damages resulting from a discharge or substantial threat of a discharge of oil" (Federal Register, 1996).

[30] "The final rule is for the use of authorized federal, state, Indian tribe, and foreign officials, referred to as 'trustees.' Natural resource damage assessments are not identical to response or remedial actions addressed by the larger statutory scheme of the Oil Pollution Act of 1990. Assessments are not intended to replace response actions, which have as their primary purpose the protection of human health, but to supplement them, by providing a process for restoring natural resources and services injured as a result of an incident involving oil" (Federal Register, 1996).

Federal agencies have prepared draft guidance for implementing "scaling" and completing some other tasks required by HEA.[31] The classification system used in these draft guidance documents identifies four specific types of comparisons that may need to be made. They include:

TYPE 1: same type, same quality, and comparable value;
TYPE 2: same type, same or different quality, and *not* of comparable value;
TYPE 3: different type (and therefore, of different quality and not of comparable value), but of comparable type and quality, nonetheless; and
TYPE 4: different type (and therefore, of different quality and not of comparable value) and not of comparable type and quality.

Applying Valuation Criteria

The draft guidance documents do not discuss specific measurement tools, but it is inferred in these documents that some combination of ecosystem assessment methods and ecosystem valuation methods will provide an analytical basis for performing HEA and for addressing issues related to "scaling." Section 1 of this paper provided evidence that neither methods provide an adequate basis for comparing ecosystem services or values. The implication is that they cannot, by themselves, provide an analytical basis for conducting HEA. Section 2 of this paper outlined criteria for developing sets of indicators to compare ecosystems in terms of their relative values. The implication was that such indicators might provide a practical alternative to conventional ecosystem assessment and valuation methods; and would, in any case, provide a way to link the results of those methods for purposes of comparing ecosystem services and values. This section gives the general framework for conducting HEA, and describes why an indicator system similar to the one discussed in the previous section will be needed to carry out HEA and to address "scaling" issues.

The Basics of Habitat Equivalency Analysis (HEA)

Consider the simple Type 1 situation listed above where the lost and restored habitats are the same type and the same quality and are of comparable value. Assume that habitat type under OPA can be defined adequately using the widely accepted Cowardin system for classifying wetlands and deep water habitats shown in Figure 4.[32] Habitats within each Cowardin classification—each habitat type under OPA—can be expected to have similar capacity to provide the functions listed in Figure 1.

Habitats of a given type may have vastly different biophysical characteristics and landscape contexts and, as a result, may have vastly different functions, services, or values. However, for simplicity, assume that they are identical and, also, that a linear relationship exists between measures of the functional capacity of wetlands and the functions it will provide and between these and the services and values it is expected to generate. Under these simplifying assumptions a percentage change in the functional capacity of habitat, as a result of restoration for example, can be expected to result in an identical percentage change in the functions, services, and values provided by the site. These assumptions greatly simplify the task of comparing ecosystems, of course, because functional capacity is relatively easy to measure and

[31] See Damage Assessment and Restoration Program (1996).

[32] The widely used Cowardin system of wetland classification includes five types of wetland systems, 11 types of wetland subsystems, and 55 classes of wetlands (Cowardin, Lewis, et al. 1979).

responds directly to restoration, whereas the others are relatively difficult to measure, and may not respond directly to restoration efforts.[33]

Graphical Depiction of HEA

Under these simplifying assumptions, relative measures of functional capacity can serve as the "currency" for comparing ecosystem values. The values lost at the site of the injury (which is also the primary restoration site) can be expressed in relative terms using the pre-injury capacity of the site as the baseline (100%) (this is depicted in Figure 5a). Likewise, the gains in values at the compensatory restoration site can be measured in relative terms using the pre-restoration capacity of the site as a baseline (this is depicted in Figure 5b). In both Figure 5a and Figure 5b the level of functional capacity is expressed in each period as a percent of the baseline level of capacity, which is shown to increase over time after the injury and as a result of primary and compensatory restoration. At both the site of the injury and the compensatory restoration site the values generated each year on a per acre basis can be expressed as "the percent of baseline value." However, for the relative units of value to be comparable—for the scale of the "y" axis to be the same for both sites—two additional conditions must exist. The baseline functional capacity and the fixed proportions between functions, services, and values assumed for the injured site must be identical at the compensatory restoration site.[34]

Criteria for Establishing Type 1 Conditions

The first place where the criteria developed in the previous section might be useful would be in determining whether or not a Type 1 situation exists. Based on the logic used in the previous section, determining if the site of the injury and the site of proposed restoration are of similar quality and of comparable value will require comparisons of site-based and landscape-based criteria. Using the terminology developed in the previous section this could be established on the basis of whether or not the two sites have the same capacity and opportunity to provide a certain mix of functions and whether these functions will result in the same levels of services.

Type 2, Type 3, and Type 4 Situations

As a practical matter, HEA will need to be performed most often to compare habitats that are not of the same quality and comparable value: Type 2, 3, or 4 situations.[35] Under OPA, the process of "scaling" is intended to equate units (acres) of restored habitat to the quality and value standard of units (acres) of injured habitat.[36] The process of "scaling" can become very

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[33] These are the assumptions that are used implicitly whenever differences in the biophysical characteristics of ecosystems (measures of functional capacity) are used to compare ecosystem values.

[34] These are the conditions necessary to conduct HEA using relative measures of restoration success.

[35] Until recently wetland mitigation provisions under Section 404 of the Clean Water Act included a clear preference for "on-site, in-kind" mitigation. This closely resembles the Type 1 situation identified under OPA. This preference, however, resulted in many restoration projects being undertaken at inferior sites and failing to meet environmental goals while diverting restoration funds away from more promising "off-site" projects. As a result, the preference for on-site mitigation has softened and off-site mitigation, including mitigation banking, has gained more acceptance.

[36] "Scaling" under OPA involves adjusting the size of compensatory restoration projects to account for differences in the quality and value of habitat at the injured and restoration sites. Scaling can be based on resource-to-resource, service-to-service, or value-to-value

complicated unless specific criteria are established as a basis for comparing ecosystem quality and value. As simplifying assumptions about fixed relationships between site capacity and functions, services, and values are relaxed to reflect real-world conditions, the process of conducting scaling without some type of indicator system may become impossible.

The first problem is one that would be encountered in comparing lost and replacement services and values even in Type 1 situations. Since compensatory restoration projects will be undertaken sometime after the injury, they are likely to provide replacement services and values sometime after those associated with the injured site are lost. Adjusting for differences in the timing of services lost and gained can be accomplished by introducing the concept of discounting into HEA; this has been described fully elsewhere.[37] What is more difficult is determining what unit of measurement the discount factor should be applied to.

The first complication is that differences in quality and value of habitats can be assessed with respect to more than one set of services and values (e.g., fish vs. waterfowl habitat, recreational vs. aesthetic value). Undertaking the process of "scaling" to account for each of the ecosystem functions listed in Figure 1 will be quite challenging unless some prearranged criteria or indicator system is used to compare site and landscape conditions with respect to each functions. Using this approach the scaling factors may be different for different functions and services.[38] Moreover, a mediocre habitat in an exceptionally good location (moderate capacity, high opportunity) may generate the same expected level of service (quality) and comparable value as high quality habitat in a poor location (high capacity, low opportunity).

In the Final Analysis

In the final analysis, HEA requires that the discounted present value of the expected interim loss of service flows at the site of the injury (Area X in Figure 5a) must equal the expected increase in the discounted present value of service flows provided over time at the compensatory restoration site (Area Y in Figure 5b). Under very simple assumptions it might be possible to use differences in measures of capacity at the two sites to determine an appropriate "scaling" factor. However, under more realistic conditions all of the issues raised in earlier sections of this paper come into play. In the typical situation, assuming that two sites are identical and in identical landscape contexts, what criteria should be used to conduct HEA and perform scaling? The criteria outlined in the previous section that allows ecosystems to be compared and ranked on the basis of differences in biophysical characteristics and landscape contexts is one way. If an indicator system based on these or similar criteria are not used to develop "scaling" factors, how can alternative compensatory restoration sites be compared, and how can the adequacy of investments in primary and compensatory restoration be evaluated? In practice, scaling restoration projects cannot take place on the basis of service-to-service or value-to-value comparisons but only on the basis of comparisons of *expected* services and values. That is the purpose of the criteria developed in the previous section: to form a basis for making resource-to-resource comparisons.

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comparisons and usually takes the form of adjusting the size or intensity of restoration projects. The process of "scaling" under OPA is described in draft guidance documents prepared by the NOAA Damage Assessment and Restoration Program (1996).

[37] For a discussion of how discounting affects the comparison of ecosystem services and values provided at different times see King (1991), King, Bohlen, and Adler (1993) and Unsworth and Bishop (1994).

[38] Perhaps some type of weighting criteria applied to functions and services on the basis of other factors and abundances and shortages identified in specific landscape contexts could provide an unambiguous basis for scaling restoration at various sites.

Conclusions about HEA and Scaling

The criteria outlined in the previous section raise more questions about performing HEA and scaling than they answers. Under the simplifying assumptions listed above to characterize Type 1 situations, HEA and scaling can be accomplished fairly easily by comparing appropriately discounted measures of capacity to represent the success of natural recovery and restoration efforts. However, under more realistic situations there are four complicating factors that need to be addressed.

First, and most important, is the enormous effect of landscape context on the functions, services, and values generated by an ecosystem. Where the compensatory restoration being considered takes place off-site, the typical situation under OPA, differences in opportunity, payoff, and equity come into play and need to be factored into "scaling" procedures. Although the effect of location may be negligible in the case of restoration that is strictly "on-site," opportunities for strictly "on-site" restoration will be rare. Even restoration sites that are identical in physical features to the injured site and are close enough to the injured site to be considered "on-site"(a distance of less than 1/2 mile for instance) may generate significantly different services and values.

Second, because there will usually be more than one habitat function to consider, using scaling procedures may yield ambiguous results. Performing HEA and applying "scaling" procedures to individual functions, services, or values may yield conflicting results. Some weighting or ranking of functions or services may be needed to make tradeoffs when applying HEA, especially if several different compensatory restoration possibilities are being considered.

Third, both HEA and scaling require forecasting flows and services from injured and restored habitats relatively far into the future. The uncertainty associated with the outcome of habitat restoration projects is well-known and is generally higher than the uncertainty associated with the service flows of undisturbed natural habitats or naturally recovering habitats. This uncertainty has several identifiable sources. For example, the restoration plan itself may be flawed, sound restoration plans may not be implemented properly, unexpected natural events (e.g., storms, droughts, sea-level rise) or man-made events (e.g. accidents, land development) may intervene, and so on. Differences in risks and uncertainty and their effects on the "expected values" of service flows are not the same for all functions and services or at all prospective restoration sites. More importantly, they may not be the same at restoration sites as they are at the site of the injury. Differences in risk and uncertainty associated with expected streams of services and values over time need to be factored into HEA and "scaling" procedures.

Finally, there are the logical bounds of achieving equivalent functions, services and values by "scaling" the size and intensity of compensatory restoration projects. A huge quantity of low quality habitat may not be equivalent to a small quantity of high quality habitat. Similarly, the difference in the expected recovery of functions and services with and without investments in restoration may not justify the level of investment necessary to undertake compensatory restoration at the preferred (on-site) location. Stated differently, investments at the extensive margin (more area restored) may not be a reasonable substitute for investments at the intensive margin (more restoration per unit area) and requiring either investment on-site may represent a wasteful use of restoration dollars after more promising off-site alternatives are considered.

Conclusions about Indicators and OPA

Section 1 of this paper described how ecosystems generate services and values and identified the factors that influence the level and distribution of those services and values. The conclusion reached on the basis of Section 1 is that the available ecosystem assessment methods and ecosystem valuation methods provide only part of the information required to compare ecosystems in terms of the services and values they can be expected to provide.

Section 2 developed a preliminary framework and identified criteria for developing "leading indicators" of ecosystem services and values. The conclusion reached in Section 2 was that it should be possible to integrate the results of ecosystem assessment and valuation studies with information about landscape context in a practical framework for comparing ecosystems on the basis of expected services and values.

This final section reviewed some of the requirements for conducting HEA and performing scaling procedures under OPA. The conclusion reached here is that performing HEA and scaling requires assessing differences in the expected services and values from different ecosystems. As a practical matter these differences need to be assessed on the basis of observable or measurable differences in site and landscape characteristics that influence future services and values. These do not necessarily require the estimation or comparison of current services or values.

Since ecosystem comparisons under OPA must determine if compensatory restoration makes "the environment and the public whole for injuries to natural resources and natural resource services," the use of conventional biophysical ecosystem assessment methods, by themselves, are not adequate. On the other hand, ecosystem valuation methods are too expensive to be applied to all ecosystems services at all prospective restoration sites and do not link expected values with specific ecosystem characteristics that change as a result of injuries and restoration efforts. These limitations seem to justify the further development and testing of value-based criteria that draw on both ecosystem assessment and ecosystem valuation methodologies to form a practical analytical basis for performing HEA and scaling.

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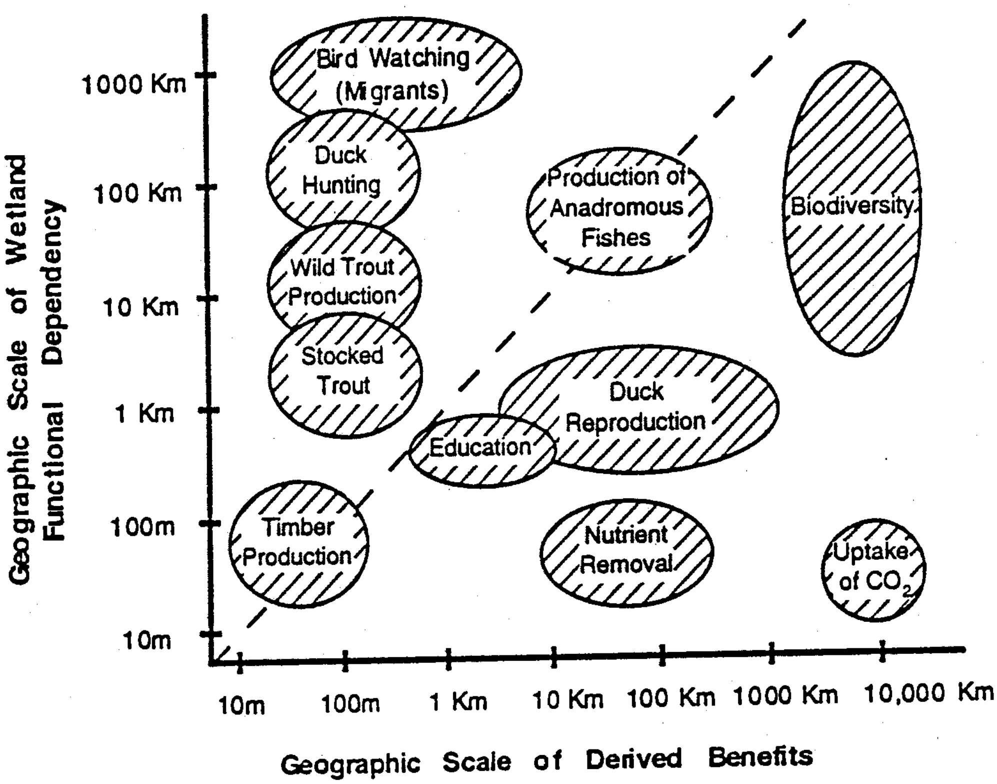
FIGURE 1

ECOSYSTEM FUNCTIONS AND ILLUSTRATIONS OF ASSOCIATED SERVICES

Ref. #	ECOSYSTEM FUNCTIONS	ASSOCIATED TYPES OF SERVICES/VALUES
1	Fishery Habitat	• Better com./rec. fishing, lower fish prices, improved int'l trade balance.
2	Waterfowl Habitat	• Better hunting and bird watching on-site, nearby, and elsewhere.
3	Fur-bearer Habitat	• Commercial and recreational opportunities
4	Storehouse of Biodiversity (onsite species diversity)	• Direct, indirect, serendipity value of scientific research, medical discoveries, genetic pools, seed banks, etc.
5	Food Chain/Biodiversity Support (offsite species support)	• Same as 4, except off-site
6	Natural Products (e.g., timber, hay, cranberries, peat)	• Wholesale and retail market value and associated jobs, incomes
7	Groundwater Recharge/Discharge	• Drinking water quality, reduced human and environmental health risks
8	Floodwater Storage, Conveyance and/or Desynchronization	• Reduced soil erosion and property damage
9	Shoreline Anchoring/Erosion Control	• Protection of beaches, private property, infrastructure, ecosystem
10	Storm Surge/Wave Protection	• Reduced soil erosion and property damage
11	Sediment Trapping	• Protects aquatic ecosystems, reduced dredging, maintains hydropower
12	Pollution Assimilation	• Reduced treatment costs, improved public health and environment
13	Nutrient Retention/Filtering	• Maintain nitrogen balance, prevent algae blooms and anoxic conditions.
14	Natural Area/Open Space	• Active and passive recreation, research, teaching/learning (K thru ...) general aesthetics, spiritual enrichment, heritage
15	Micro-climate Regulation	• General life support; ill-defined but important local/regional linkages
16	Macro-climate Regulation	• General life support; ill-defined but important national/global linkages
17	Carbon Cycling	• General life support; ill-defined but important national/global linkages

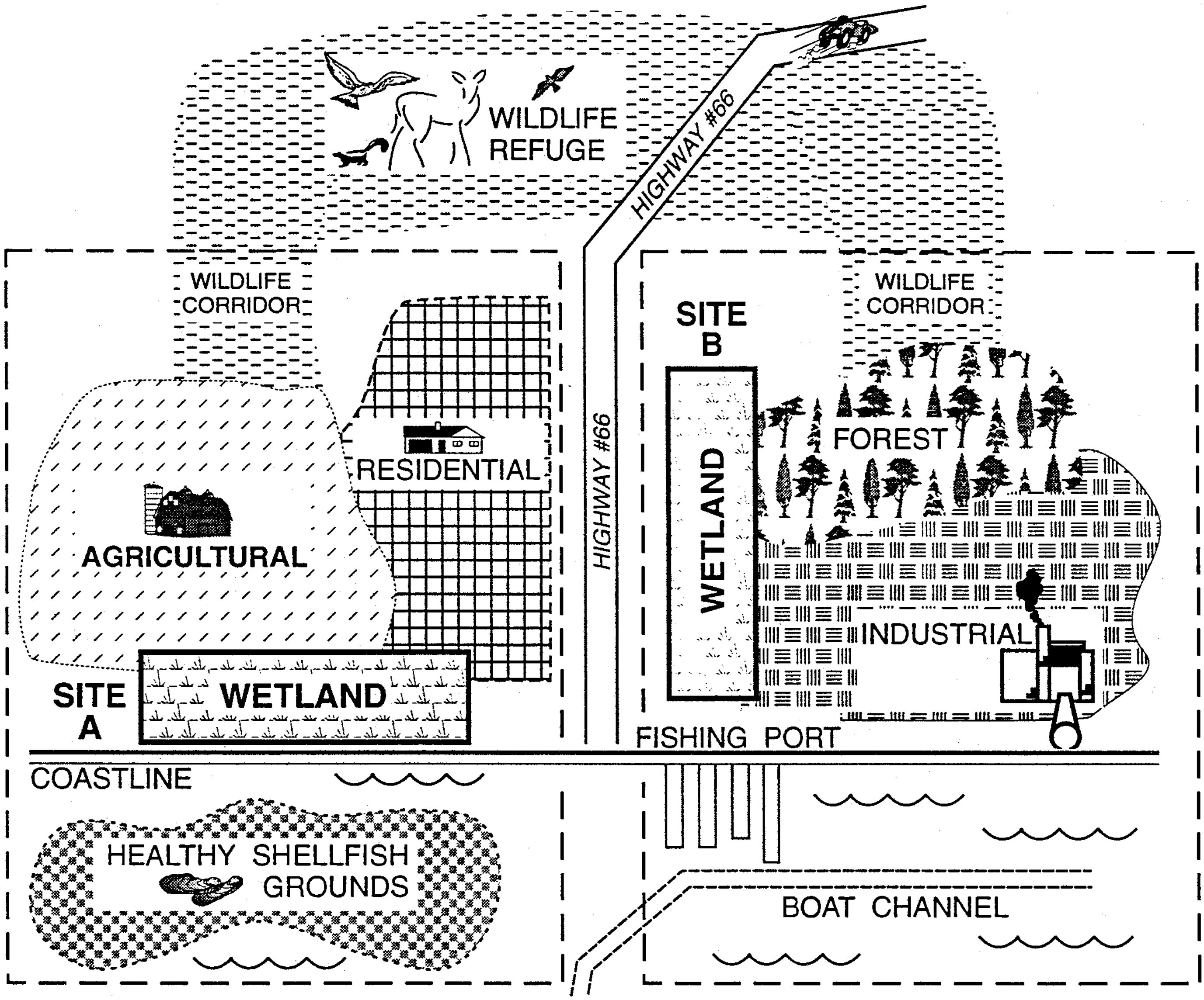
FIGURE 2

LANDSCAPE CONTEXT OF WETLAND FUNCTIONS*



*These are illustrative ranges only, and are based on extrapolations of "typical" dependencies and derived benefits. They show how far from the wetland site benefits may accrue (x-axis), and how far from the site features of the natural landscape have influence (y-axis).

Figure 3 a.
Effects of Wetland Location on Function, Service and Value



Site Characteristics

Wetland Site A and Wetland Site B are identical in size, shape and bio-physical characteristics and are located in the same sub-watershed on either side of Highway 66.

Landscape Context

SITE A
› near the coast, downstream is a beach area
› adjacent to large healthy shellfish grounds that are accessible to the community
› upslope is agricultural land (nutrient runoff)
› wildlife corridor open from the North
› near residential areas (aesthetics, scenic)
› good access, adjacent public lands
› access to many urban poor people

SITE B
› slightly off coast, downstream is industrial site
› adjacent to fishing port and small shellfish beds that are contaminated and remote
› upslope is forest (no nutrient runoff)
› wildlife corridor is blocked by Highway 66
› nearby industrial sites (no proximity to people)
› poor access, surrounded by private lands
›access to few suburban rich people

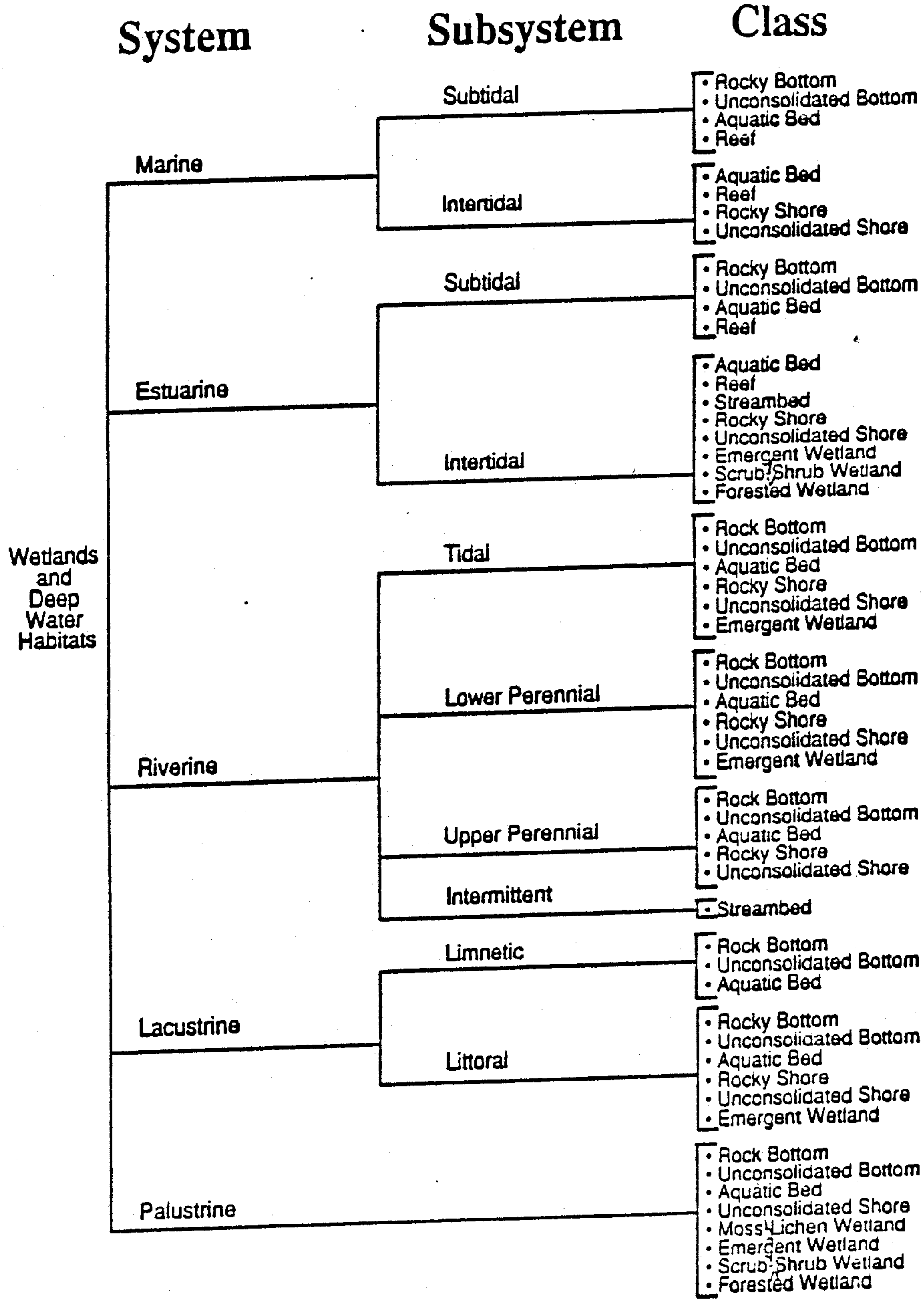
FIGURE 3b

Relative Values of Site A and Site B

	WILDLIFE HABITAT SITE A	WILDLIFE HABITAT SITE B	FISHERY SUPPORT SITE A	FISHERY SUPPORT SITE B	NUTRIENT TRAPPING SITE A	NUTRIENT TRAPPING SITE B
CAPACITY	Identical size, shape, bio-physical characteristics Score: Medium	Identical size, shape, bio-physical characteristics Score: Medium	Identical size, shape, bio-physical characteristics Score: Medium	Identical size, shape, bio-physical characteristics Score: Medium	Identical size, shape, bio-physical characteristics Score: Medium	Identical size, shape, bio-physical characteristics Score: Medium
OPPORTUNITY	Wildlife Corridor <u>open</u> from the North Score: High	Wildlife Corridor <u>blocked</u> from North by Highway 66 Score: Low	Traps agricultural sediment, near coast, adjacent to fishing grounds Score: High	Little sediment to trap, off the coast, adjacent to boat channel Score: Low	Upslope is farm land generating nutrient flow, natural water flow, non-point discharge Score: High	Upslope is industrial sites and forests (little nutrients) channelized water flow to point discharge Score: Low
PAYOFF	Near residential areas, accessible, public land Score: High	Surrounded by industrial sites, inaccessible, private land Score: Low	Adjacent to large healthy shellfish area, public access, nearby parking Score: High	Few shellfish nearby, little access if there were, near point source discharge Score: Low	Adjacent to large healthy shellfish area, public access, nearby parking Score: High	Few shellfish nearby, little access if there were, near point source discharge Score: Low
EQUITY	Aesthetic, recreational opportunities for many poor Score: High	Access limited to few rich Score: Low	Aesthetic recreational opportunities for many poor Score: High	Access limited to few rich Score: Low	General water quality improvements, but also adjacent to healthy shellfish area accessable to poor Score: High	General water quality improvements only Score: Medium
OVERALL RATING	**High**	**Low**	**High**	**Low**	**High**	**Low**

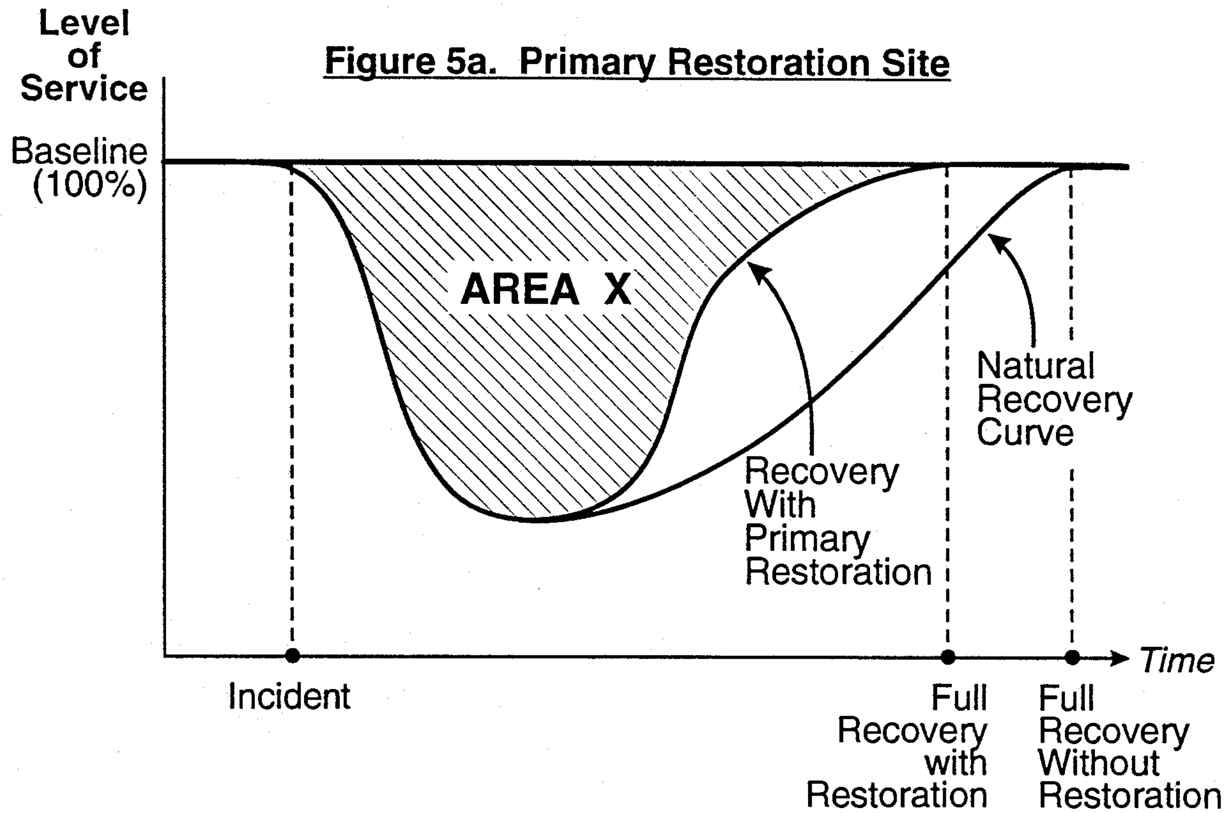
FIGURE 4

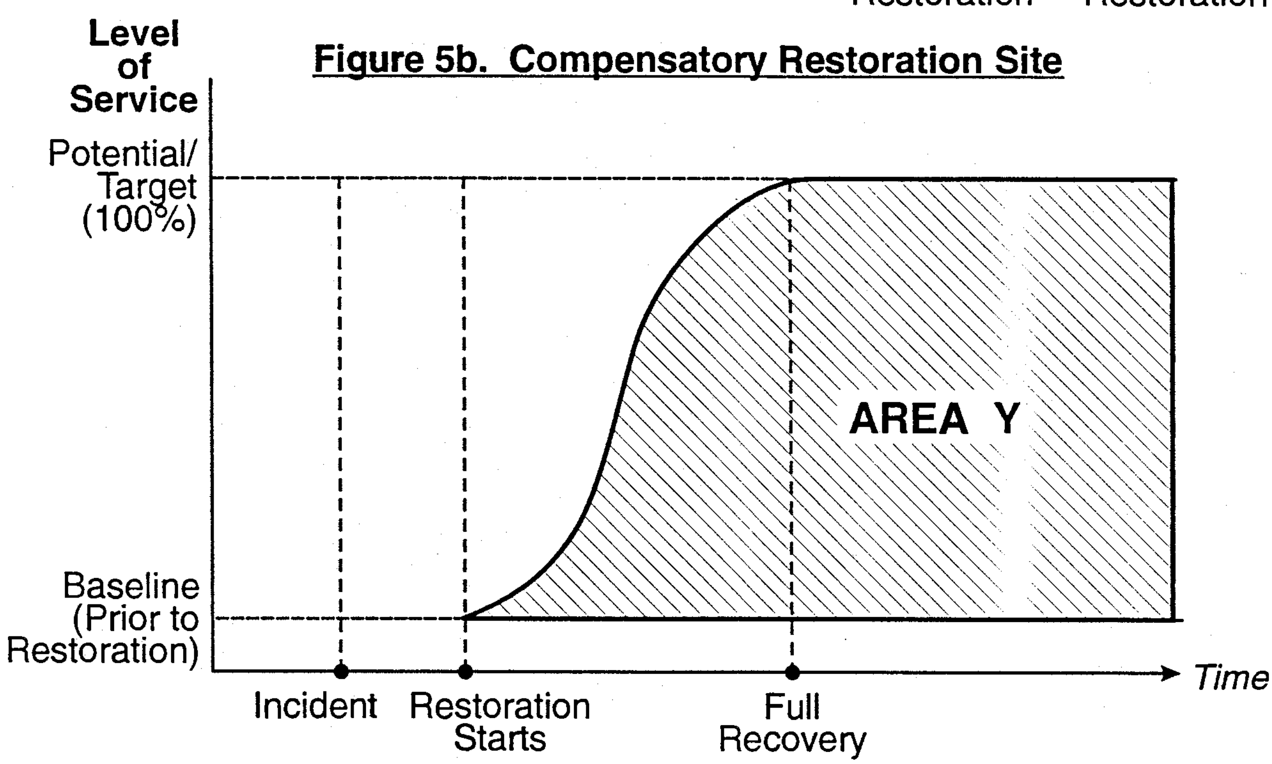
OVERVIEW OF THE COWARDIN SYSTEM FOR CLASSIFYING WETLANDS AND DEEP WATER HABITATS*



*The Cowardin wetland classification system is widely used in the U.S. It defines 5 wetlands types, 11 wetland subsystems, and 55 classes of wetlands and deep water habitats and is described in Cowardin, Lewis M., et al., Classification of Wetlands and Deep Water Habitats of the United States, FWS/OBS79?31, Office of Biological Services, Fish and Wildlife Service, U.S. Government Printing Office, Washington, D.C. 1979.

Figure 5.
Relationship between Service Flows at Primary and Compensatory Restoration Sites





AREA X: Services **lost** at injury site with Primary Restoration expressed as % of baseline or expected level

AREA Y: Services **gained** at Compensatory Restoration Site expressed as % of potential/target level less baseline (pre-restoration) %