Wetland Value Indicators for Scoring Mitigation Trades

Lisa A. Wainger,* Dennis King,**
James Salzman,*** James Boyd****

* Assistant Research Scientist, University of Maryland Center for Environmental Science, Solomons, MD; email: wainger@cbl.umces.edu. This research was supported by the United States Environmental Protection Agency's (EPA's) Science to Achieve Results (STAR) program grant R82612-01 and used data and analysis from previous work funded by the US Army Corps of Engineers, Waterways Experiment Station. Because this article has not been subjected to any EPA review and does not necessarily reflect the views of the Agency, no official endorsement should be inferred. A more detailed treatment of these issues (with complete data and graphs) will be available in the final research report and is currently available as a manuscript from the authors.

** Senior Research Scientist, University of Maryland Center for Environmental Science, Solomons, MD.

*** Professor, Washington College of Law, American University.

**** Senior Fellow, Resources for the Future.

I. Introduction

Over the last decade, mitigation banking has become an increasingly popular market-based solution for resolving conflicts over wetland development. Wetland mitigation banking permits developers, once they have taken steps to avoid and minimize wetland loss, to compensate for wetlands losses during land development by ensuring wetland restoration elsewhere. Under this banking mechanism, land developers must either purchase credits from specific mitigation banks or pay into "in-lieu fee" trust funds in order to receive permits to alter wetlands. Currently in the United States, at least 230 wetland mitigation banks (in various

stages of development) either are or will soon be involved in trades of existing wetlands for restored or preserved wetlands.[1]

As described by Ruhl and Gregg,[2] prior to 1990 there was a clear regulatory preference for on-site wetland mitigation, and off-site mitigation trades were rare. Over the past decade, the balance has shifted in favor of mitigation banking and restoration of wetlands distant from the destroyed wetlands. Mitigation banking was expected to improve the success of wetland mitigation efforts and make it easier for wetland regulators to achieve no net loss of wetland functions and values. Several recent national and state reviews of mitigation banking have concluded, however, that while the concept of wetland mitigation trading is sound, the practice of mitigation banking often fails to provide wetland gains that offset wetland losses.[3]

There are two major reasons for this failure. First, unavoidable technical challenges can limit the effectiveness of wetland restoration.[4] Second, the methods for calculating the units of exchange (credits) have proven inadequate because they fail to sufficiently address differences in the value of wetland services provided at different locations; they do not adequately incorporate risks or fully assign them to mitigation buyers or sellers; and they do not provide mitigation trade regulators with the analytical basis to defend their scoring methods against technical, legal, or political challenges. In other words, we have not yet found a way to determine, with reasonable certainty, whether and when "trading" existing wetlands in one location for restored wetlands in another promotes social welfare.

Two general approaches are used for comparing wetlands and determining the "mitigation ratio"—that is, the number of acres that need to be created (or the number of credits that need to be

1. U.S. CORPS OF ENG'RS, INSTITUTE FOR WATER RESOURCES, EXISTING WETLAND MITIGATION BANK INVENTORY, *at* http://www.iwr.usace.army.mil/ iwr/regulatory/regulintro.htm (visited Apr. 24, 2001).

2. J. B. Ruhl & R. Juge Gregg, *Integrating Ecosystem Services into Environmental Law: A Case Study of Wetlands Mitigation Banking*, 20 STAN. ENVTL. L.J. 365, 369-70 (2001).

3. *See, e.g.*, DEP'T OF ENVTL PROT. & THE WATER MGMT. DISTRICTS, FLORIDA OFFICE OF PROGRAM POLICY ANALYSIS AND GOV'T ACCOUNTABILITY, WETLAND MITIGATION 99-40 (2000); Phillip H. Brown & Christopher L. Lant, *The Effect of Wetland Mitigation Banking on the Achievement of No-Net-Loss* 23 ENVTL. MGMT. 333-45 (1999).

4. *See* Teresa K. Magee et al., *Floristic Comparison of Freshwater Wetlands in an Urbanizing Environment* 19 WETLANDS 3, 517-34 (1999) (indicating that after five years at recently restored sites more than 50% of the plant species present were invasive species, and thus sites were not providing the habitat and functions typical of the region).

purchased from a bank) for every wetland acre (or credit-equivalent) destroyed at the development site.[5] Most commonly, mitigation ratios are established using biophysically based wetland assessment methods. Less commonly, but particularly in permit disputes, wetlands are compared using economic valuation methods. Wetland assessment methods focus on measuring the physical, chemical, and biological structure of wetlands, and on their resulting ability to provide the natural functions that are generated by their interactions.[6] The biophysical focus of traditional wetland

5. There are several methods used to mitigate wetland destruction, including wetland creation, preservation, restoration, or enhancement. Each method will typically have its own mitigation ratio.

6. Numerous *wetland assessment* procedures have been developed over the past few decades. *See* CANDY C. BARTOLDUS, COMPREHENSIVE REVIEW OF WETLAND ASSESSMENT PROCEDURES: A GUIDE FOR WETLAND PRACTIONERS (1999) (providing an overview of thirty wetland assessment methods); *see also* WORLD WILDLIFE FUND, STATEWIDE WETLAND STRATEGIES: A GUIDE TO PROTECTING AND MANAGING THE RESOURCE (1992).

The Habitat Evaluation Procedures (HEP), developed in 1980, was one of the first and most comprehensive attempts to show that wetlands provide services beyond those associated with recreation or land market values. U.S. FISH & WILDLIFE SERVICES, DEPT. OF THE INTERIOR, HABITAT EVALUATION PROCEDURES 102 (1980). HEP is still a widely used method for establishing non-monetary currencies of habitat value. However, HEP focuses primarily on site characteristics that satisfy the needs and preferences of wildlife species (e.g., breeding and feeding conditions), not on site and landscape characteristics that determine how improved habitats are likely to satisfy the needs and preferences of people. A significant amount of conceptual work went into the development of a component of HEP called the "human use and economic evaluation" (HUEE) module, which does incorporate human values. However, the concepts underlying HUEE were never fully developed or field tested; and, unlike the rest of the HEP method, the HUEE module has not been widely used. *See* U.S. FISH & WILDLIFE SERVICES, HUMAN USE AND ECONOMIC EVALUATION HANDBOOK (1985), *available at* http://policy.fws.gov/872fw1.html (visited Apr. 23, 2001).

Numerous wetlands assessment procedures have been developed since HEP. Some attempt to address wetland values based on the presence or absence of notable features, such as endangered species or designated historic or archeological areas. *E.g.*, CANDY C. BARTOLDUS, ET AL., EVALUATION FOR PLANNED WETLANDS App. (1994); ANNA L. HICKS, NEW ENGLAND FRESHWATER WETLANDS INVERTEBRATE BIOMONITORING PROTOCOL (1997); Ted T. Cable, et al., *Simplified method for wetland habitat assessment*, 13 ENVTL MGMT. 207 (1989). A few procedures include simple models or questions that are used to assign scores to wetlands based on social categories such as recreation, aesthetics, agricultural potential, and educational values. *E.g.*, MINN. BD. OF WATER AND SOIL RESOURCES, MINNESOTA ROUTINE ASSESSMENT METHOD FOR EVALUATING WETLAND FUNCTIONS, DRAFT VERSION 2.0 (1998); EMILY ROTH ET AL., OREGON FRESHWATER WETLAND ASSESSMENT METHODOLOGY (1996); Alan P. Ammann, et al., *Method for the Evaluation of Inland Wetlands in Connecticut* (Conn. Dep't of Envtl. Protection Bull. No. 9, 1986); Alan P. Ammann & Amanda L. Stone, *Method for the Comparative Evaluation of Nontidal Wetlands in New Hampshire* (N. H. Dep't of Envtl Services NHDES-WRD-1991-3, 1991); G.G. Hollands & D. W. Magee, *A Method for Assessing the Functions of Wetlands, in* PROCEEDINGS OF THE NATIONAL WETLAND ASSESSMENT SYMPOSIUM 108 (J. Kusler & P. Riexinger eds., 1985). Some of them also weave concepts of function and value into a measure called "functional value." *See, e.g.*, Ammann & Stone. However, the criteria

evaluation has been reinforced, and partly driven, by regulatory requirements.

While biophysical measures may describe the capacity of a site to provide services, they reflect nothing about the actual delivery and the relative value[7] of those services at a given location, as would be required for an objective evaluation of how scarce resources may best be allocated. Relating a site's biophysical features (e.g., dominant plant type) and functions (e.g., nutrient trapping) to its value requires considering aspects of its location. The surrounding environs may enhance or detract from a wetland's ability to provide valued services depending on landscape configuration. For example, a site may gain value when the landscape contains complementary features such as nearby parkland that allows access by humans, protects the wetland from effects of land use change, or allows it to serve as a wildlife corridor. Wetland valuation requires considering these and other aspects of landscape context that allow functions to become valued services.

In contrast, economic valuation methods are clearly aimed at determining wetland value in terms that do provide the necessary inputs for making resource trade-offs, and they do consider the context of the wetland's neighborhood, if not the full landscape context. However, these methods suffer from some significant weaknesses: namely their narrow focus, high cost, and methodological impenetrability to non-economists. Because many ecosystem services are non-market public goods—accruing to the public rather than the wetland owner—the bulk of wetland value must be measured with non-market valuation techniques.[8] These types of

for assigning relative values to different wetlands or distinguishing between levels of function and associated values are not clearly defined in any of these methods.

7. "Value" in this case includes the value of marketed and non-marketed goods and services that result from wetland functions including active uses such as resource extraction or recreational activities, and passive uses such as spiritual enrichment or existence values.

8. There are three general *economic valuation* approaches to estimating the economic value of wetland services that are not traded in markets. People can *reveal* the dollar value they place on some services by their purchasing decisions. People can *express* the dollar value they place on some services through surveys. And people's willingness to pay for some services can be *imputed* based on the costs they would incur if the services were not provided (e.g., the cost to provide flood control if wetlands were not present).

A variety of methods are employed to estimate these values. In contingent valuation, surveys are used to directly solicit people's "willingness to pay" for specific environmental services, based on a hypothetical scenario. In the travel cost method, it is assumed that the value of a site is reflected in how much people are willing to pay to travel to visit the site. Hedonic pricing is used to estimate economic values for ecological characteristics that di-

valuation studies typically attempt to assign values to wetland services (e.g., provision of recreational opportunities) in absolute (dollar) terms without much regard for the specific wetland features or functions generating the value.[9] Many of these studies depend on questionnaires or individual interviews in which respondents are asked to place a dollar value on a single ecosystem service. In other cases, a wetland's value may be assessed by asking tourists what they spend while visiting a site in order to estimate monetary effects on the local economy. While these studies may serve to illustrate the existence of certain ecosystem values, they are too cumbersome and expensive to be widely applied.[10] Even when used, they have generally been applied to only a subset of services and therefore they fail to provide a comprehensive accounting of values. The fact that ecosystem valuation results are highly site-specific also limits the usefulness of these methods since values from one study area cannot be readily applied to another.[11]

---

rectly affect prices of some marketed good. Benefit transfer methods use valuation studies from one time or place and transfer the values to another location or issue by considering how values may change with new circumstances. Damage cost methods, and related methods of replacement cost or substitute cost, are used to estimate economic values based on costs of avoided damages resulting from lost ecosystem services, costs of replacing ecosystem services, or costs of providing substitute services. Detailed description of the pros and cons of each of these methods, illustrations of how they have been used and references are available at a web site developed by King and others: Dennis M. King & Marisa Mazzotta, *Ecosystem Valuation,* http://www.ecosystemvaluation.org (visited Apr. 18, 2001). Examples of some potential pitfalls of using these methods can be found at *id.*, http://www.ecosystemvaluation.org/Indicators/food.htm#dwv. *See also infra* note 11.

9. For example, in a hedonic pricing valuation study, the value of houses near wetlands may be compared through statistical modeling to values of comparable houses far from wetlands in order to estimate the proportion of a house's value that is derived from being near a wetland. However, these studies typically ignore differences in the wetland that cannot readily be perceived by people. So, while they may include vegetation height, they will not include any information about the wetland's ecosystem functional capacity, such as the ability of the wetland to filter nutrients or serve as wildlife habitat.

10. Several recent textbooks outline non-market methods of assigning value to wetland services and discuss the practical problems of applying the methods usefully. *See, e.g.,* VALUING ENVIRONMENTAL PREFERENCES (Ian J. Bateman & Kenneth G. Willis eds., 1999); Vincent K. Smith & Raymond J. Kopp, *The Spatial Limits of the Travel Cost Recreational Demand Model, in* ESTIMATING ECONOMIC VALUES FOR NATURE 234 (Vincent K. Smith ed., 1996).

11. The difficulties associated with dollar-based wetland valuation do not end when researchers have generated dollar estimates and made them available to policymakers. We present several brief cautionary case summaries of "valuation backfires" based on experiences of the authors. They provide sobering warnings to those who decide to use dollar-based wetland valuation to influence environmental policy without understanding how the numbers are generated or how they can be abused.

Case # 1. The "Willingness to Pay" Survey. At a coastal zone hearing, a wetland advocate testifies that a recently published survey shows that people are willing to pay $100 per

Thus, decision-makers attempting to manage wetland mitigation banking trades face a dilemma. Based on conventional wetland assessment methods, trade regulators have only a limited technical basis for allowing or denying mitigation proposals. At the present time, assigning credible monetary values to wetlands is generally not possible because services are not adequately captured by these conventional methods. Equally, there is no credible way to make most wetland management decisions using comparisons that are based purely on biophysical measures of wetland characteristics. Yet, without appropriate valuation methods, wetland trading may lead to losses in wetland functions and values—even if there is no net loss of wetland area.

Unlike "natural" markets, markets for wetland mitigation cred-

---

household to protect wetlands within twenty-five miles of their homes. The opposing expert points out that the survey did not specify the type, size, or condition of the wetland or specify the number of wetlands already in the survey area. She presents evidence that the survey results in a nearly infinite dollar value being placed on tiny degraded wetlands in urban settings—where there are many households—and a very low value being assigned to large pristine wetlands in rural areas. Admitting that the cost of doing the survey correctly would be prohibitive, the wetland advocate withdraws his testimony and the wetland in question is permitted for development. Moral: Landscape context is important to value—one reason why valuation studies can not be used directly for areas other than the study area.

Case # 2. The Derived Fishery Value Approach. Studies show that coastal wetlands in Massachusetts support over 75% of commercially valuable fish species. However, Massachusetts's fisheries have been so mismanaged and overfished over the past twenty years that their economic value is near zero. A contracted study to estimate the "derived value" of wetlands to the state's fisheries yields estimates that are less than one dollar per wetland acre. Moral: Using this method or similar dollar valuation method can result in very little justification for protecting wetlands.

Case # 3. Benefit Transfer Approach. An environmental group presents testimony in Oregon, based on a widely disputed study in Louisiana, that wetlands generate economic value of $28,000 per acre. The opposing side agrees to accept the number as fact, and points out that the county already requires $40,000 per acre in compensation for wetland impacts as part of its "in lieu" mitigation fee program. Later, a group of wetland developers who are paying $40,000 per acre as impact fees sue the state to reduce the fee using the evidence presented by the environmental group and get the fee lowered to $28,000. Moral: Valuation numbers taken out of the context where they were developed can be dangerously misleading when they fail to take into account local land markets.

Case # 4. The Replacement Cost Approach. At the request of state wetland managers, local engineers estimate that the cost of trying to restore a non-tidal wetland area that is being threatened with development to pre-colonial conditions is over $300,000 per acre. This figure is used at a public hearing as an indicator of the wetland's value. Under questioning, the wetland manager agrees that "no one in his right mind" would spend $300 million to try to restore this 1,000-acre site. When asked if it was fair to offer the $300,000 per acre figure as an estimate of the wetland's economic value, the wetland manager admits he is not sure. Moral: Replacement costs are not valid methods of valuation if people would adapt or find substitutes rather than pay the estimated amount.

its are driven entirely by regulators who control every nuance of supply and demand, including who can buy and sell.[12] Buyers of mitigation credits are price-conscious, sellers of mitigation credits are cost-conscious, and neither is quality-conscious. The cost of providing mitigation credits, and concomitantly the price of buying them, increases with the quality of wetland mitigation. Both parties, therefore, have incentives for quick and cheap mitigation. Consequently, buyers and sellers of wetland mitigation provide only as much quality as trade regulators require. In other words, because wetland mitigation trading is a regulatory construct, buyers and sellers have no independent incentive to ensure quality at the mitigation sites. As the sole guardians of quality in wetlands mitigation trading, regulators walk a thin line between allowing trades that result in net losses of wetland value and raising standards so high that environmentally and economically beneficial trades are deterred.

In the absence of a sufficiently comprehensive framework for objective wetlands valuation, current wetland trade regulation occurs somewhere along a spectrum from purely political negotiation at one end to a system based solely on narrow measurable criteria at the other. At one extreme, the political approach ignores specific site-based criteria altogether. In these cases, government agencies base mitigation debit/credit criteria almost exclusively on political criteria or on ad hoc negotiations among interested parties (e.g., permit-seekers, mitigation providers, and wetland trade regulators). At the other extreme, predetermined, clearly defined wetland assessment criteria are used to develop predictable trading rules that are implemented uniformly in all wetland trades with no room for case-by-case regulatory discretion.

The majority of current permitting, however, occurs somewhere in the middle of these two extremes. In this "semi-political" approach, a few simple debit/credit criteria are used but wetland regulators have significant discretion to consider other factors, including the preferences of interest groups, to arrive at acceptable trades. The semi-political approach is popular because it uses some scientific criteria, which provides political cover, but also offers regulators the flexibility to negotiate mitigation deals that allow development to go forward. Needless to say, however, a system subject to the political winds may not provide the means to make politically

12. James Salzman & J.B. Ruhl, *Currencies and the Commodification of Environmental Law*, 53 STAN. L. REV. 607 (2001).

difficult choices when and where they are needed to preserve valuable wetland services.

Commercial mitigation bankers and wetland permit seekers naturally propose wetland trades that bring themselves financial benefits. Under current regulatory approaches, these trades often result in no net loss in wetland area and reasonable exchanges in the functional capacity of wetlands. But they can also result in significant losses in public wetland values. For example, the costs of providing wetland functional capacity in remote undeveloped areas are typically far lower than the costs of providing the same functional capacity in relatively developed areas. These market incentives appear to have resulted in a migration of wetlands from high-development areas, where wetland development pressure is high, to remote areas where mitigation costs are low.[13]

Without information beyond the basic biophysical measures of wetlands lost and gained, it is difficult to determine if patterns of mitigation trades that result in the migration of wetlands from urban or suburban to more rural areas benefit society. While some wetland functions may be of higher quality, more readily utilized, and therefore more valuable, at remote inaccessible sites (e.g., reproductive and feeding habitats for rare birds, fish or other wildlife), others that depend on the proximity of the wetland to people in order to provide valued services (e.g., aesthetics/local recreational viewing/educational opportunities, flood protection, groundwater recharge, sediment trapping) tend to make urban sites more valuable. The wetland trading problem, in other words, is not a built-in bias for more urban or more rural sites. Rather, the problem is that no practical and systematic methodology currently exists for incorporating important value-based tradeoffs into wetland trading criteria.

This article presents and demonstrates a wetland value indicator methodology (WVI) that can be used to define credit criteria for wetland trades or evaluate priorities for public investments in restoration or preservation of natural areas. The indicator system we propose strikes a middle ground between conventional biophys-

13. In a limited study of wetland mitigation banking in Florida, King and Hebert determined that the population densities in areas of wetland losses were four to eight times higher than those in the vicinity of existing and proposed wetland mitigation banks. Although such differences do not establish any overall loss of wetland value, the authors provided evidence that values associated with at least some wetland services clearly declined as a result of the trades. Dennis King & Luke Herbert, *The Fungibility of Wetlands* 19 NATIONAL WETLANDS NEWSLETTER 10 (1997).

ical wetland assessment methods, which capture ecological function but ignore wetland economic valuation, and economic methodologies which attempt to provide objective criteria for evaluating trade-offs but often ignore many sources of value. We have addressed the challenge of integrating both concepts by developing indicators that act as surrogates for economic values, focusing not on specific monetary values or specific biophysical features but, rather, on the ecological-economic linkages. We develop wetland indicators that capture the relative level of benefits of individual wetlands without attempting to assign precise monetary values to those benefits.[14] The method is designed to help manage wetland mitigation and "score" wetland mitigation trades, but it has broader applicability and could be used to prioritize wetland conservation and restoration efforts, watershed planning, and land use zoning.

Our study is a pioneering effort that demonstrates some of the principles of a location-based valuation system and provides a practical demonstration of its use. The study's greatest value, however, lies not in providing a single "score" of a wetland's service value or developing the definitive indicators for valuation but, rather, in assessing which parts of the valuation methodology are practical and informative and which require greater development. It also demonstrates that it should be possible to implement the WVI system without requiring more assessment of wetland site conditions than is typically done now. Thus, we hope that the lessons from this research will provide the foundation for future valuation work—transforming the valuation methodology, indicators, and GIS information into a robust and standardized method of analysis.

Part II of this paper describes the traditional methods of scoring wetland trades and their shortcomings in providing useful valuations. Part III explains the ecological and economic concepts that form the basis of our proposed indicator system. Parts IV and V present a case study implementing the methodology to evaluate actual trades at an operating mitigation bank. Part IV introduces the site and explains how we developed site-specific wetland value indicators. Part V summarizes and interprets the results of the analysis. Part VI discusses the obstacles we encountered in assessing the

14. For greater detail regarding the economic foundations of the method, see James Boyd, Dennis King & Lisa Wainger, *Compensation for Lost Ecosystem Services: The Need for Benefit-Based Transfer Ratios and Restoration Criteria*, 20 STAN. ENVTL. L.J. 393 (2001).

mitigation trades and provides recommendations for further research.

II. VALUING WETLANDS

This section examines the various characteristics of a wetland that determine its value and discusses why consideration of all these factors is necessary for appropriate management of natural resources. A wetland's features, such as size and vegetation, and its functional abilities are the biophysical basis of its value. Equally important, however, are factors such as a wetland's setting in terms of local and regional land use configurations and related human activities. Ultimately, valuation requires combining these ecological and socio-demographic considerations.

A. *Why a New Valuation Methodology?*

A valuation approach to assessing trades has only recently become necessary, since the regulations guiding wetland mitigation previously maintained a clear preference for "on-site, in-kind" mitigation.[15] Under this regime, mitigation providers were required to replace wetlands with the same type of wetland at or contiguous to the site of wetland loss. Because the impacted and replacement wetlands were always at the same location, as long as regulators forced mitigation providers to restore the same level of wetland functional capacity, it was reasonable to assume that the replacement wetlands would generate similar services and benefits. Furthermore, because they were provided at the same location, these services and benefits were reasonably expected to accrue to the same segment of the population that was losing wetland benefits. As a result, with on-site mitigation, the indices of wetland functional capacity produced by science-based wetland assessment methods were acceptable proxies for wetland service values.[16] Moreover, with on-site mitigation, "distributional" questions about who gained and lost from wetland mitigation trading were unimportant, and the likelihood of loss of social welfare was minimized.

Unfortunately, "on-site" mitigation was not always possible and, because surrounding development often jeopardized the viability of the replacement wetland, on-site mitigation was often undesirable on purely environmental grounds. In practice, on-site mitiga-

15. This history is drawn from Salzman & Ruhl, *supra* note 12.

16. A recent review of these methods is provided in BARTOLDUS, *supra* note 6.

tion resulted in many fragmented and isolated replacement wetlands of questionable value, some of which were enormously costly to create. Forcing on-site wetland mitigation did not succeed at providing "no net loss" on either environmental or economic grounds. In contrast, allowing off-site wetland mitigation has potential advantages. At least in theory, it allows mitigation to be provided in more favorable locations and can result in "trading up," or achieving net gains in wetland functions and values through trading.

By the late 1980's, therefore, regulators were reluctantly accepting that on-site mitigation didn't always make sense and they began allowing permit applicants to provide compensatory mitigation off-site. The transition from on-site to off-site mitigation was a logical step, but it created many practical problems that wetland regulatory agencies have not managed well. Most notably, the traditional wetlands assessment methods used to compare wetlands on the basis of on-site functional capacity alone were no longer adequate to determine the equivalency of wetland functions and values gained and lost as a result of a wetland trade.

B. *The Influence of Landscape Context on Value*

To illustrate the importance of landscape context on the value of services provided by a wetland, consider the two wetland areas in Figure 1 (Sites A and B). Assume that the two wetlands are the same size, the same shape, and have identical biophysical characteristics (e.g., soil, vegetative cover, and hydrology). Based on conventional wetland assessment methods, they appear to have the same capacity to provide all wetland functions and, therefore, would be assigned the same monetary value. This assessment is misguided, however, because the different landscape contexts of the two sites determine the relative value of the functions they will actually provide.

Consider how the following differences depicted in Figure 1 affect the relative value of the wetland areas:

- Site A is more likely than Site B to provide wildlife support because it is accessible (through the open space of the farm field) to wildlife from the upland wildlife refuge area, whereas the road blocks the wildlife corridor to Site B.
- Site A is more likely to support fish habitat than Site B because it is adjacent to fish habitat, whereas Site B is not.
- Site A is more likely to improve water quality than Site B be-

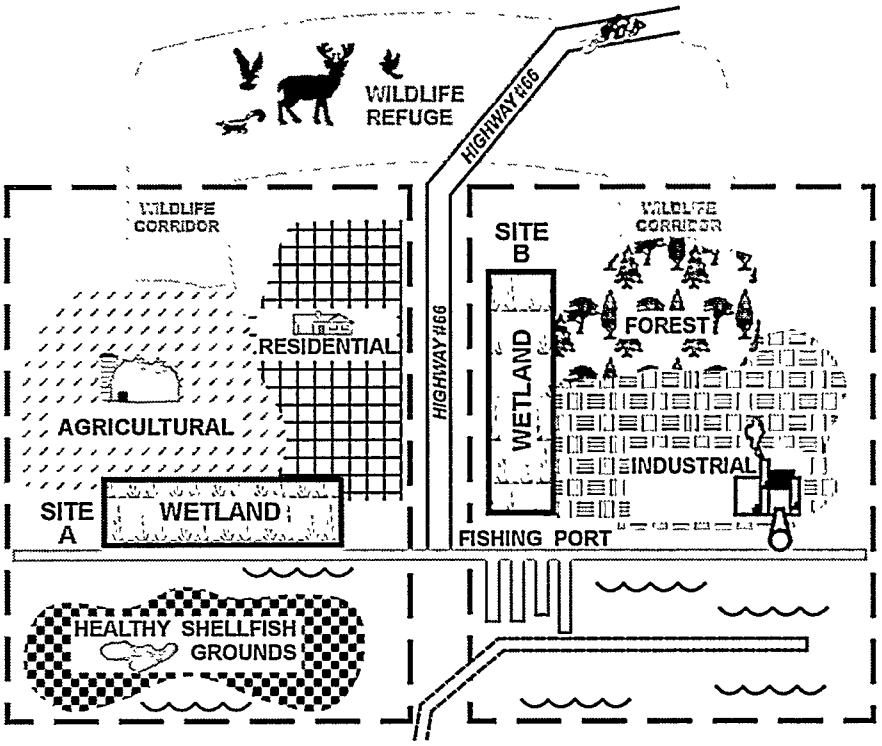

FIGURE 1. Schematic depiction of two hypothetical wetland sites (A and B), illustrating the potential effect of landscape context on the value of wetland services. See text for discussion.

cause of its proximity to the coast and because its longest dimension is parallel rather than perpendicular to the coast, thereby providing greater "buffering" potential than the wetland in Site B.

- Site A is downslope of agricultural production generating harmful levels of nutrients that would enter the water body if not absorbed by the wetland. Site B, on the other hand, creates a narrow "buffer" away from the coast and has no significant upslope source of nutrients to filter.
- Site B is adjacent to a polluted and fast-moving section of the water body where harmful effects from additional pollutants would be negligible. Thus, even with a source of nutrients, the payoff from filtering nutrients at Site B would be less than at Site A.
- Site A provides aesthetic and educational opportunities to a nearby residential population, whereas Site B is surrounded by industrial sites and private forest lands which limit its amenity values.

As a result of differences in landscape context, Site A is more valuable than Site B, despite having identical site characteristics, because it actually delivers services (e.g., provision of edible shellfish through the functions of water purification and nutrient retention) to populations that value them (e.g., users of the shellfish beds).

Landscape context may also affect the reliability of services provided by each area. In this case, the two sites may have significantly different susceptibilities to natural and man-made risks that could disrupt the flow of beneficial services they provide. For example, if a new ten-year land use plan for the region designates the area around Site A as "environmentally sensitive, no industrial use" and the area around Site B as "industrial use, fast-track permitting," then not only are the values provided at Site A higher than those provided by Site B under current landscape conditions, they are less likely to decline in the future as a result of land use changes. Other sources of risk that may further differentiate the present value of the two sites include, for example, water diversion, sea level rise, and invasive species.

C. *Clarifying Terms*

Before further developing the WVI methodology, it is important to clarify our usage of five key terms—features, functions, services, goods, and values—that are interrelated and, unfortunately, often used interchangeably in the literature. Features refer to on-site wetland characteristics; functions refer to biophysical or chemical processes that depend on those features; services and goods both refer to the beneficial outcomes of those processes but are differentiated by whether they provide continuous benefits or discrete units; and values refer to the preferences or importance that people attach to those services. To illustrate, consider a wetland mitigation project that increases the vegetative cover of a wetland (a feature). This feature of the wetland may increase the nutrient trapping capacity of the wetland (a function), which may improve fish habitat and therefore downstream fishing (a service), or produce fish for commercial sale (a good). The value of goods or services is the aggregate amount that users would pay to have the good or use the service. Value is dependent upon the features and functions of the natural systems that generate the service, but also relies on other factors such as people's preferences for the service and the scarcity and substitutability of the service. Only the services

or goods can readily be assigned value since people do not have direct preferences for biochemical processing or other functions on which the services depend. Table 1 offers definitions for these and other terms commonly used in our analysis.

Maintaining clear distinctions between these terms is important for at least three reasons. First, the data and criteria needed to evaluate and assess each aspect of wetlands are significantly different. For example, measuring functions requires biophysical data, while valuing services requires economic and demographic as well as biophysical data. Second, even though functions may be similar between wetlands, they can still differ in value. Differences in the locations of wetlands allow them to provide different types of wetland services.[17] Wetlands located in undisturbed natural habitats may generally provide one set of services best (e.g., endangered species habitats), while wetlands located relatively close to people may provide another set of services better (e.g., educational opportunities or flood damage prevention). Similarly, only wetlands located near disturbed landscapes where runoff is a problem can provide services associated with sediment, nutrient, or contaminant trapping. The third and most important reason for distinguishing between these terms is that the most widely used analytical methods for assessing and comparing wetlands do not focus on the socio-economic components that are necessary for assessing a wetland's contribution to human welfare. Standard wetland assessment methods may refer to "functional values" or "value" indices when describing functional capacity, but they rarely provide information appropriate for determining how resources are valued by people.

III. THE WETLAND VALUATION INDICATOR (WVI) MODEL

The premise underlying the WVI is that wetlands and other natural systems should be evaluated as economic assets that generate various goods and services. Therefore, the wetland's value is based on its ability to produce ecosystem goods and services, the quality and quantity of those goods and services, and the demand for those services where they are produced. Production quantity and quality is a function of the wetland's site and landscape characteristics. The demand or need for services at a particular location is

17. For another discussion of this topic, see Geoffrey Heal et al., *Protecting Natural Capital Through Ecosystem Service Districts*, 20 STAN. ENVTL. L.J. 333 (2001).

TABLE 1. Building blocks of wetland value: some definitions.

Features – on-site characteristics of a wetland that establish its capacity to perform or support various environmental functions (e.g., soil, ground cover, hydrology).

Functions – the biophysical processes that take place within a wetland (e.g., carbon cycling, nutrient trapping, habitat provision). The level of wetland function depends on site and landscape characteristics and can be assessed independently of any human context.

Landscape context – proximity of the wetland to other natural and human-made features in the surrounding landscape. Landscape context influences: a) the wetland's opportunity to function at capacity, b) the services that will flow from those functions, c) the value of those services, and d) the risk that the services will not persist.

Relative preferences – the rank of wetland services in order of importance. Relative preferences for various wetland services are much easier to determine than differences in dollar measures of service values. Although less common than dollar measures of value, individual and community indices of ranked preferences can be used to aggregate service values and compare wetlands using a single number.

Risk – the volatility of potential outcomes. In the case of wetland values, the important risk factors are those that affect the possibility of service flow disruptions and the reversibility of service flow disruptions. These are associated with controllable and uncontrollable on-site risk factors (e.g., invasive plants, overuse, restoration failure) and landscape risk factors (e.g., changes in adjacent land uses, water diversions).

Services – the beneficial outcomes that result from wetland functions (e.g., better fishing and hunting, cleaner water, better views, reduced human health and reduced ecological risks). These require some interaction with, or at least some appreciation by, humans. However, they can be measured in physical terms (e.g., increased catch rates, greater carrying capacity, more user days, reduced risk, property damage avoided). The capacity of a wetland to provide services can be estimated without any ethical or subjective judgements about how much the services are worth. The types of potential services depend to some degree on the level of functions but predominantly on other factors (e.g., access, proximity to people).

Values – defined in strict economic terms, the full range of wetland values includes each person's "willingness-to-pay" in dollars for each wetland service summed across all people and all services. In most cases, tracing and estimating the absolute (dollar) value of a wetland is impossible. However, overall willingness to pay for a wetland service depends on the number of people with access, their income and tastes, the cost of access, the availability of substitutes, and other factors related to local, regional, and national supply and demand.

determined by landscape configuration (arrangement of other land uses), as well as externally imposed conditions.

This conceptualization of the source of ecosystem value leads us to a three-part analysis. First, we evaluate on-site characteristics and the functions they generate using biophysical wetland assessment methods (which is usually conducted as part of the wetland permitting process). Second, we evaluate landscape conditions that affect various location-dependent components of value using landscape data at both local and regional scales. Finally, the on-site and off-site measures are combined to evaluate the value of the goods and services that are produced.

Our framework for wetlands valuation, resolves on- and off-site characteristics into value indicators using both site and landscape-level measurements. The landscape measurements are developed from spatially represented data (e.g., GIS databases) of ecological and socio-demographic characteristics. Identifying a sufficient and tractable set of indicators has constituted the bulk of the work in developing the WVI system to its current state.

As illustrated in Figure 2, the relative value of a wetland can be developed from four types of indicators. On-site aspects of value are captured by functional capacity indicators which are the biophysical measures describing which functions are performed on the site and to what level (relative to a reference wetland). Once functional levels have been determined, off-site characteristics are evaluated with three types of indicators: capacity utilization (measures of how much of the functional capacity is used in creating valued services at that location); scarcity and substitutability (measures of the amount of replacement or substitute services available); and risk (measures of potential loss of service value). Each indicator type is measured using a set of sub-indicators, so that every wetland is evaluated with numerous metrics. The contribution that the individual sub-indicators make to a wetland's relative value must be determined using a system of weights. The system is envisioned to be hierarchical so that sub-indicators are weighted and aggregated for each of the three indicator types and then the scores generated for the three indicator types are weighted and aggregated to arrive at a single score for a wetland.

We say our system measures "relative" value because the indicator values for a particular wetland have only limited meaning when viewed in isolation. They do not represent the value of goods and services as dollars do, but they can be used to compare wetlands

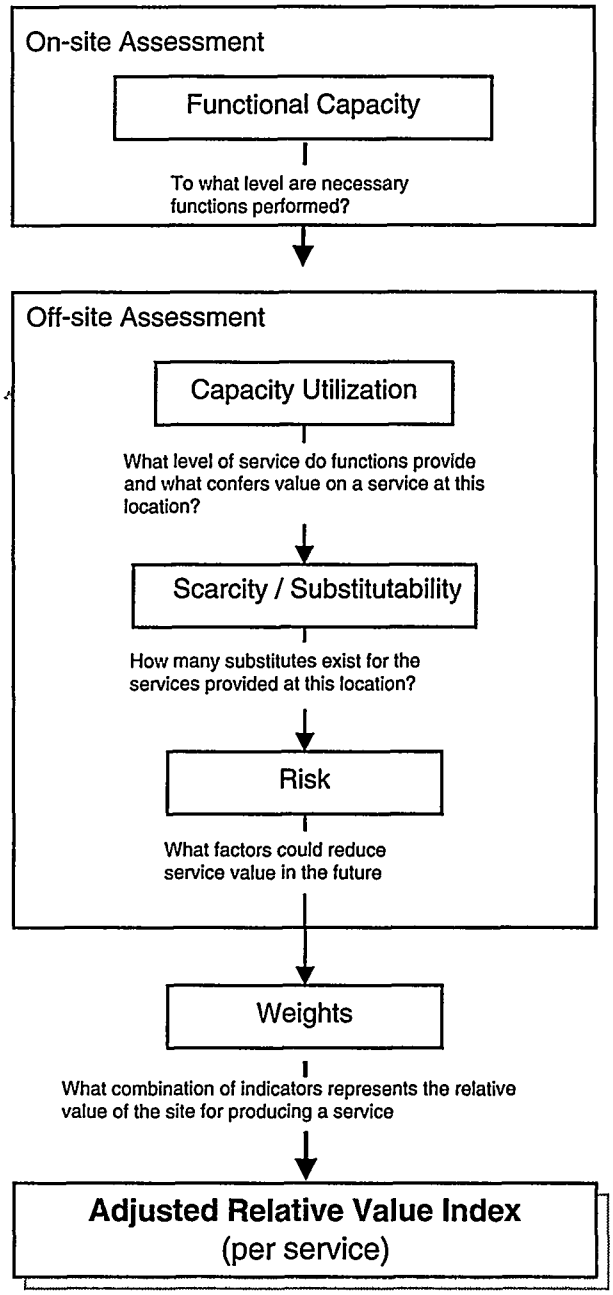

FIGURE 2. Steps in Wetland Value Indicator Development. Using the WVI, a wetland's value is assessed by examining the quantity and quality of the services that it may produce relative to other wetlands and how those services might be valued in that location. Each off-site component may enhance or detract from the ability of on-site features to provide wetland services and the value of that service.

within an appropriate group of wetlands. The appropriate set of wetlands is defined by the question being addressed and could be a group of wetlands nominated for restoration, all wetlands within the state, all wetlands in an ecoregion, or any number of subsets of the world's wetlands. Many land use decisions do not require an absolute measure of value, but instead require selecting among a set of sites (e.g., when deciding where to build a park). Therefore, when we compare wetlands being impacted with those being created in mitigation banks, we only need to capture the relative benefits generated by the sites to understand whether service value is being lost or gained. We still need a system to incorporate a community's preferences for various services into the framework in order to compare gains and losses between service types, which we describe further in Part VI.F.

The following sections explain the four WVI indicator types in more detail, highlighting how they account for both on-site features and the effects of landscape context on wetland value. Table 2 lists some specific questions that we would like to answer in determining which mitigation sites provide the greatest social benefit. In a world of limited resources, however, we cannot find or evaluate all the data needed to reply to questions in Table 2. One of the challenges in practical application of the WVI methodology, therefore, is to determine which data are necessary and, of that, practically available in order to minimize analysis costs.

A. *Functional Capacity*

Functional capacity reflects the capacity of the site to provide a particular function independent of its landscape context. It is based on biophysical characteristics of the site including soil, topography, vegetative cover, and hydrology. In simple terms, it is a measure of the site's capacity, all other things being equal, to provide the ecosystem services or goods. Practically speaking, functional capacity is often scored by comparing a wetland's function level to a reference wetland that shares most, if not all, biophysical features with the wetland being evaluated, but has had minimal disturbance. A site's functional capacity is assessed using any of the biophysical wetland assessment methods that rank wetland conditions with respect to particular functions (i.e., that "score" sites in terms of their functional capacity).[18] The hydrogeomorphic

18. The review of wetland assessment methods provided by BARTOLDUS, *supra* note 6, provides evidence that most methods focus on site conditions and ignore landscape link-

TABLE 2. Key questions in assessing wetland value.

Functional Capacity
What environmental functions does this wetland have the capacity to provide?
Does the wetland's landscape context allow it to provide these functions?
If so, are there factors that will cause it to function at less than full capacity?
Are there factors that may cause it to function beyond its sustainable capacity?
Capacity Utilization
What services, products, and amenities will these wetland functions generate?
Over what geographic area will people benefit from these services and products?
How many people benefit from the services provided?
What is the income, ethnicity, etc. of the service users?
Scarcity / Substitutability
How scarce are these services, products, and amenities in this area?
Are there near-perfect natural substitutes that exist or could be developed?
Are there near-perfect human-made substitutes that exist or could be developed?
How could the affected population adapt to having fewer of these services?
How much would the affected population benefit from having more of these services?
Risk
How vulnerable are services generated by this site to temporary/ permanent disruptions?
How restorable are these services in this region compared to other regions?
How might future development make the services provided here more/less vulnerable?
Will demographic/land use change increase/decrease preferences for these services?
Will demographic/land use changes increase/decrease availability of these services?

(HGM) method,[19] for example, results in Functional Capacity Indicators (FCIs) for roughly ten wetland functions (the exact number varies by locale and wetland type) from sediment and nutrient trapping to waterfowl habitat.

B. *Capacity Utilization*

The level of output (services) that flows from a wetland depends on more than the level of biophysical function it provides. Capacity utilization is a measure of the degree to which landscape setting allows functional capacity to be used. Each type of service associated with a wetland function can be examined in terms of landscape features that limit or enhance the ability of a service to be provided at a given location.[20] Some of the most common landscape features affecting capacity utilization are visitor access, adjacent land use, and downstream resources. The opportunity of a wetland to contribute to certain services, such as recreational fishing, depends in part on the hydrologic and biological connection between the wetland and open water fish habitat and accessibility of the fishing grounds to humans.

Physical and biological distinctions that allow a wetland site to provide water purification and nutrient retention services can be reflected by landscape variables that measure upstream and downstream land uses and land configurations. For instance, using GIS-generated maps of surface water flow, site differences can be quantified by calculating the upland area that would generate runoff into a particular wetland. The likely constituents of runoff and, therefore, the likelihood of providing water purification and siltation services, can be predicted by considering the types of land covers and land uses in areas generating the runoff. For example, if an unregulated animal feeding operation was in the upslope area that

---

ages. They usually do not address landscape factors that link biophysical functions with services that matter to people, and tend to establish the *capacity* of a wetland site to provided functions rather than the *opportunity* the wetland site has to provide functions.

19. *See* Mark M. Brinson, *A Hydrogeomorphic Classification for Wetlands*, (Wetlands Res. Program Technical Rep. No. WRP-DE-4, 1993); Mark M. Brinson & Richard Rheinhardt, *The Role of Reference Wetlands in Functional Assessment and Mitigation*, 6 ECOLOGICAL APPLICATIONS 69 (1996).

20. Capacity and capacity utilization are terms used frequently in economics to characterize the value or potential value of manufactured capital. Capacity usually refers to site characteristics that limit productivity; capacity utilization refers to other factors (e.g., location) that affect how much site capacity is actually used. The productivity of a hotel, for example, depends on room capacity and the capacity utilization in terms of room occupancy rates.

contributes runoff to a wetland, we could reasonably assume that the wetland is receiving excess nutrients. If the biophysical function measure indicates that the wetland has the ability to sequester nutrients, the presence of excess nutrients in runoff allows us to conclude that capacity utilization for the function of water filtration will be high. If the wetland is also upstream of a water body used by swimmers or fishers, it will be able to contribute to the service of providing clean water for swimming or fishing.

Capacity utilization indicators reflect aspects of the socio-economic or biophysical setting that confer added value to a service by distinguishing that service as unusual or by demonstrating particular demand or need for that service at that location. They account for the simple fact that equivalent services provided by different sites may not be equally valuable. As we illustrated with Figure 1, a wetland directly upstream of an industrial site is less able to provide valuable services of providing shellfish or swimming opportunities since there are no shellfish grounds or bathing areas that would benefit from the water purification function in the immediate downstream area.

This class of indicators is used to examine such characteristics as the number of potential users of a service, the opportunity of a site to influence home values or to be used in educating children, and other landscape characteristics that would tend to increase the value of the service at its location. Typical data we would want to evaluate include population densities, demographic statistics, preferences of various demographic groups, habitat preferences of various plants and animals, and regional patterns of recreational use. All these data types may play a part in inferring likely use rates of wetland services.[21]

Information from valuation, preference, and opinion surveys provide another tool for assessing use rates or preferences based on demographics or landscape conditions. For example, the

21. This method of assessing demand is not typical for many reasons. In markets that economists consider "normal," people will tend to reduce consumption of a good when it becomes scarce and its price increases. With many wetland services, however, increasing scarcity is reflected in a lack of availability rather than an increase in the price facing users. We do not take into account how demand for services might decline as services become less available. We believe the substitutability indicators help us consider whether reduced supply might be offset by changes in demand, but this will require more research. Substitutes for new sources of clean drinking water, for example, might become available by reducing waste of potable water. The value of this service, therefore, should depend on the availability of substitutes as well as the affected population and current per capita consumption rates.

United States Fish and Wildlife Service collects survey data every five years on fishing, hunting, and wildlife-related recreational participation rates by state and region and by various age, income, and ethnic groups.[22] This information on likely users of recreational services may be compared to characteristics of the neighborhood and surrounding area to relate the desirability of recreational opportunities.

Potential use rates can be assessed within a small localized neighborhood, or within an entire watershed or county depending on the service. Services that accrue at the regional or state level may be assessed by determining whether a wetland plays a role in regional planning goals. For example, a wetland may provide regional services as part of a coastal protection zone, a drinking water protection area, or a wildlife corridor.

Whether a site provides a service of relatively high value may depend on the configuration of land use types adjacent to and near the site. For example, an animal may view a landscape as being more or less hospitable depending on its ability to move across the landscape without being seen by predators. Therefore, the distance between, say, forest patches and the width and size of the patches, can determine whether an animal will choose that area as habitat. Numerous landscape fragmentation measures[23] have been developed and tested for correlation with species richness, density, and abundance.[24] These fragmentation indicators can be combined with such measures as biodiversity indices, property ownership, trail miles, and hunting restrictions to reflect the level of such

22. U.S. FISH & WILDLIFE SERV., THE 1996 NATIONAL SURVEY OF FISHING, HUNTING, AND WILDLIFE-ASSOCIATED RECREATION (1997), *at* http://fa.r9.fws.gov/surveys/surveys.html (last modified Dec. 10, 1999). Data on licenses for fishing and hunting are also available. This information can be used to show the magnitude and trends of demand for recreational fishing, hunting, and wildlife-viewing over broad areas and can help to prioritize decisions on preserving recreational opportunities overall. However, the data we reviewed did not help us distinguish between demand at or near the various impact sites, which fell within a small geographic area. We believe the data and methodological limitations that prevented a more explicit characterization of the recreational fishing service area of a wetland can be overcome with further study.

23. These measures are spatial statistics that describe the degree of fragmentation of the landscape.

24. *See* Curtis H. Flather & John R. Sauer *Using Landscape Ecology to Test Hypotheses about Large-Scale Abundance Patterns in Migratory Birds* 77 ECOLOGY 28 (1996); James P. Gibbs, *Distribution of Woodland Amphibians Along a Forest Fragmentation Gradient*, 13 LANDSCAPE ECOLOGY 263 (1998); Eric J. Gustafson, *Quantifying Landscape Spatial Pattern: What is The State of the Art?* 1 ECOSYSTEMS 143 (1998); Monica G. Turner, *Landscape Ecology: The Effect of Pattern on Process*, 20 ANN. REV. ECOLOGY & SYSTEMATICS 171 (1989).

services as biodiversity support and viewing or hunting opportunities. These indicators allow us to consider more than the capacity of the site to grow plants favored by particular birds, by accounting for further aspects of land use that determine service value.

C. *Scarcity and Substitutability*

The value of any service is closely related to the service's scarcity and the availability of substitutes. Other things being equal, scarcity and a lack of substitutes increase a service's value. Scarcity and substitutability indicators measure the abundance of the services and the availability of substitutes for those services. While capacity utilization indicators measure the necessary conditions for a service to exist, those conditions, such as the requirement that a wetland be hydrologically connected to an aquifer that is used for water supply, may only represent a necessary but not sufficient condition for the service to have value. For example, groundwater recharge only has value in an economic sense if it improves the availability or quality of the water that is used. Therefore, needed components of value indicators are those that show water is scarce or has degraded quality.

The appropriate scale at which to judge scarcity and substitutability depends on the nature of the service being examined. For example, people may be willing to drive further to try for a rare trophy fish than for a relatively common fish, and anglers may be more likely to find substitutes for the latter than the former. Defining the service area, or geographic range of wetland services, will depend on defining the geographic range of users and the area containing potential substitutes (e.g., saltwater fishing opportunities instead of freshwater). This delineation is confounded by the difficulty of defining which services may be substituted and the fact that some substitutes may be acceptable to some users, but not to others. This service area definition problem is common to most valuation studies of non-marketed goods. While many valuation analyses have attempted to measure willingness to pay for particular recreational or aesthetic experiences, few have attempted to define the service area associated with the ecosystem service. The difficulty of defining service areas for recreation and the effect of that definition on values assessed with travel cost analyses have been noted previously.[25]

25. V. Kerry Smith & Raymond J. Kopp, *The Spatial Limits of the Travel Cost Recreational*

For some wetland services, service area delineation is straightforward because the areas can be defined on the basis of biological or physical characteristics (e.g., a watershed for flood protection services). The service areas for other wetland services (e.g., recreational and educational opportunities), however, cannot be defined without a great degree of knowledge about how frequently existing sites are used, whether certain sites are or may become overcrowded, which sites are considered substitutes, and the influence of local preferences in site selection. The high informational burden of determining appropriate service areas led us to focus on services that cannot easily be transported between regions or accessed by users traveling between regions. Certain simplifying assumptions that we used to define service area are described in the next section.

D. *Service Risk*

The economic value of a wetland depends on the expected flow of services it provides over time. Therefore, risk, insofar as it suggests disruptions in the future service flow, depresses service value. Service disruptions can arise from natural processes (e.g., floods, droughts, fire, and disease) as well as from human activities outside the wetland (construction, pollution), and may affect both functional capacity and landscape components of value. The effect of these potential changes on value is separate from the effects of "discounting," which adjusts the value of future service flows to their "present value" for purposes of valuation.[26]

Natural processes, including those controlled indirectly by human activities, such as sea level rise and dispersion of invasive plants, pose significant risks to wetland function, services, and value in many regions. Known risk factors can be assessed through trend or scenario analysis. For example, detailed maps of predicted

Demand Model, in ESTIMATING ECONOMIC VALUES FOR NATURE 234 (V. Kerry Smith ed., 1996).

26. Discounting is used to reflect our "time preference" for goods, which can be described as our general preference to receive goods or services now versus in the future. References and illustrations of how discounting affects ecosystem values are provided in the Concepts section of Dennis M. King & Marisa Mazzotta, *Ecosystem Valuation*, http://www.ecosystemvaluation.org (last visited April 23). A description of discounting is beyond the scope of this paper, but its effect on the scoring of mitigation trades that involve exchanges of mature (impacted) wetlands for young (restored or created) wetlands can be enormous. The effect of discounting may be particularly important in the case of wetland mitigation trades in Florida which may involve the restoration of cypress swamps that take many years to mature to full functional capacity.

sea level rise have been created for many areas. It may not be necessary to consider natural sources of risk in areas where human risk factors markedly outweigh natural risk factors and vice versa.

Many data sources can provide reliable information about what anthropogenic threats exist and which wetlands are at risk. Zoning plans, sewer extensions, and road construction, in combination with population projections, can suggest a great deal about the potential effects of population growth on wetland services and values.[27] For example, a wetland's functional capacity may be exceeded by new inputs of pollutants. Intensive agriculture or feedlot operations and unsewered medium- and low-density residential areas are land uses that strongly predict pollutants in groundwater and surface water.[28] Plans for these land uses adjacent to or upstream of a wetland that already receives a high nutrient load suggest risk of service flow disruptions. Other zoning or regulatory factors can mitigate (or exacerbate) risk from development such as limits on allowable population densities, limits on land parcel size, and stormwater zoning regulations.

It is useful to note that the site characteristics that increase service risk may tend to be the same ones that cause the service level at a site to be high. The value of bird watching, for example, goes up with proximity to residential development or access, but so does site risk. Tradeoffs associated with these kinds of conflicting goals may need to be evaluated through a regional analysis. Depending on its management goals, a government may choose to create a "portfolio" of wetland sites providing different types of services in different locations, or may concentrate on maintaining a particular type of rare service.

E. *WVI Checklist*

As discussed in this Part, the Wetlands Value Indicator system disaggregates wetland valuation into a five step analysis:

---

27. Many jurisdictions generate predictions of population growth by locale or zip code. Although many variables contribute to the density, type, and location of new development, zoning and sewer extensions are particularly strong predictors of future development. *See* Nancy E. Bockstael & Elena G. Irwin, *Economics and the Land Use - Environment Link*, *in* YEARBOOK OF ENVIRONMENTAL ECONOMICS, 2000-2001 1 (Henk Folmer & Tom Tietenberg eds., 2000).

28. See Carolyn R. Harper et al., *Groundwater Protection in Mixed Land-Use Aquifers*, 16 ENVTL. MGMT. 777 (1992); Lenwood W. Hall et al., *Status Assessment in Acid-Sensitive and Non-Acid Sensitive Maryland Coastal Plain Streams Using an Integrated Biological, Chemical, Physical and Land-Use Approach*, 3 J. AQUAT. ECOSYS. HEALTH 145 (1994).

- Step 1: Functional Capacity. Characterize a wetland in terms of its biophysical functions using a conventional wetland assessment methodology (i.e., assess level of function and confirm that necessary functions exist in order for a site to produce services).
- Step 2: Capacity Utilization. Translate measures of wetland functions into services and examine features that enhance or detract from service being performed at a particular location (i.e., identify potential proximate users who may derive value from the site; characterize access to the site or adjacent sites as necessary for service to be performed; and evaluate opportunities for rare services to be provided).
- Step 3: Service Scarcity/Substitutability. Establish the relative abundance of these services using indicators of regional supply and demand. Examine whether substitutes exist for the services being examined.
- Step 4: Service Risk. Evaluate signs that service flows may be disrupted in the future or that the value of services may change over time (i.e., assess likelihood that function will be diminished; assess likelihood of future increases or decreases in the demand for the service).
- Step 5: Adjusted Wetland Value Index. Indicators created in steps 1-4 are combined to create an adjusted value index for each ecosystem service being considered. Existing functional capacity and certain aspects of capacity utilization will be necessary for services to exist at a location. The factors that will tend to increase value of the service are high capacity utilization, high scarcity and low substitutability, and low risk of service disruptions. (This step is discussed but not demonstrated here).

This brief explanation and illustration of the WVI approach reveals three fundamental features of the method. First, assessments of biophysical wetland characteristics provide an inadequate basis for comparing wetland values because a site's capacity to produce a service does not necessarily mean that the service will be used or valued at that location—in short, location matters. Second, estimates of wetland value based on typical or superficial wetland conditions (as is done in many dollar-based methods) provide an inadequate basis for comparing wetland values because trades involve specific parcels, not generic wetlands. Third, even when wetlands are identical in most respects, it is possible to develop indicators of their relative value based on differences in their landscape contexts.

Proposing a valuation methodology without the theoretical shortcomings of current methodologies is, of course, easy to do

when not facing resource constraints, political opposition, and time pressures. The more difficult, and more useful, test lies in applying the recommended methodology in real life. In the next section, we set out to do just that. The case study presented in the following sections employs the basic aspects of the WVI model and explores the methodology's ease of use, ability to provide defensible answers, time requirements, and costs. Since our aim is to minimize the system's implementation cost, our analysis uses data that are generally available from government agencies[29] and suitable for geographic information system (GIS) analysis.[30] This "road test" not only provides an opportunity to refine the methodology, but also provides important practical insights for others trying to value ecosystems and their services.

IV. CASE STUDY: THE LITTLE PINE ISLAND MITIGATION BANK

The following case study illustrates how wetland value indices can be developed and applied to improve the evaluation of wetland mitigation trades. In addition to showing how the WVI method could have affected the outcome of actual wetland mitigation trades, the analysis also provides guidance for further method development and practical insights for applying the method elsewhere. In doing so, we: (1) select a set of actual wetland mitigation trades; (2) collect site and landscape information about wetland areas gained and lost; (3) apply the information to develop relative wetland value indicators, (4) illustrate how using the indicators might have influenced the trades; and (5) interpret the results to guide further research.

A. *Little Pine Island*

This section explains how the Little Pine Island site was selected, describes the site, and identifies several assumptions made about the site.

1. *The case selection process.*

The Little Pine Island Mitigation Bank (LPI) was selected after reviewing generally available trade and landscape data for wetland mitigation banks in Maryland, Virginia, and Florida. Time and

29. For example, the United States Bureau of the Census, the United States Environmental Protection Agency, and state regulatory agencies.

30. GIS refers to computer software used to organize, manipulate, analyze, and display spatial data (i.e., computer-generated maps).

budget constraints limited the case study to one area. Initial screening identified sites where functional indicators had been used to score trades in significantly different landscape contexts and where geographic information system (GIS) data were available to assess landscape variables. The quantity and quality of information about trades and criteria used to score them varied enormously, but the search quickly narrowed to private banking operations in Florida because (a) official records regarding mitigation and trading criteria provided by the private sector were generally more accessible and easier to interpret than those of government mitigation programs; (b) there have been a particularly large number of wetland trades in Florida; (c) Florida's Wetland Mitigation Banking Review Team uses clearly-defined credit-scoring methods; and (d) extremely good GIS data are available for the state.

In the end, we selected the LPI due, in part, to its unusual landscape context. In particular, because LPI is located on an island and wetland gains at the LPI mitigation bank are used to mitigate wetland losses primarily on the mainland, the site is a good candidate for illustrating the importance of landscape context in determining the value of ecosystem services.[31]

2. *Description of the case study area.*

LPI, as shown in Figure 3, is a 4,670 acre, state-owned uninhabited island just off the southwest Florida coast near Ft. Myers and within the South Florida Water Management District (SFWMD). The island originally contained four distinct wetland types: (1) coastal forested freshwater, (2) coastal forested saltwater, (3) herbaceous freshwater/brackish coastal, and (4) herbaceous saltwater. However, the island's wetlands have been colonized by harmful invasive species, including melaleuca (*Melaleuca quinquenervia*), Brazilian pepper (*Schinus terebinthifolius*), and Australian pine (*Casuarinas spp.*).[32] These changes have disrupted the natural hydrology, displaced native vegetation, and reduced habitat quality on the island. Over 1,600 acres of LPI has been transformed from "wet savannas dotted with hammocks and pine islands to a thick impenetrable exotic forest."[33]

31. We emphasize that LPI was not picked because we thought it involved bad trades but, rather, because we expected the landscape differences between developed and mitigated sites to clearly illustrate service differences that result from location.

32. KEVIN ERWIN, THE RESTORATION OF LITTLE PINE ISLAND FLORIDA (1998) (on file with author).

33. *Id.*

Access to LPI is via a public two-lane road that connects the mainland, LPI, and Pine Island, a larger and residential island to the east of LPI. Bridges link the road to the mainland and to Pine Island, and use is primarily by residents of and visitors to Pine Island. There is an abandoned waste treatment plant near the center of LPI but there are no other permanent structures. In 1993, Mariner Properties, Inc. entered into an agreement with the state of Florida and the federal government to form a wetland mitigation bank on LPI. The bankers agreed to restore wetlands on the island in return for the right to sell wetland mitigation credits to permit-seekers wishing to develop wetlands elsewhere in the bank's service area.[34]

Mariner Properties, Inc. is earning mitigation credits by restoring up to 1,616 acres of wetland, or roughly one-third of LPI, over a period of seven to ten years. Wetland restoration at LPI began in 1997 with the removal of exotic vegetation and efforts to restore the island's natural hydrology. Since 1997, more than 400 acres, or 25% of the most heavily impacted areas on LPI, have been restored. More than 3,000 meters of canals have been filled and backfill areas excavated to restore historical elevation. Although no documentation is available concerning wildlife on LPI prior to restoration, numerous species such as otters, osprey and other migratory birds are likely to have used the habitat. A nesting pair of bald eagles has been documented on LPI since restoration efforts began.

The functional assessment method used to score wetland trades at LPI[35] is among the best we have encountered for assessing gains and losses in functional capacity, and seems well suited for the wetlands in this region. However, like other wetland assessment methods, it does not address the value of wetlands, or how wetland exchanges are likely to affect the geographical distribution of service loss and gain over the bank service area.[36] As a result, the even-

34. The service area includes portions of coastal Collier, Lee, Charlotte, and Sarasota Counties inland from the coast to the 100-year flood plain boundary.

35. ERWIN, *supra* note 32, at 3-4.

36. Mitigation bank service areas are determined as part of the permitting process by the government agencies involved. Trades between the bank and wetland impact sites are restricted to the bank's service area unless a permitting agency grants an exception. Service areas are typically defined using watershed boundaries, although consideration of political boundaries or other bank's service areas may sometimes take precedent. LPI's service area is unusual for Florida in that it follows flood plain rather than watershed boundaries.

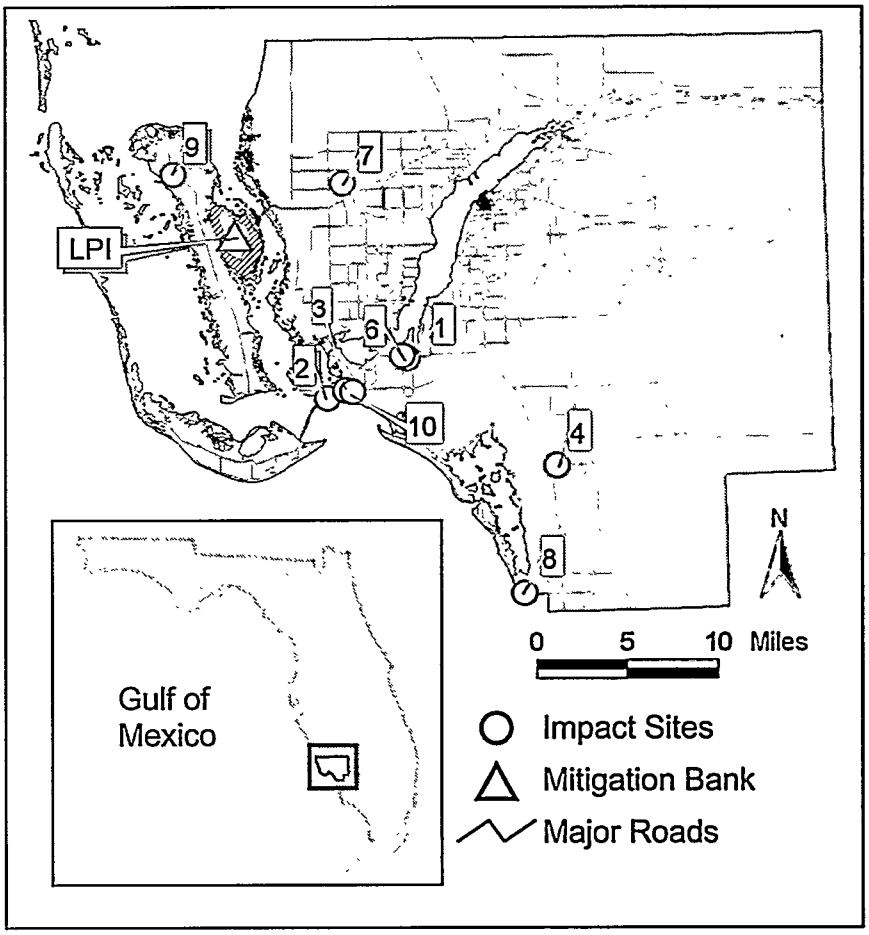

FIGURE 3. Site map of little pine island (LPI) mitigation band and impacted sites. Some linear impact sites are represented as single points. Site numbers are arbitrary. (Site five is missing because it fell outside the county and was omitted from the study.)

tual sale of the 1,616 acres at LPI for mitigation could constitute a significant loss (or gain) in wetland value.

The LPI wetland mitigation bank is being used to offset wetland impacts under regulations implemented by the state of Florida, two regional water management districts, and the federal government. For purposes of this study, we examine only those wetland mitigation trades with LPI that occur between late 1997 (after the bank opened) and September 1999, and that fall under federal jurisdiction. The locations of ten of the wetland sites using Little Pine Is-

land for mitigation are shown in Figure 3.[37]

3. *Simplifying assumptions.*

For ease of analysis, we made a series of assumptions. First, we posited that gains and losses in wetland functional capacity resulting from each mitigation trade were equal, and that the only potential sources of differences in wetland values were differences in location. Second, to isolate the effects of wetland location on wetland value, we assumed that the functional capacity lost at a wetland impact site is fully and immediately replaced at the mitigation site. We also assumed that the risks of future landscape changes affecting wetland value at the impacted and mitigated sites were equivalent. Of course, in reality not all the functions are equal and it may take many years for a restored mitigation site to replace the lost functional capacity at the impacted site. Indeed the mitigation wetland may never achieve the same level of wetland function that is lost.

We further simplified our investigation by focusing on four wetland services: flood damage avoided,[38] safe and abundant drinking water supply,[39] recreational fishing opportunities,[40] and a combined service of neighborhood aesthetic amenities or recreational viewing opportunities (e.g., having nice views or seeing song birds in your yard or neighborhood) and educational opportunities (e.g., having wetlands near schools or along public paths to allow structured or unstructured educational opportunities).[41] We chose

37. In order to simplify data collection and analysis, we calculated indicators only for wetland impact (i.e., development) sites that fell within Lee County and we truncated watershed analyses at the Lee County boundary. Locations shown are not exact for all sites due to information limitations in some public documents. Also, we included trades that were not finalized and, thus, some sites may not have been given final approval.

38. Wetlands retain standing water and reduce the velocity of surface water flows more effectively than other land types. The services provided by this function are primarily associated with damage avoidance.

39. Wetlands, by trapping water that would otherwise be lost to drainage or evaporation, are an important source of freshwater to underground aquifers. Also, because they trap nutrients and filter impurities, wetlands often improve aquifer water quality.

40. Wetlands influence surface water quality and, therefore, recreational fishing opportunities, through filtration of runoff and by routing runoff water to aquifers which later discharge water into streams and estuaries. In estuarine areas, the recharge of groundwater aquifers can affect surface water salinity.

41. One service that most wetlands have the capacity to provide is described as "natural, open space, and visual amenities." The value of these services is enhanced if the wetland exists where open space and scenic vistas are scarce, and where the wetland attracts wildlife that adds to visual amenities and recreational viewing activities (i.e. bird-watching). All else equal, beautiful wetlands in remote inaccessible areas are less valuable than beauti-

these four because they generally encompass the range of services both dependent on ecosystem functions and valued by large numbers of people. Lessons learned from examining these services, however, should be directly transferable to other services.

B. *Data Sources*

After selecting the case study area, we collected as much socioeconomic, demographic, topographic, hydrological, and environmental data as possible for areas around the wetlands that were mitigated at the Little Pine Island bank. We necessarily cast a wide net because it is difficult to determine a priori how many indicators might be needed to reflect relative wetland values, how many would be required to provide a useful basis for scoring trades, or how many could be determined from available data. Fortunately, such data are becoming easily available at low cost in many regions of the United States.

The extent of available GIS data for Florida is perhaps the best one can hope for, with roughly 240 layers (computer maps) available.[42] These maps encompass the following categories of information: cultural (e.g., demographic data, city boundaries, historical sites), habitat (for various wildlife or overall biodiversity), hazards (e.g., flood plains, flight obstructions), satellite imagery, physical (e.g., topography, bathymetry), planning (e.g., natural areas, marine sanctuaries), property (e.g., land values), and transportation (e.g., roads). While GIS data are not absolutely necessary to analyze differences in landscape conditions or to develop indicators, they do make the analysis much easier. Florida is unusual in that a large variety of data from government agencies has been referenced using spatial coordinates (georeferenced) and is thus suitable for GIS analyses.[43] The database also provides detailed spatial

ful wetlands where they can be seen. There are obvious tradeoffs between this wetland service and others, such as endangered species protection, where lower accessibility and exposure to people may be preferred.

42. The data are readily available on the internet. GeoPlan Center, University of Florida, *Florida Geographic Data Library, at* http://www.fgdl.org (last visited April 23, 2001) [hereinafter FGDL].

43. We used a variety of the available GIS data types for our wetland landscape analyses. Many of these data types, such as roads or land use classifications, are available for almost all areas of the United States. However, Florida's land cover data have exceptionally detailed land cover categories, including ones for invasive plant species. In addition, uncommon data have been gathered such as the location of driving, paddling, and hiking paths; aggregate land values for each section-township-range; the locations of sensitive species of fish, bird, fur-bearing animals; and the locations of biodiversity hotspots. *Id.*

data from federal government agencies including the full set of census demographic data such as age, income, and residential housing type.[44]

The amount of landscape data collected for this project was generally satisfactory. However, time and budget constraints precluded organizing, standardizing, and analyzing all available data to develop all the indicators that might be useful. Certain critical types of data, such as information on the prevalence and distribution of storm water management devices (useful for evaluating the availability of substitute services for flood protection), were unavailable and are shown as blank columns in the analysis tables to stress their importance and the need for additional data.

C. *Setting the Geographic Range of Analysis*

As Heal et al. explain in this issue, different services operate on different spatial scales.[45] Appropriate areas for evaluating services were defined using 1) watershed boundaries in cases where water movement was a concern,[46] 2) county boundaries when human choice was the important variable, and 3) immediate surrounding neighborhood for services that accrue locally. These boundaries were used to define the likely users of a service, the area in which access to a service is possible, and the area over which services might be scarce or have substitutes. Physical boundaries were varied based on the nature of the functions providing the services or the mobility of users.

We use different spatial scales to assess potential use rates of various wetland services: the local neighborhood, the watershed, or county. Use rates are assessed locally, for example, if we assume that only local populations will benefit. This would be the case for example, with certain aesthetics benefits, such as the enjoyment of scenic vistas or open space that require ownership, access, or adja-

44. In addition to the data made available from FGDL, the SFWMD planning department provided a GIS coverage to us that reflected predicted changes in land use and another showing locations of public water supply wells and wells used for agricultural, landscaping and commercial/industrial uses. Additional data were collected from a variety of government web sites. Some of this web-based information was used in tabular form and related to GIS files. Some EPA data were included on the FGDL data disks. Other EPA files were downloaded from the EPA website or requested directly from EPA.

45. Heal, *supra* note 17.

46. The watersheds we used were defined by the SFWMD and were generally smaller than the 8-digit HUC code watersheds defined by the USGS and used by the EPA in the Index of Watershed Indicators web site. *See* U.S. Environmental Protection Agency, *Index of Watershed Indicators*, http://www.epa.gov/iwi (visited Apr. 16, 2001).

cency. We use a 0.5-mile radius circular neighborhood for most services related to viewing and aesthetic services, assuming that wetlands within one half mile of a park or public trail have the potential to increase the presence of birds and other wildlife at those public areas. The presence of parks indicates that more people will have access to those services than they would if the service were provided in remote or inaccessible areas. We assume that proximity to wetlands improves aesthetic benefits to neighborhood homeowners. The presence of school-age children and/or schools in the vicinity of a wetland is assumed to indicate greater potential demand for educational opportunities. Land cover data (e.g., percent urban land use and the associated impervious surface area within this neighborhood) were used to estimate the relative amount of surface water a wetland might receive, the likely level of pollutants in that water, and the resources the wetland might protect.

Use rates for other services are best assessed over broad areas when resources and beneficial outcomes are distributed over space and time. For example, recharge to the aquifer improves drinking water supply over the extent of the aquifer, and improvements in nearby fish habitats can improve fishing opportunities and fishing success rates many miles away. Just as groundwater may cross watershed boundaries,[47] humans will likely cross county boundaries when selecting recreation sites, especially if living near a county border. Nonetheless, the county is a convenient political boundary with distinct zoning ordinances, regulations, and land use patterns, and thus proves useful when examining human use and preference aspects of landscape setting. Unfortunately, data for the county as a whole provide little information for differentiating between sites, since all of the study sites and the mitigation bank are within the same county. For example, fishers, boaters, and birders in the county could not be further differentiated with respect to their proximity to each recreational site, although we considered the locations of boat ramps, hiking trails, and other features that reflected local access levels to wetlands. In situations where wetland trades take place across a broad geographic area, a great deal of

47. Groundwatersheds may be defined for a particular aquifer to include the recharge and discharge zones of that aquifer and may also include areas in which wells have been or may be drilled to access the aquifer. Although groundwater does not necessarily remain within watershed boundaries, which are defined based on surface water drainage patterns, the lack of information defining "groundwatersheds" forced us to use watersheds as the service area for groundwater recharge as well as other hydrological wetland functions.

data reported at the county level are available that would permit inter-county comparisons of likely users.[48]

In some cases, biophysical information can be used to create measures of a particular wetland's service area that are more detailed than a simple circular neighborhood. For example, a wetland's immediate service area for surface water purification services can often be delineated using elevation data and GIS analysis techniques.[49] Unfortunately, the relatively straightforward GIS technique that makes this delineation possible relies on fine scale elevation data (i.e., Digital Elevation Model data of 1:24000 resolution or better), which were not available for Florida. Even if such elevation data were available, however, the flat terrain and the numerous canals in the region, which strongly influence water flow, may still have precluded accurate delineation of contributing and receiving areas.[50]

D. *Identifying Site-Specific Indicators for Valuation*

This section discusses the development of quantifiable indicators for three of the four building blocks of value—capacity utilization, scarcity and substitutability, and service risk.[51] Indicator selection proved the most challenging, and important, part of the case study. The questions we ideally would have answered were listed earlier in Table 2; however, data and time limitations prevented us from including or measuring all of the indicators that might answer those questions and thus contribute significantly to capturing a site's value. Our eventual goal is to refine the list of

48. For example, county-level information can be used to examine recreational use rates as collected by the US Fish and Wildlife Service.

49. An elevation map can be manipulated to generate an outline of the areas from which surface water would drain into the wetland (contributing areas) and the areas that are likely to receive runoff downslope of the wetland (receiving areas). The contributing areas represent the areas from which wetlands can sequester nutrients, sediments and contaminants in order to protect downstream resources in the receiving areas or beyond.

50. Other researchers looking at water flow have been forced to make similar compromises. *See, e.g., Comprehensive Conservation, Permitting and Mitigation Strategy, A Water Quality Functional Assessment of South Florida Wetlands, at* http://www.sfwmd.gov/org/pld/proj/wetcons/waterq/wq_techpub.pdf (visited Apr. 16, 2001) (using a 300-meter (0.2-mile) circular neighborhood to study wetland risks from nutrient and toxic runoff).

51. Functional capacity, as measured by a biophysical wetland assessment system, is the foundation of our value indicator system. Functional capacity indices would be derived outside of the analysis presented here and would be combined with the other value indicators to ensure that appropriate functions exist on a site in order to create a service. Because functional capacity indices had not been developed for all sites, we assume, for this demonstration, that all sites have equal functional capacity.

indicators into a minimum set that captures the bulk of value, but for this initial implementation we included a broad list of measures even if they were somewhat redundant with other indicators. The sections below explain how we selected indicators and, hopefully, will provide insights for other researchers' efforts. Because the purpose is illustrative, detailed data and rankings for each site are tabulated only for the provision of one service—safe abundant drinking water. Indicators selected for the other services (flood control, recreational fishing, and aesthetics/viewing/educational opportunities) are included in footnotes.[52] The scoring method is described and its results are analyzed in section V.

1. *Capacity utilization.*

Our service capacity indicators for the service of supplying safe, abundant drinking water are chosen to reflect landscape components that cause a function to have value in that location. The indicators will therefore also reflect the loss in value that might result from converting a wetland site to another use. In developing the indicators, we considered many potential sources of value for wetlands services, however, a function only becomes a valued service where and when that service is in demand. For example, a wetland's ability to improve and protect drinking water quality has social value due to health and aesthetic improvements, wherever or whenever quality is impaired. Similarly, a wetland's ability to increase drinking water quantity through aquifer recharge is valuable wherever there are municipal and industrial users with non-artesian wells or declining water tables, since it reduces the costs of extracting water (deeper wells are more expensive wells). Recharge is also valued if it prevents the need for costly desalinization in aquifers with the potential to be intruded by salt water (which can result from insufficient recharge). These linkages between the aquifer recharge function of wetlands and private and public benefits provide a basis for defining wetland value indicators.

The initial steps in representing the value of wetland depend first on the existence of necessary functions (which can be determined through the functional assessment scores of HGM or other conventional techniques) and second on the necessary landscape

52. A full account of the indicators we chose and their scores for all services is forthcoming. Lisa A. Wainger, Dennis M. King, James Boyd, & James S. Wakeley, *Expanding Wetland Assessment Procedures: Development of Relative Wetland Value Indicators* (2001) (manuscript on file with author).

conditions being present to allow the function to become a service. For example, whether or not a wetland site may provide drinking water services depends on whether the wetland is hydrologically connected to an aquifer used for drinking water. Once these basic criteria are met, the relative benefits of having the service in a particular location are examined. In particular, the landscape setting must facilitate delivery of the service in a location where people will use and value it. In the case of safe and abundant drinking water, the wetland's ability to increase recharge and purify incoming water becomes increasingly valuable (i.e., the site has a higher capacity utilization) as more and more people use the water for drinking and as the water entering the wetland becomes more and more polluted. Therefore, capacity utilization indicators for safe drinking water services should reflect either the number of water users, the quality of incoming water, or both.

Available data were resolved into five different indicators of a site's relative capacity utilization in providing safe and abundant drinking water (Table 3). The first indicator, the number of major public water supply wells within 0.5 miles of the site, directly reflects the extent to which the wetland provides valuable recharge and purification services (i.e., safe and abundant drinking water services) for current drinking water supplies with the largest numbers of beneficiaries. While aquifers supplying drinking water may be recharged by wetlands located throughout the aquifer's recharge zone,[53] which typically extends further than 0.5 miles from the supply well, keeping land near a well in wetlands can be particularly valuable in preventing local drawdown and contamination.[54]

The second capacity utilization indicator, the number of permitted water supply wells within 0.5 miles of the site, reflects the number of water users competing with municipal sources, although not all wells may be accessing the same aquifer. In Florida, permits are required for all commercial wells (landscaping, agricultural, industrial, etc.) and for public water supply, but not for private (household) water supply. As a result, the number of

53. U.S. CORPS OF ENG'RS, CENTRAL AND SOUTHERN FLA. PROJECT COMPREHENSIVE REVIEW STUDY, FINAL INTEGRATED FEASIBILITY REPORT AND PROGRAMMATIC ENVTL. IMPACT STATEMENT, at 3-26 (1999) [hereinafter S FLA FR/PEIS].

54. Drawdown refers to lowered watertables that occur in the vicinity of a well when pumping rates exceed the rate at which water can flow in from the surrounding aquifer. Increased recharge from wetlands can prevent drawdown, but persistent drawdown can in turn cause wetlands to dry up and thus be a risk factor for wetland persistence.

permitted wells may significantly underestimate the number of drinking water wells in areas of low and medium density residential development, where houses are likely to be on private, unpermitted wells.

The third capacity utilization indicator, the total population in the census block containing the wetland, is used as a proxy for local water consumption.[55] A more sophisticated approach to estimating water use could weight water consumption on the basis of income.

The fourth and fifth capacity utilization indicators—the distance to the nearest animal feeding operation, and the existing row crop area within 0.5 miles of the site—both relate to the quality of the water likely to enter the wetland and, therefore, the degree to which the wetland's ability to purify water will be utilized. The proximity to animal feeding operations and row crops were selected because these are two of the most common and significant sources of surface water contamination.[56]

2. *Scarcity and substitutability.*

All other things being equal, the value of a particular service increases as its scarcity increases. The availability of either identical services or substitutable services will determine scarcity. Ideally, drinking water scarcity would be directly reflected in drinking water prices. However, for a number of reasons, the price paid for water in a region is typically not a good indicator of its scarcity or

55. Census "block group" data provide information on a wide range of demographic variables such as population age structure, education, ethnicity, income, and neighborhood characteristics such as housing type for relatively small areas. The size of a block group is dependent on population density; block groups usually contain between 250 and 550 housing units. In our study, the median size of a block group was roughly two square miles.

56. We produced similar tables of the sites' service capacity for flood control, recreational fishing opportunities, and aesthetics/viewing/educational opportunities. The indicators used for flood control were: number of house units in floodplain, percent major roads in floodplain, distance of wetland to watershed outflow point, neighborhood property value, and presence of culturally important structures in flood plain. The indicators used for recreational fishing were: distance to watershed outflow point, whether wetland is on major river or shoreline, whether there are seagrass beds within 0.5 miles, whether there is rowcrop area in the site vicinity, number of boat ramps in county, and fishing pier capacity in county. The indicators used for aesthetics / viewing / educational opportunities were: number of recreational trails within 0.5 miles, number of parks / recreational facilities within 0.5 miles, number of schools within 0.5 miles, number of households in census block, number of children ages 5-17 in census block, distance to park, preserve or patch of large forest, neighborhood land values, and number of globally rare species in neighborhood.

TABLE 3. Capacity utilization for the service of safe, abundant drinking water. The minimum and maximum values are shown in bold.

Site		1,6	2	3,10	4	7	8	9	LPI
Total population in census block	score	1475	237	237	778	123	736	723	0
	rank	3	2	2	3	1	3	2	1
# Permitted water supply wells within 0.5 miles	score	9	2	3	11	0	0	40	0
	rank	3	2	2	3	1	1	3	1
Major public water supply wells within 0.5 miles	score	N	N	N	2	N	N	N	N
	rank				3				
Distance to nearest animal feeding operations	score	10	14	13	4.3	2.7	3.4	none	none
	rank	2	1	1	2	3	3	1	1
Existing rowcrop area (m^2) in site vicinity (within ~0.5 miles)	score	841626	0	0	212230	0	0	1253480	0
	rank	3	1	1	2	1	1	3	1

value.[57] As a result, we must find other indirect indicators of scarcity and substitutability.

Identifying reliable indicators to quantify these components of value, however, can be extremely difficult. Because wetland services that are locally scarce may be regionally abundant (e.g., freshwater fishing opportunities), scarcity indicators may depend largely on the choice of geographic range. And while some services can be replaced by distant ecosystems rather easily (e.g., existence value of rare species), others cannot (e.g., safe drinking water). We largely avoid this problem, however, because three of the four services we chose to analyze cannot be easily transported between regions or accessed by users traveling between regions (drinking water provision, flood prevention, and the combined service of neighborhood aesthetics/local recreational viewing/educational opportunities). Only the service of providing fishing opportunities presents major problems for determining an approximate service area range.[58]

We examined scarcity to a first order by examining the abun-

57. There are relatively few real markets for water. While water is often priced, distortions and imperfections in the markets for water mean that prices bear only a loose relationship to value. *See* Kenneth Frederick et al., *Economic Value of Freshwater in the United States* (Resources for the Future Discussion Paper 97-03, 1997).

58. Fishing opportunities are clearly abundant within the county, so we did not deal extensively with this issue for the scarcity indicators. However, we considered the scarcity of water quality functions that contribute to regional fishing quality by examining scarcity of natural land uses on a watershed basis.

dance of wetlands over the appropriate service area, which for safe, abundant drinking water was the watershed. While wetlands were assumed to be the major source of supply for the services under consideration, other natural systems were considered as potential substitutes for some services (e.g., upland forests can trap nutrients, sediments, and contaminants). In identifying substitute services, we focused on finding variables that had the greatest likelihood of differentiating between sites. We also limited our analysis to substitutes that provide essentially the same service (e.g., other birding opportunities) as opposed to possible substitutes that varied greatly from the original service (e.g., substituting golf for birding as recreation). These assumptions may be adequate for short-term analyses, but they should be revised as new information warrants.

The scarcity of and availability of substitutes for a wetland's service of contributing to the supply of safe abundant drinking water (i.e., purification and recharge functions) depends largely on actions of aquifer users (residential, agricultural and industrial withdrawal rates relative to supply), land use in the aquifer recharge zone, aquifer characteristics (for which data is usually poor), and climate characteristics (which are largely imposed on the region as a whole rather than being site-specific). In Florida, where shallow aquifers are used for drinking water, the linkage between land use and water supply is unusually direct. According to a Corps of Engineers report for the area,[59] any natural area has the potential to be a groundwater recharge area to shallow aquifers, and, therefore, any wetland loss can be considered to decrease the drinking water supply incrementally for Lee County.

Six indicators were identified to evaluate the relative scarcity and substitutability of the safe abundant drinking water services provided by each wetland site (Table 4). Four of the six indicators are framed in terms of land use and its relationship to recharge and water usage. These include percentages of the wetland's watershed that are currently wetlands, in natural uses (including but not limited to wetlands), and being used for agriculture, as well as the ratio of developed to natural land in the watershed. The percentage of the watershed in wetlands is a direct measure of wetland scarcity, and the percentage of land in natural uses reflects the availability of substitutes (assuming that natural uses are the next

59. S FLA FR/PEIS, *supra* note 53.

best substitutes for recharge provided by wetlands). The percentage of land in agricultural uses represents the relative importance of the purification and recharge functions provided by wetlands (i.e., more agricultural acreage means more need for purification and recharge from wetlands). The ratio of developed uses (residential, industrial, agricultural) to natural uses was used as a proxy for the proportion of impervious surfaces to potential recharge surfaces within the basin (i.e., the relative scarcity of recharge area within a watershed).[60]

The last two indicators—the total population in the watershed, and the level of excess capacity in the current drinking water supply—both address scarcity by quantifying the relative intensity of drinking water use within the watershed. In Lee County as a whole, the scarcity of safe and abundant drinking water is apparent from growth projections and remaining aquifer capacity as reported in a recent water supply assessment.[61] Population is expected to grow by almost 60% by 2020 and industrial, commercial and agricultural needs are also projected to increase for a total increase in demand of 54%. Although excess capacity of water supply is not quantified, the report states that alternative water supplies will be needed to meet demand in the near future. The high cost of alternatives mentioned (e.g., aquifer storage and recovery, desalinization of deeper aquifers, and increased use of reclaimed water), demonstrates the lack of readily available substitutes for supplying the county with water. We used this information to qualitatively rank water scarcity in Lee County (relative to the other counties that were discussed in the report) with the indicator "Level of excess capacity in current drinking water supply." This indicator did not vary over the scale of this analysis but may be more important in areas which use deeper aquifers for drinking water and which are not as rapidly replenished by precipitation. Since water supply was not quantified, excess capacity is a qualitative measure of drinking water scarcity (supply – demand), and the watershed population is a proxy for regional water use (demand).[62] While some of these

60. Land use classifications have a typical percentage of impervious areas associated with them. As impervious area increases with land development, surface runoff to waterways increases, thereby preventing aquifer recharge.

61. DISTRICT-WIDE WATER SUPPLY ASSESSMENT TEAM, DISTRICT-WIDE WATER SUPPLY ASSESSMENT: South Florida Water Management District, West Palm Beach 227 (1998).

62. The indicators used for flood control were percent flood plain in wetland, percent of watershed in natural uses, percent development with stormwater management devices (SWM), and percent existing development in watershed that could be retrofitted with

scarcity indicators may seem redundant with certain capacity utilization indicators, these indicators are calculated at a larger scale and are intended to represent regional aspects of scarcity versus the more local aspects of demand relative to supply that are considered under capacity utilization.

TABLE 4. Scarcity/substitutability for the service of safe, abundant drinking water. The minimum and maximum values are shown in bold.

Site		1,6	2	3,10	4	7	8	9	LPI
% Watershed in wetland	score	7	32	7	32	12	32	30	90
	rank	3	1	3	1	3	1	2	1
Total population in watershed	score	246000	60000	246000	60000	18000	60000	6500	0
	rank	3	3	3	3	1	3	1	1
% Watershed in natural uses	score	19	56	19	56	33	56	70	100
	rank	3	2	3	2	3	2	1	1
% Watershed in agriculture	score	13	24	13	24	6	24	9	0
	rank	2	3	2	3	1	3	1	1
Ratio developed to natural land in watershed	score	4:1	2:3	4:1	2:3	3:2	2:3	2:5	0:1
	rank	3	2	3	2	3	2	1	1
Level of excess capacity in current drinking water supply	score	low	low	low	low	low	low	low	n/a
	rank	2	2	2	2	2	2	2	

3. *Risk.*

Since the value of a wetland as an economic asset is based on the value of the stream of services it is expected to provide through time, value depends on current as well as future conditions. Therefore, a wetland value indicator should reflect expected changes in site and landscape features that affect service flows and their val-

SWM. The indicators used for recreational fishing opportunities were percent riparian zone of major streams or shoreline in wetland (by watershed), percent watershed with impervious surfaces, percent watershed in agriculture, abundance of alternative fishing sites, trends in fish catch per unit effort or other evidence of fishery health, and percent economy dependent on recreational fishing. The indicators used for aesthetics / viewing / educational opportunities were percent riparian zone of major streams or shoreline in wetland (by watershed), percent watershed with impervious surfaces, percent watershed in agriculture, abundance of alternative fishing sites, trends in fish catch per unit effort or other evidence of fishery health, and percent economy dependent on recreational fishing.

ues. In areas of rapid economic development or expected land use changes, the risk of future service flow disruptions can have a significant effect on wetland values. Since these risks are not uniform throughout a watershed, their effects can vary widely from one wetland location to another.

We examined three types of risk that could disrupt wetland services. The first was the risk that the wetland will be unable to provide a function at all because it is developed (e.g., drained and filled to support structures or agricultural production). The second was the risk that the wetland will not be able to function at current levels due to its functional capacity being exceeded or degraded (e.g., excessive nutrient inputs). The third was the risk that changes in adjacent land uses that may not affect the wetland directly, would preclude or diminish the capacity of the wetland to provide functions or services (e.g., by disconnecting the wetland from other watershed features).

The first type of risk—that the wetland will be developed—is only relevant in this case for the mitigation bank. We know the impact sites were at maximum risk of development since they were, in fact, developed. Obviously, a site for which a development permit is being sought would also be at higher risk for development than the bank site. The risk of development is at a maximum at the impact sites and at a minimum at the bank site, regardless of how the indicators are measured. Although this particular type of risk provides no basis for comparing wetland impact and mitigation banking sites, we left this variable in the analysis because of its usefulness when applied to choosing wetland preservation / restoration sites.

Using the available data, we were able to identify five indicators to represent the risk that a wetland would be unable to continue contributing to the service of providing safe, abundant drinking water (Table 5). These indicators dealt almost exclusively with the risk of service flows being disrupted or capacity being exceeding. The first indicator shown in Table 5 attempts to capture the effect of invasion by woody exotic species (melaleuca, Brazilian pepper, or Australian pine) which often negatively influence drinking water availability by lowering the water table (to a much greater degree than native species) and potentially allowing salt water intrusion. To indicate the degree of the future threat from woody invasive species, we used the minimum distance between each wet-

land site and sites that are either mapped as invaded by woody exotics on the land use coverage or recently treated for invasives.

Research has shown that proximity to animal feeding operations, increases in agricultural land within the watershed, and increases in impervious surfaces in the watershed can threaten surface and ground water quality.[63] We have considered these threats using two indicators, one for current "nitrogen risk" and the other for future contamination issues due to proximity to animal feeding operations. The indicator for nitrogen risk is intended to represent risks to wetland function from pollution runoff. The risk level was developed by researchers at SFWMD to represent the relative threat to functional loss in wetlands based on the expected pollutant loads associated with existing nearby land uses.[64] The distance to animal feeding operations may look similar to the one used in capacity utilization, but here it is being measured from predicted future land use. A map of predicted land use for the year 2020, which was developed by the SFWMD Planning Office was used to assess this risk.

The remaining two indicators in Table 5 deal with aspects of water level change due in one case to human activities and in the other to natural forces (which are augmented by human activities). We did not calculate values for the "hydrologic change" indicator because we did not have appropriate data, although useful data undoubtedly exist. We nevertheless included the column in our tables because we feel that a risk assessment is incomplete without assessing either the trends in groundwater levels or changes in groundwater level due to the planned water infrastructure projects mentioned in some of the planning documents. Finally, we used wetland elevation to represent the risk of the wetland being inun-

63. SUZANNE B. BRICKER ET AL., SPECIAL PROJECTS OFFICE & THE NAT'L CENTERS FOR COASTAL OCEAN SCIENCE, NOAA, NATIONAL ESTUARINE EUTROPHICATION ASSESSMENT: EFFECTS OF NUTRIENT ENRICHMENT IN THE NATION'S ESTUARIES (1999); Pamela A. Matson et al., *Agricultural Intensification and Ecosystem Properties*, 277 SCIENCE 504 (1997); William T. Peterjohn & David L. Correll, *Nutrient Dynamics in an Agricultural Watershed: Observations on the Role of a Riparian Forest*, 65 ECOLOGY 1466 (1984).

64. Comprehensive Conservation Permitting and Mitigation Strategy, A Water Quality Functional Assessment of South Florida Wetlands (2000) (unpublished manuscript, prepared for the South Florida Water Management District, on file with the author). The particular assumptions made regarding pollutant loads are described in this report. We selected only the nitrogen risk indicator as an example, but information on phosphorus and toxics were also available. U.S. Envt'l Prot. Agency, et al., *Wetland Water Quality Functional Assessment: Results for Lee County Wetlands, at* http://www.sfwmd.gov/org/pld/proj/wetcons/waterq/wq_lee.htm (last visited April 23, 2001)

dated due to sea level rise. We produced similar tables of the sites' service risk for flood control,[65] recreational fishing opportunities,[66] and aesthetics/local recreational viewing/educational opportunities.[67]

TABLE 5. Risk for the service of safe abundant drinking water. The minimum and maximum values are shown in bold.

Site		1,6	2	3,10	4	7	8	9	LPI
Distance to area invaded with non-native invasive plant species (mi) (malaleuca, brazilian pepper, australian pine)	score	2.1	1.8	0.4	0.1	1.3	0.3	2.4	0.9
	rank	3	3	1	1	2	1	3	2
Nitrogen risk from land use w/in 300 m	score	med	low	low	med	med	low	low	low
	rank	2	3	3	2	2	3	3	3
Distance to animal feeding operation (mi) (future land use)	score	10	14	13	4.3	2.7	3.4	none	none
	rank	2	3	3	2	1	1	3	3
Hydrologic change (gw table drop, stream diversion)	score								
	rank								
Wetland elevation (5 ft accuracy)	score	5	5	5	15	10	10	5	5
	rank	2	2	2	3	3	3	2	2

V. ANALYSIS OF RESULTS

In this section we examine our indicator results for each site and compare scores between sites and the mitigation bank. Since the indicators we have developed do not use a common unit of measure, such as dollars, the scores cannot be directly combined. We are currently developing methods for combining indicators that will account for the relative importance and comparability of individual measures. However, for this initial report on our work,

65. The indicators used were: wetland elevation (5ft accuracy), planned water diversions, development planned on adjacent sites, and change in percent impervious surface within watershed.

66. The indicators used were: distance to area invaded with non-native invasive plant species, nitrogen risk from land use w/in 300 m, distance to animal feeding operation (mi) (future land use), stream buffers required along nearest stream, and trends in recreational fishing demand.

67. The indicators used were: distance to area invaded with non-native invasive plant species, wetland elevation (5ft accuracy), and projected population change of school-aged (5-17) children.

we do not create a single score representing value for each wetland. Instead, we present a simple system for aggregating the indicator scores for each indicator type and for aggregating all indicators for a given service. We do not deal with the issue of weighting the services relative to one another as this is beyond the scope of this study.

A. *Score Ranking*

We use a ranking procedure to produce a simple qualitative measure of the relative strength of each value indicator at each site. For each indicator, scores for all impact sites and Little Pine Island are fit to a normal distribution; values in the top (highest value) third of the distribution (percentile rank > 2/3) are assigned a value of 3; the middle third are assigned a value of 2; and the bottom third (percentile rank < 1/3) are assigned a value of 1. If an indicator value falls on the 1/3 or 2/3 percentile, we give the site the higher rank. When indicators are binary (yes/no), sites with a favorable (value enhancing) characteristic are given a ranked value of 3 and other sites are given no score.[68]

Ranks are assigned so that high rankings always indicate high value. In other words, capacity utilization indicators are ranked high when location gives a wetland an increased likelihood of providing a valued service; scarcity indicators are ranked high when the services are scarce over the measured area; and risk indicators are ranked high when the risk of future service loss is low.

1. *Graphical displays.*

We display these ranks in two formats. First, we use vertical bar graphs to show the distribution of scores between sites for each type of indicator: capacity utilization, scarcity, and risk (Figures 4-6).[69] This allows us to compare the performance of different sites for a particular indicator type. Second, we examine all of the indicator types for a single service by summing the number of occurrences of each possible score and dividing by the total number of scores (Figure 7). In the horizontal bar graphs, the percent of all

68. For example, in Figure 3 we examined whether a major public water supply well fell within 0.5 miles of a site. Since site 4 is the only site for which this was true, that site is given a score of 3 and scores for other sites are blank. When aggregated scores are compared using the percent of indicators at each rank (Fig. 6), the score of a site with a rank on a yes/no question will improve, but scores do not decrease at sites with no score.

69. Functional indicators were not available for all sites.

scores that ranked either 1, 2 or 3 are shown as separate colors. This representation allows us to examine whether scores for all indicator types were consistently high or low for a particular service.

2. *Caveats.*

In presenting scores, we try to minimize subjective choices, but even our limited manipulation of the indicator results involves certain biasing assumptions. For example, when we present the percentage of scores at each rank (Figure 7), we implicitly assign equal weight to each indicator, although we recognize that not all indicators influence site value equally. Moreover, in assuming that the index scores are normally distributed, we introduce several other potential sources of bias into the scoring.[70]

In considering the analysis results, it is also important to remember that we have not incorporated the site-based functional indices that would be a critical component of a complete analysis. Due to inconsistent data availability for sites, and to emphasize the landscape component of the analysis, we instead assume that all the sites had the same functional capacities. Therefore, our results only differentiate among sites based on the presence of landscape factors that enhance or detract from service value. Further, by not including the functional indices, we fail to confirm that the necessary functions exist for services to exist.

B. *Comparison Between Impact Sites*

Indicator scores and ranks for wetland sites are shown in Tables 3-5 and presented graphically in Figures 4-6. The indicator ranks clearly distinguish between the best and worst sites for each indicator and when ranks are compared across all the services, they identify sites that consistently rank either high or low, regardless of the service being examined. The relative value of the intermediate sites, where scores for a particular indicator type were inconsistent, is less clear and requires further analysis. Figures 4-6 show how sites compare for each of three indicator types—capacity utilization, scarcity/substitutability, and risk—for the service of safe and abundant water provision.

1. *Capacity utilization.*

Capacity utilization indicators were the easiest to develop given

70. See discussion *infra* Part VI.F.

available spatial data and, therefore, we measured more of these indicators than any other type when all four services are considered. For the drinking water service (Figure 4), we saw consistency among indicators for particular sites. Scores tended to clump in the low, middle, or high range for some sites (e.g., sites 1,6 and 4 had three high scores and one or two medium scores, and LPI had low scores for all 4 indicators),[71] although some sites' indicators scored at both extremes (e.g., sites 8 and 9).

When viewed for all four services (not all data are shown), the indicators again show clear distinctions between some, but not all, sites. Sites 1,6 and 4 score high in capacity utilization across three of the four service types, but they score somewhat lower for support of recreational fishing. On the other hand, site 7 ranks the lowest across all service types and has only one high score for all the indicators. The remaining sites show a mix of indicator scores. Site 9, on Pine Island, remains inconsistent when all services are considered: it has relatively high scores for recreational fishing, low scores for flooding and drinking water, and moderate scores for aesthetics. The mitigation bank on the adjacent island scores low for all services except for a moderate to high score for the recreational fishing service.

2. *Scarcity and substitutability.*

The scarcity indicators showed the same pattern as the capacity utilization indicators in that some sites' scores were consistent, while others were not (Figure 5). Sites 1,6 and 3,10 scored consistently high, site 7 is inconsistent, and sites 9 and LPI scored consistently low. Sites 2, 4, and 8 ranked consistently high except for the percent of watershed in wetland indicator. These sites are within the same watershed, and because this watershed had the highest proportion of wetlands among the watersheds studied, this indicator was ranked the lowest for all sites within the watershed.

The rankings for sites 2, 4, and 8 show the sensitivity of our results to the scale chosen for examining scarcity. Although these sites have fairly different characteristics in their immediate neighborhood, they received the same low scarcity score because they fall within the same watershed, which is rich in wetlands. In addition, because this indicator scores lower than any other indicator for these sites, the weight it is given will tend to decrease the overall

71. Sites shown as #,# represent sites (e.g., 1,6) that are too close to differentiate for the analysis.

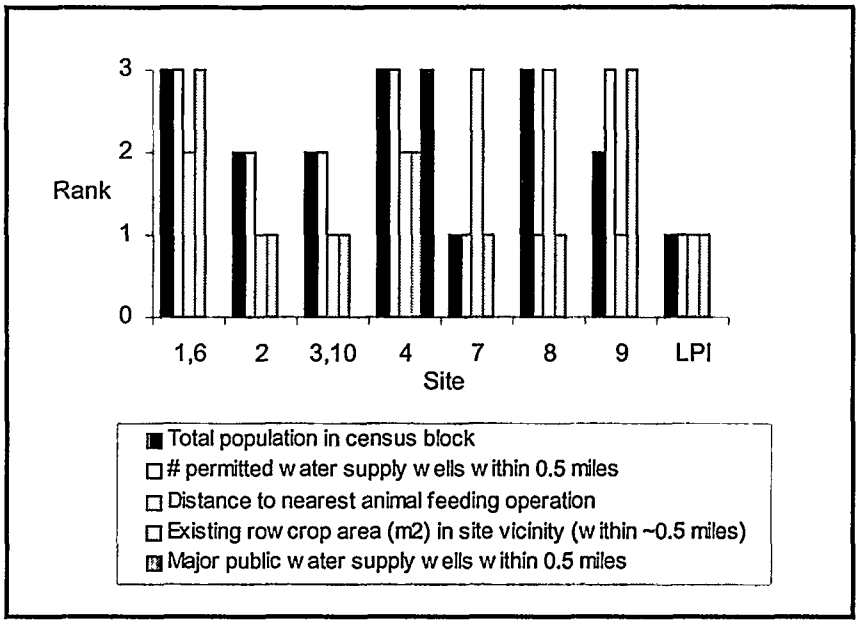

FIGURE 4. Bar graph showing the ranked scores at each site for capacity utilization indicators for the service of providing safe abundant drinking water (1 = low value to 3 = high value). See Table 3 for raw scores and Figure 3 for site locations.

site scores proportionally. In this demonstration, we gave each indicator an implicit weight of one and, therefore, these sites appear to have relatively high value since most ranks are two or three. However, if wetland percentage within the watershed is a major concern, as it arguably could be, this indicator should carry more weight, which would reduce the overall scarcity rank and value accordingly.

3. *Risk.*

Viewed as a group, risk indicators show some consistency within sites and work moderately well to distinguish between sites when there are more than two risk indicators for a service (Figure 6). For instance, site 4 shows generally low scores for risk across all four service types, meaning that the site is considered to be at relatively high risk of service flow disruptions regardless of which service is being considered. For drinking water, site 4's scores are somewhat inconsistent since the proximity of agriculture and an animal feeding operation create one type of risk,[72] whereas the site is the most

72. Simultaneously, proximity to agriculture contributes to high value for capacity

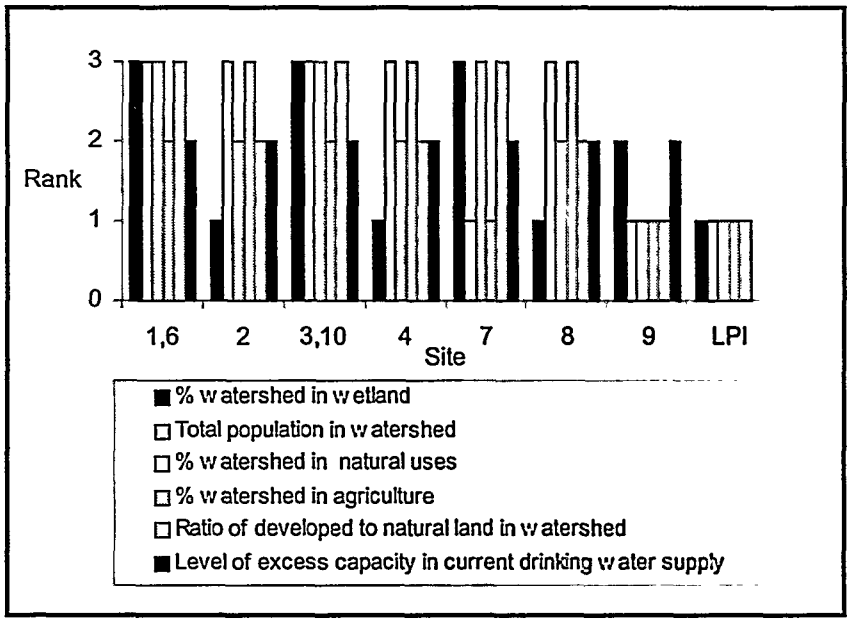

FIGURE 5. Bar graph showing the ranked scores at each site for scarcity/substitutability indicators for the service of providing safe abundant drinking water (1 = low value to 3 = high value). See Table 4 for raw scores and Figure 3 for site locations.

protected from risk of sea level rise compared to other sites due to its higher elevation. On the other hand, Site 9 on Pine Island shows high scores for risk across indicators and service types, indicating that the site is relatively protected from risk factors.

C. *Comparison Across All Indicator Types*

Figure 7 shows how sites compare for all indicator types combined (capacity utilization, scarcity, and risk) for the service of providing safe abundant drinking water. The figure shows a nice separation of values between sites and a clear distinction between impact sites and LPI. LPI and site 7 have the largest proportions of indicators ranked 1 and the lowest proportion ranked 3, tending to indicate an overall low value for these sites for this service. Sites 4 and 1 show the lowest proportion of indicators ranked 1 and among the largest proportions ranked 3, indicating that these sites are high value relative to other sites.[73]

utilization scores. Scoring will eventually need to consider the balance between the contributions to and deductions from value due to these competing influences.

73. Flood damage avoided showed a similar pattern to the drinking water service in

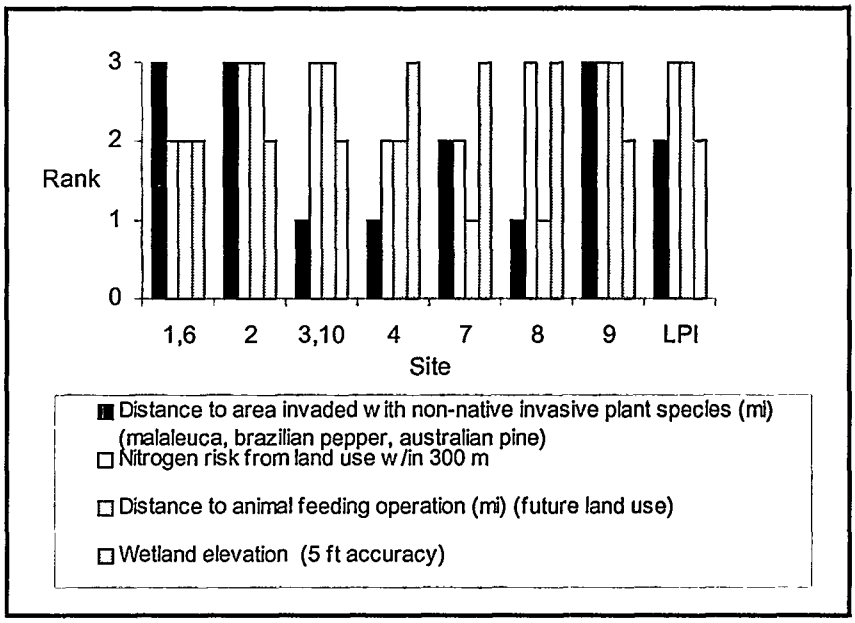

FIGURE 6. Bar graph showing the ranked scores at each site for risk indicators for the service of providing safe abundant drinking water (1 = low value to 3 = high value). See Table 5 for raw scores and Figure 3 for site locations.

D. *Were Scores Consistent Across Services?*

When we compare the results for all services, we see that some sites consistently rank higher than others regardless of the service being evaluated. In other words, sites that rank highly for one ser-

---

that Sites 4 and 1 again ranked highly by showing the lowest proportion of indicators ranked 1 and Site 1 shows the highest proportion of Rank 3 (>85%). Sites 9 and LPI ranked poorly, showing both the lowest percent of rank 3 and the highest percent of rank 1.

Site scores for Recreational Fishing services show less of a distinction between sites than indicators for other services. Site 1 remains one of the best sites with the lowest proportion of rank 1, but shows only the 3rd highest proportion of indicators of rank 3. Sites 2 and 3 show the greatest proportion of indicators at rank 3, followed by Site 9. LPI and Site 9 no longer come in last for this service, but still have among the largest proportions of indicators of rank 1. Overall, it is more difficult to show distinctions between sites because of the low variability in rank scores.

Results for Aesthetics / Viewing / Educational fit a pattern similar to Recreational Fishing or Flood Damage services. Again, LPI is clearly one of the least valuable sites for this service since it has only a small proportion of indicators ranked 3 (~5%) and greater than 60% of indicators ranked 1, which is the greatest proportion of any site for this service. Site 1 again looks relatively valuable for this service since it has the greatest proportion of indicators at rank 3 and the lowest proportion of rank 1 of all sites.

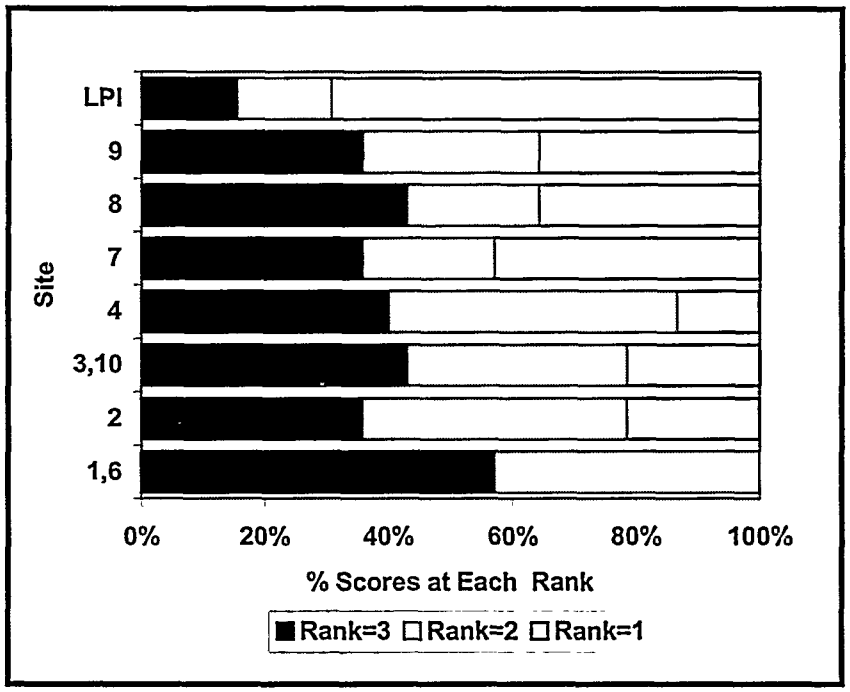

FIGURE 7. Bar graph showing the percentage of all indicator scores falling within each of the three ranked values (1 (white) = low value to 3 (black) = high value). Rank scores (1-3) are summed over all three indicator types (capacity utilization, scarcity/substitutability, and risk) for the service of providing safe abundant drinking water. The percentage of all indicators at a given rank is shown for all impact sites and the LPI mitigation bank. See Figure 3 for site locations.

vice tend to rank highly for other services. For example, site 1 (and adjacent site 6) shows consistently high service values and site 7 generally shows consistently low service values. This consistency across services suggests that determining the relative importance of individual services is not necessarily critical to determining relative site values.[74]

E. *Factors Affecting Final Scores*

The level of land development was an important factor in rela-

74. If we want to create a single score for each wetland site, and if wetlands score differently on different services, we would have to determine how much the score for each service should count towards the final score. If wetlands score consistently regardless of service, then the weighting of services is not important. Helping locales set priorities to determine service weights can be done through various means and has been accomplished using "stakeholder" groups made up of interested parties and citizen juries. *See*, Thomas C. Brown et al., *The Values Jury to Aid Natural Resource Decisions*, 71 LAND ECON. 250 (1995).

tive site scores. For example, sites 1 and 3 both score highly because of their ability to offer a "last defense" of the estuary. Both sites sit near the outflow of a watershed that has become relatively developed and depleted of wetlands; they scored highly because of upslope pollutant sources and local wetland scarcity. Conversely, site 7 had consistently low values due to the minimal benefits it could offer in its sparsely populated suburban, but still disturbed setting.

The poor performance of site 7, apparent in the aggregate scores (e.g., Figure 7), shows the potential importance of an indicator weighting scheme. While site 7 did not offer a wide range of services—few opportunities to contribute to recreational services, for example—the services it did offer were scarce in its setting. Therefore, weighting scarcity more highly than other indicators of value would have led to a higher relative value for site 7. Similarly, Site 4 has special characteristics that generate high ratings in many categories, but loses value due to the fact that its watershed is rich in wetland resources and at risk from agricultural activities (see Parts V.B.2 and V.B.3). The high values for Site 4 originate from location characteristics such as its position between agricultural land (including an animal feeding operation) and two public water supply wells. It also borders a nature preserve. In spite of these significant advantages, Site 4 scores only in the middle to high range of all sites and does not achieve the highest score as we originally thought it might.

Both these examples indicate the importance of using a weighting scheme to assign relative importance to indicator types before aggregating scores. Such a system would allow the relative importance that a community places on, say, future risk factors versus current scarcity, to be included in the final scoring. However, the aggregation of many indicator scores creates other difficulties. When multiple indicators are combined using a weighted sum, for example, indicator values may counterbalance one another and effectively cancel each other out. The greater the number of indicators, the greater chance there is for the message to get muddled. Alternative aggregation methods exist to combine indicators such as multi-criteria statistical methods, which allow the overlap between indicators to be statistically removed. Other techniques use set theory to create sets of indicators that either support or oppose

an objective and ranks their degree of membership in that set.[75] The fact that there is no single agreed-upon strategy and, ultimately, no completely objective method for weighting scores, suggests that indicators may need to be represented in several ways to be effectively interpreted. For example, weights could be based on various risk profiles that weight future risk to a greater or lesser degree.

F. *Passive Services*

Throughout this article, we have focused on "active use" wetland services, which require some interaction between people and wetlands (e.g., access, proximity, and adjacency). However, wetlands may also provide significant value through so-called "passive use" services, which do not require active participation by people.[76] We used active use services in order to clarify the distinction between functions and services (as discussed in Part II.B.) and to show how maintaining a distinction between them can improve the criteria for comparing wetland values. However, a focus on active use services may tend to bias our results such that sites readily accessible to people score more highly in all cases.

In some cases, ignoring passive use values could be an important omission when comparing wetland sites. For example, if small wetlands of low functional level for habitat are traded for a large wetland capable of supporting rare species, passive use services such as existence and bequest values are well supported and the large wetland is more valuable for these services.[77] In these cases, the functional capacity indicators which measure biodiversity or habitat quality may need only minor supplementation from landscape indicators to serve as sufficient measures of a wetland's value for passive uses. For example, landscape indicators are important for understanding whether a given site plays a critical role in preserving connections between habitat patches.

G. *Were Services Compensated at the Mitigation Bank Site?*

The key question of this pilot project has been whether our

75. *E.g.*, P. NIJKAMP ET AL., MULTICRITERIA EVALUATION IN PHYSICAL PLANNING (1990).

76. *See supra* note 7.

77. Existence and bequest values are values that people place on knowing a species exists or that certain species might be available for their children or future relations to enjoy even if they or members of their family never plan to see the animal or visit the habitat.

WVI system provides a different, more accurate assessment of mitigation trades than current practice. Although we were not able to address all the key questions identified in Table 2 in our evaluation of wetland trades, we nevertheless found that the site-based indicators provided some useful distinctions, at least between the sites whose scores were most consistent.

Overall, the wetland impact sites that are on the mainland score consistently higher than the LPI mitigation site. However, judging whether these were "good" trades, in terms of whether services lost at the impact sites were compensated at the bank, requires more information than presented here. A comparison of service value will be influenced by the relative weight assigned to the indicators and to the services considered, the services not considered (e.g., passive use services), and the values of the functional indicators—none of which were included in the scores. However, with more work on indicator aggregation, we will be able to address how landscape components contribute to value in order to augment existing functional indicator methods. Despite the limitations, our analysis shows that the LPI mitigation site provides only a subset of the services provided by the impacted wetland sites. The bank, which sits on an undeveloped island, scores low on value indices that rely on having upland sources of pollutants, such as agricultural land and developed land. It scores low for flood damage avoided because it does not have structures in the flood plain to protect. And it scores low for drinking water services because the groundwater on the island does not appear to be connected to regional water sources. This is not surprising, since services that rely on the presence of resource users and sources of pollutants in order to be valuable cannot be scored as highly when the wetland providing services is in an isolated island setting distant from people, structures, and agriculture.

The story is more complicated, though, since the LPI mitigation site scores reasonably well on indices related to recreational fisheries because of its proximity to shoreline, sea grass beds, and aquatic preserves. It also ranks moderately high on indices related to aesthetic benefits and viewing-related recreational uses. These scores result from a scenic driving road, statewide rare species, and nearby recreational facilities—although the same trans-island road that provides viewing opportunities for drivers and passengers also harms many species. Currently, there are no globally rare species on the island, and the island setting prevents most terrestrial spe-

cies from immigrating. Therefore, the value of passive use services, while not evaluated explicitly, is likely not exceptional at this location.

The relative level of risk for the bank and the impact sites may be one of the critical issues for examining trades. For mitigation banks that are being created through restoration, the issue of future performance (i.e., functional level achieved) due to management actions must be considered. For example, the LPI site is currently overrun with invasive species which are being removed in stages as the bank credits are sold. If, for some reason, the bank is not completely bought out and the invasive species remain, the risk to functions and services from re-infestation could be considerable. However, various mechanisms (e.g., insurance bonds) are typically used to manage risk in these situations as is being done on LPI. Other types of risk were judged to be quite low since the property is state-owned and is set aside as a preserve.[78]

VI. Lessons for Future Research

As the first application of a theoretical model to a real-life problem, we anticipated a number of challenges and were not disappointed. By identifying these challenges we hope to facilitate continued development of a standardized wetlands valuation methodology. Below we list the major issues that require further refinement.

A. *Functional Assessments*

Because site scoring information was incomplete or inconsistent between sites used in LPI trades, we ignored potential differences in the functional capacity at various sites. Instead, we assumed that the same level of functional capacity was gained and lost as a result of each mitigation trade, and that the only differences occurred as a result of where functions were provided. Differences in the level of functional capacity at different sites could, however, be at least as important as differences in landscape factors in determining wetland value. Therefore, considering the gains and losses in functional abilities could have significantly changed our results.

78. An important indicator we did not include was whether the land was publicly or privately held. Government ownership of the LPI mitigation bank tends to imply low risk of land conversion and accessibility for recreational uses, although, at present, a fence prevents access to the site.

The ability to carry out this type of analysis would be greatly improved by developing datasets that reflect functional assessments scores from assessment systems that have been standardized and applied uniformly to all wetland sites being compared. Without comparable assessment methods being applied to all wetland sites, we cannot judge whether services that could be performed are, in fact, being performed at various locations.

B. *Scale Issues*

All of the trades we examined fell within a single county and we limited data collection to that county in order to facilitate data accumulation. Unfortunately, however, truncating our analysis at the county boundary caused biases in certain indicators. For example, indicators that were calculated based on the land use in a circular neighborhood around a site tended to be lower for areas where data for the complete circle were not available because the site was near the edge of the county. Coastal wetlands also tended to have some downward bias for the same reason. This truncation problem did not occur in indicators that were based on some larger analysis regions (e.g., census blocks) because those indicators used average or total values for the regional unit. These data, though, may be unrepresentative of the immediate area around a wetland site (which may or may not be a problem depending on what the indicator represents).

Additionally, cutting off analysis at the county boundary could have biased indicators of service scarcity. A service that appears scarce within the county may not be scarce when a larger region is considered, and that larger region may be accessible to the population within a county. Also, we only considered the portion of the watersheds within the county to evaluate scarcity of water resources, but these watersheds extended beyond the county borders and therefore route water resources from outside the county into the area. Including data for areas larger than the county would have allowed us to better understand the setting of the county in terms of neighboring resources.

Further work is needed to more adequately address what constitutes a service area over which goods can be considered either scarce or substitutable. We also need to develop an understanding of what are acceptable substitutes by further studying the effectiveness of human-made substitutes such as storm water management devices.

C. *Data Availability and Applicability*

The SFWMD was wonderfully cooperative in providing information for this study, and the data distributed by the University of Florida GeoPlan Center were invaluable. We found, however, that comparable agencies in other locales are not equally willing and prepared to distribute information. To facilitate analyses such as these, data need to be organized, standardized across regions, and made available to researchers. Despite excellent data availability in Florida, we were missing critical data, particularly for scarcity and risk analyses in several cases.[79] For example, an important risk factor that could not be easily included was risk to drinking water from saltwater intrusion. More specific information about pumpage rates at particular well fields would have allowed a spatial comparison of that risk, particularly if it could be combined with aquifer characteristics. While more and better data would obviously improve the analysis, the cost of improving data is not always justified by the gains in indicator precision. Future research will need to determine the acceptable level of precision for these types of indicators and whether focusing on a limited number of relatively robust indicators is sufficient to make reliable wetland comparisons.

The biophysical data for wetlands are currently collected as a matter of course when applying the standard assessment methodologies (e.g., HEP and WET). This makes determination of functional capacity indicators relatively straightforward. To impute the value of the wetlands, however, requires social and economic data that are more difficult to come by. Although people are required to keep track of and report economic information for a wide variety of reporting uses (whether it be tax returns or SEC filings), the data necessary for valuing ecosystems are not collected specifically for this purpose. As a result, we had to develop site-specific indicators by relying on data generated for other purposes. As the biophysical and socio-demographic data are combined, data that may have been averaged over different spatial units (census block groups, watersheds, counties) will be brought together and compared even though some data more accurately represent conditions near the site than others.[80] These data scaling issues are

79. Fine-scale data such as tax assessment maps, for example, need to be made available, preferably in a format appropriate to GIS software.

80. The more homogeneous the area around a geographic point of interest, in our case a wetland site, the more likely it is that the values related to the area will remain

certainly not new, but further examination of the influence of aggregation on results is required. The key challenge, then, is to develop appropriate data at appropriate scales.[81]

D. *Umbrella Indicators*

With further analysis of the relative contribution of individual indicators to value, we hope to find some variables that can act as screening variables (e.g., when a globally critically endangered species is present), and preclude the need to collect multiple variables for each sub-indicator. Such "umbrella indicators" were also used as part of the Exxon Valdez restoration effort. Rather than creating many indicators to assess the health of the marine ecosystem, the restoration plan used a small number of indicators that "bundled" many others. For example, the bulk of the oil spill research effort was directed at studying and restoring individual species, and the Restoration Plan outlined restoration strategies for twenty-four different species or categories of organisms.[82] Restoring habitat for the most sensitive species incorporated the requirements for restoring the area's ecosystems services.[83]

E. *Local Knowledge*

Interviews or more extensive collaboration with local management officials could have provided a deeper understanding of the important local issues and easier access to data to improve indicators. The influence of some data that were not readily available to us, such as zoning and other restrictions, would have allowed us to quantify some critical scarcity and risk indicators. For example,

---

consistent as the scale of analysis changes. In general, smaller areas will tend to contain more homogenous conditions while larger areas will tend to mask heterogeneity over space.

81. A related challenge is to develop data that allow comparisons between regions locally and nationally. An example of data that support this goal is the EPA web site that examines watershed indicators nationwide. U.S. EPA, *Index of Watershed Indicators, at* http://www.epa.gov/iwi (last visited Apr. 16, 2001). However, states could also develop and publish data that compare counties in terms of natural resources. Understanding the county's role in providing services would help to interpret the standing of indicator values estimated for the county. For example, if the county holds the last remaining habitat in the state for a species, that information would help to judge the importance of a scarcity indicator, even if the value was the same (e.g., all showed low scarcity) for the population of sites being compared.

82. EXXON VALDEZ OIL SPILL TRUSTEE COUNCIL, 1997 STATUS REPORT 4 (1997).

83. James Salzman & Juge Gregg, Ecosystem Services and the Exxon Valdez Restoration (2001) (unpublished manuscript, submitted to EPA as part of EPA grant R82612-01, on file with the author).

water supply does not appear to be an immediate problem in Lee County, but an understanding of expected growth rates and planned development in the county would improve the understanding of the importance of the contribution of any given wetland to drinking water supply. Some adjacent counties are developing expensive water supply projects such as desalinization plants—understanding the population and development levels that necessitated such undertakings could help Lee County make decisions based on an understanding of the costs involved with the incremental loss of wetlands.

F. *Scoring Methods and Index Development*

The indicators explored in our analysis are bits of information that in some way identify the value of a wetland site's services. Individually, the indicators tell us something about whether the service is present, its relative magnitude, and the benefits generated by a service at a particular location. Compiling the indicators paints a more coherent picture of wetland value and provides a much richer sense of the landscape-driven service value of wetlands than was apparent with any individual indicator.

The method we used to assign ranks to indicator values is relative. The value of a site depends on the set of other sites being examined. We used our relatively small data set to define the distribution of which sites would be considered "good" for a particular indicator, or not. We chose to use the range of measured scores as the full range of values for scoring, though further thought needs to be given to the relevant population used for determining the distribution and whether the differences between sites are important given the variability of the population.

Additionally, we have not adequately dealt with the issues of index construction (the combination of sub-indicators into a single index) and indicator aggregation. A composite index would be desirable for a variety of reasons. By their very nature, composite indices summarize a complex array of data and rankings into a smaller set of more easily digested information. In principle, the indicators we developed can be aggregated to derive a single number indexing a site's wetland value. In practice, aggregation is laden with methodological difficulties although new techniques are being actively investigated.

Aggregation procedures are naturally subjective. For an aggregate index to have institutional validity it must be constructed with

a transparent methodology. In general, the validity of a given methodology is often determined by its ease of use, its ability to help interpret masses of data, and its broad scale acceptance. A sound scientific basis that considers a range of social viewpoints should guide the judgments made in the index's construction.

Perhaps the most fundamental issue is the degree to which aggregation is pursued. The more aggregation, the greater the simplicity and interpretability of the final result. The tradeoff is that aggregation obscures potentially valuable data and insights regarding the sites. Composite indices are averages of sub-indices, and the single values that they produce may conceal inconsistencies between the individual components or sub-indices. More condensed (aggregated) data may be more easily presented and digested by decision-makers but less condensed data have greater scientific utility. The question is, how should the appropriate balance of complexity and simplicity be achieved?

A set of relevant indicators will often exhibit redundancy or cross-correlation. For example, in Table 4 (scarcity of drinking water), the percentage of the watershed in wetland and the percentage of the watershed in agriculture are almost perfectly inversely correlated. These indicators are measuring complementary things since more wetland (or more natural area) will tend to mean less agricultural area. An aggregate index must ultimately cull redundant or correlated indicators that provide relatively little additional information regarding service values.

An understanding of indicators' comparability is necessary for them to be meaningfully aggregated. For instance, a simple additive weighting model requires that indicators be standardized (i.e., normalized) to common units. Standardization ensures that equal changes in the level of different indicators reflect equal changes in the social value they measure.[84]

Sub-indicators used in developing composite indices must be weighted to arrive at a meaningful composite score. Too often, the specific weighting used is implicitly presented, rather than explicitly derived. For example, a simple average across indicators implies that each indicator indexes social value equally. This will almost never be true. Instead, different weights should be assigned to different indicators and sub-indices based on statistical analysis

84. *See generally* Scott G. Leibowitz & Jeffrey B. Hyman, *Use of Scale Invariance in Evaluating Judgement Indicators*, 58 ENVTL MONITORING & ASSESSMENT 283 (1999) (discussing the use of standardization to compare ecological end points).

and techniques designed to illuminate relative social preferences for the various services.

To create a composite score of wetland value, it is not just the sub-indicators and indicators that must be combined, but also the individual services must also be weighted. Indicators representing a single service value allow for an unambiguous ranking of sites only when the sites are identical in all respects but the one influencing that service. But, as was the case in our study, sites will be better than others in certain respects and worse in others. When this is the case, the overall ranking must consider the different weights given to service values derived from a set of different services. Is a site that provides more effective flood prevention preferred to one that more effectively supports drinking water supply? These kinds of tradeoffs are unavoidable.

Ultimately, there is no substitute for using some method to elicit public preferences and to assign relative weights to specific ecosystem services. Ranked preference surveys can be used to assign relative weights to services. Such surveys can be simple and inexpensive and can yield defensible indicators associated with individual and community preferences for ecosystem services. As an example, respondents can be asked to express the intensity of their preferences for one service over another by ranking pairs of services on a 1-5 scale (equal importance to absolute importance). Various statistical methods can be applied to the results of paired preference surveys to arrive at relative service weights or rank orderings of services.[85]

Despite their necessity for making tradeoffs fairly among user groups, value comparisons across services create the same problems as any non-market valuation exercise, because revealed values are unavailable.[86] There is no entirely satisfactory method by which "true" social preferences can be elicited. Because a significant portion of the regional economy in Florida is based in tourism, people may have quite different perspectives about whether

85. *See* Paul Smith & John Theberge, *Evaluating Natural Areas Using Multiple Criteria: Theory and Practice*, 11 ENVTL MGMT 447 (1987).

86. Revealed values are those that are measured by observing what people pay directly for goods and services or by observing their actions. Techniques such as hedonic pricing or travel cost are used to reveal the dollar amount that people pay for environmental amenities. The environmental values measured include what an individual is willing to pay to enjoy an environmental amenity (e.g., the additional amount they are willing to pay to buy a house in an area with clean air) or their willingness to accept compensation for the loss of an amenity.

their economic wellbeing depends on maintaining aesthetics and recreational fishing opportunities. Direct surveys of the preferences people have for various wetland services, and how they believe their livelihoods and quality of life depend on them would provide more evidence to help define tradeoffs among wetland services. Particularly instructive might be differences in service preference weights assigned by sample respondents selected from populations at different geographic scales. It may also be useful to have services scored from particular points of view (e.g., developer, wetland scientist, water engineer, economist, resident, tourist, etc.) and examine the degree of consensus among the viewpoints.

G. *Develop Information to Quantify Risk to Wetland Functions or Services*

Sophisticated techniques have been developed to assess risk, but these techniques rely on making many simplifying assumptions about the systems under study that are often not appropriate for evaluating risk to ecosystem services.[87] Risk in complex ecological systems is not easily quantified, but the variety of techniques available to deal with aspects of complex systems such as non-linear behavior (e.g., thresholds) continues to grow. We developed indicators for a few known risk factors, but the exact relationship between such issues as nutrient inputs and loss of function is not well established. It is not realistic to expect to find simple relationships given the complexities of natural systems, but continued research into ecosystem collapse and recovery will continue to improve our risk assessments, as will new techniques for dealing with uncertainty.

VII. CONCLUSIONS

In this study, we set out to demonstrate how location-specific information could be used to improve decision-making regarding wetland mitigation banking by including objective measures of economically important variables. We sought to explore the feasibility of a landscape-based wetlands valuation system. This exploration required laying out a clear system of indicators that represent significant values, identifying data that were capable of measuring

87. *See, e.g.*, U.S. ENVTL. PROTECTION AGENCY, *Health Risk Assessments, at* http://www.epa.gov/ncea/healthri.htm (last visited May 3, 1999).

these indicators for each site, and combining the data into a form amenable to policy decisions.

We were able to show how certain aspects of available geo-referenced data sets could be used to evaluate the importance of wetlands at particular locations and thereby infer value from landscape characteristics. These location-specific characteristics indicated the wetland's relative advantage in providing valued services, and revealed when scarcity or risk of service disruption tended to adjust the value of a wetland's services upward or downward.

We stopped short of quantitatively evaluating relative value for three reasons. First, we did not have consistent or reliable data for all impact sites on the level of function (though we note that much of this information is gathered as a matter of course for current wetland trades). Second, we did not have sufficient information to weight indicators in terms of their importance to social value. Finally, we did not have sufficient information to weight services relative to one another (although this was more a function of the limited budget and pilot nature of this project than analytic difficulty). Our difficulties demonstrate that there are inconsistencies in the way wetland functions are evaluated for trades and present some interesting challenges for incorporating scientific understanding and public preferences. However, they do not present insurmountable obstacles to implementing such a system.

Our results show that some sites were ranked about the same (high or low) in relation to the other sites regardless of the service being addressed. This outcome implies that determining the relative weights (i.e., social preferences) for different services may be relatively unimportant for assigning relative wetland values in this particular case study. In other mitigation trades, this result may not hold. On the other hand, understanding the relative contribution of types of indicators to the overall value of a site is likely to be extremely important for distinguishing between sites because these indicators vary widely with location. To make these indicators useful, though, requires that the weighting factors used when combining sub-indicators be carefully assigned so that each indicator reflects a comparable contribution to a site's social value.[88]

While our system may not yet be at the "easy-to-use" stage, we hope that it provides a useful framework for making land use plan-

88. In particular, risk factors were important in reducing the value of one site that otherwise would have rated highly (Site 4).

ning and conservation priorities more explicit and transparent and, thereby, more easily reviewed by the public and discussed among interested parties. Determining the relative importance of indicators, particularly risk factors, to wetland value will undoubtedly be quite challenging. Yet, the fact that we are basing value on easily observable characteristics provides the solution to the problem. The indicator value method we describe offers a way to extend the usefulness of existing economic studies by breaking ecosystem service value into measurable component parts. This disaggregation of the landscape components that contribute to value should allow information about one wetland to be transferred to other wetlands with similar characteristics. Adequate characterization of the wetland in terms of its landscape setting would need to be added to existing economic studies to forward this goal. With enough information, a conjoint, or other analysis,[89] of wetland characteristics and their relationship to expressed values could allow us to devise weights for some of the indicators used here.

In spite of the difficulties, we believe this method warrants further development. The explicit recognition of the landscape characteristics that confer values to wetland services in various locations can make an important contribution to understanding and maintaining the aspects of wetlands that people value the most. Many of the difficulties we encountered could be overcome in time, particularly if economic studies were developed to address some of the issues raised by this demonstration.

The indicator system provides a new and more inclusive means for considering land use tradeoffs. The wetland functional indices used traditionally for wetland "valuation" fail to address many of the most critical aspects of value. The system developed here helps to reveal the value of wetlands at particular locations and offers the ability to make decisions based on both sound ecological and economic understanding, thereby providing a basis for more effective and efficient trade in ecosystem services.

89. *See, e.g.*, M.D. Morrison et al., *Valuing Improved Wetland Quality Using Choice Modelling, in* U. SOUTH WALES, CHOICE MODELING RES. REP. No. 6 (1998) (discussing the use of "choice modeling" to quantify non-use environmental values of the Macquarie Marshes).