Habitat Benefit Assessment and Decisionmaking: A Report to the National Marine Fisheries Service

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Abstract

Habitats and the services they provide are part of our nation's portfolio of natural capital assets. As for many components of this portfolio, it is difficult to assess the value of their services, and this complicates regulators' investment decisions, especially when the alternative use is measurable. The principal objective of this report is to suggest possible strategies for the National Marine Fisheries Service (NMFS) as it applies economic analyses and arguments in support of the agency's trustee responsibilities. Many approaches are possible, and as we discuss, the "right" strategy will depend on the questions asked, the resources available, and the agency's role in the consultation process. We discuss in detail bioeconomic modeling and ecosystem indicator approaches to habitat value assessment. Although the approaches are discussed independently, multiple tools could be used simultaneously across different regions or within the same region on different aspects of one consultation.

Key Words: Bioeconomic, ecological indicators, ecosystem services.

JEL Classification Numbers: Q20

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1. Introduction

Understanding the socioeconomic benefits and costs of decisions that affect the condition of aquatic habitats is critical for rigorous decisionmaking, especially when resources are limited. The principal objective of this report is to suggest possible strategies for the National Marine Fisheries Service (NMFS) as it applies economic analyses and arguments in support of the agency's trustee responsibilities. Many approaches are possible, and as we discuss, the "right" strategy will depend on the questions asked, the resources available, and the agency's role in the consultation process. To provide focus, we will refer mostly to essential fish habitat (EFH) consultations, in which NMFS is called upon to represent the social interest in habitat resources important to the nation's marine fisheries.

We develop and propose general themes to guide the agency as it seeks to make effective economic arguments for any given consultation. First, we urge a focus on the services generated by habitat. This requires a linkage between biophysical analysis and socioeconomic analysis. Second, we encourage an explicitly spatial approach to economic assessment. Third, we urge NMFS to rationally expend its limited assessment resources.

We acknowledge that addressing these themes is a challenge. Aside from the unanswered scientific questions, a significant difficulty in applying sophisticated economic logic and principles is the limited data and resources available to NMFS staff. Toward that end, we urge that resources be invested in a watershed or regional-scale analysis in advance of any particular decision. Such large-scale analysis then can be the foundation for subsequent consultation comments as actions that can affect habitat in the region are proposed in the future.

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1.1 Report Organization

In Section 2, we characterize the NMFS mission as a trustee in habitat consultations. We draw a distinction, for purposes of this report, between consultations and situations in which NMFS has the principal, if not absolute, decisionmaking authority, as under the Endangered Species Act (ESA). In the consultation role, as illustrated by essential fish habitat reviews, agencies with different missions are the final decisionmakers and can consider but need not abide by NMFS recommendations. Therefore, the goal for a NMFS consultation on any decision (a permit or investment decision) is to advance arguments favoring the single objective of habitat protection, including compensation for habitat losses that occur. The responsibility of the other agencies is to accommodate NMFS habitat protection and loss compensation arguments in recognition of the legal requirements governing their programs and the need to balance habitat protection against other objectives.

In Section 3, we briefly summarize the multiple meanings of "economics" in habitat decisionmaking and narrow the focus of this report to economic assessment of environmental benefits. The section also describes the foundations of economic valuation, nonmarket valuation techniques, and the reasons academic economists favor formal—usually econometric—estimation methods.

Section 4 discusses trade-offs associated with the use of monetary-benefit assessment tools. We comment on the cost of monetary-valuation studies, and we summarize the critique of monetary estimation as a credible measure of value relevant to public choices. The section also includes an overview of regulatory decisionmaking contexts and discusses the types of assessment tools used.

In Section 5, we explore the complex relationship between biophysical analysis, institutional conditions, and economic valuation of habitat services, paying special attention to spatial analysis. Using formal models with both biophysical and economic features, we illustrate causal linkages between habitat, ecosystem functions, and services derived from those functions. We conclude the section with an interpretation of the concept of cumulative effects—which we suggest be called "threshold effects"—and describe the economic case for minimizing the likelihood of threshold effects. This section illustrates concepts that are fundamental components of any rigorous ecological benefit assessment. The section also provides concrete examples of bioeconomic modeling and its role in habitat benefit assessment.

In Section 6, we discuss an ecosystem benefits indicator approach that is similar to the use of ecological indicators that summarize biophysical characteristics. This approach requires a spatial focus. However, as we demonstrate in the section, place-specific spatial analysis will require an up-front investment in the development of region-wide EFH habitat plans. The term "plan" is used here to describe decisionmaking rules that will be used to define the location of protected habitat and the compensation requirements for the habitat that is altered. In effect, we propose that NMFS support its case-by-case consultation comments with system-scale analysis done in advance of any particular case. We also illustrate how an ecosystem indicator approach would support the development of special area management plans (under the Coastal Zone Management Act) and wetlands-watershed plans under the administration's mitigation action plan.

Section 7 concludes the report by highlighting our main points regarding the use of economic arguments in consultation decisions and our recommendations on the investments in benefit assessment research at NMFS.

2. The Consultation Mission

Habitat alterations are intentional changes made to the physical structure (e.g., placement of fill) or function (e.g., diversion of flow) of the coastal landscape. Such changes are made in conjunction with valuable transportation, housing, or other development; however, these changes also may threaten valuable environmental services. The term "environmental services," as used here, has a decidedly economic content. Specifically, it refers to services valued by people arising from biophysical functions. The value of these services may not always be reflected in markets.[1] Natural resource trustees, including NMFS and other federal and state agencies, are often expected to participate in public decisionmaking to represent society's interest in environmental services.[2] The trustee mission is supported by laws governing NMFS authority to act in a consulting capacity. More specifically, NMFS is called on to advise decisionmakers in other federal agencies who have primary authority to issue a permit or make an investment

[1] Whether such values are represented in market exchanges, the reasons why they would or would not be, and the means to solve such "market failure" problems are matters of professional and policy debate beyond the scope of this paper. See Carl Dahlman, The Problem of Externality, *Journal of Law and Economics*, 1979, for a rigorous discussion of the history of the market failure concept; and Richard O. Zerbe and Howard McCurdy, The Failure of Market Failure, *Journal of Policy Analysis and Management* 18(4): 558, 1999, for a less technical review.

[2] The public trust doctrine is the common law foundation upon which the concept of trusteeship is based. Joseph Sax, The Public Trust Doctrine in Natural Resource Law: Effective Judicial Intervention, *Michigan Law Review* 68: 471, 475, 1970.

decision. NMFS, in its role as a natural resources trustee, is charged with representing the social interest in habitat and its functions and services.

This report is organized around a particular trustee responsibility: the National Marine Fisheries Service consultative role under the Magnuson-Stevens Act. The act requires the identification of essential fish habitat for federally managed species. Any federal agency undertaking or approving an action that may have an "adverse effect" on EFH is required to consult with NMFS. An adverse effect is defined as "an impact which reduces quality and/or quantity of EFH…[and] may include direct (e.g., contamination or physical disruption), indirect (e.g., loss of prey, reduction in species fecundity), site-specific or habitat wide impacts, including individual, cumulative, or synergistic consequences of actions."[3] Other impacts that may cause adverse effects include functional changes to estuarine resources, wetlands, mudflats, seagrass, or shellfish beds. Water quality changes, such as increased turbidity, flow alterations, or nesting and breeding habitat disruptions, may also be adverse effects. Many different types of actions will trigger the consultation requirement. For example, public infrastructure projects, such as harbor dredging, road construction, or water supply reservoirs, may require Section 404 permits or an environmental impact statement (EIS) under the National Environmental Policy Act (NEPA) of 1969.

Other agencies may have permitting or investment authority that will result in habitat alterations. However, as a trustee, NMFS should make the most compelling argument for these interdependent goals: 1) protection of aquatic habitat from alteration, by avoiding habitat disruption to the maximum extent practical, and 2) mitigation of the adverse consequences of habitat that is altered, paying special attention to minimizing the possibility of unexpected loss of habitat services from a series of seemingly small incremental habitat alteration decisions. Different decisionmaking contexts will provide different opportunities to make these arguments and to secure these goals. For example, the Section 404(b)(1) permitting guidelines call on permit applicants to avoid placement of fill in waters and, if avoidance is not possible, to minimize and then compensate for losses. NMFS comments on specific permits might then be focused on the extent to which avoidance is practical and whether the losses are fully compensated.

[3] Essential Fish Habitat: New Marine Fish Habitat Conservation Mandate for Federal Agencies, A Primer for Federal Agencies, NMFS, December 1998, 6.

The case for protection and compensation must be credibly argued within the analytical framework and legal authorities governing the primary federal agency. Interagency consultations, such as those arising from the Magnuson-Stevens Act, involve a kind of bargaining among agencies with distinct mandates and missions. Following NMFS comment on a proposed action, agencies may be required to formally respond to NMFS recommendations but are not required to accept those recommendations in their entirety. Therefore, the habitat goals promoted in the consultation may not be fully accommodated by the primary agency. In trying to make the most effective case, NMFS may choose to employ "economic arguments" in its consultation comments and in promoting program design reforms. For example, when habitat alterations reduce environmental services, trustees can seek to "make the public whole" in an economic, not just a biophysical, sense, if the legal framework calls for such compensation.[4]

3. Economic Argument, Benefit Monetization, and the NMFS Mission

This section describes the foundations of economic valuation, nonmarket valuation techniques, and the reasons academic economists favor formal—usually econometric—estimation methods.

"Economic argument" is a term we want to define with care. Economic arguments can take a variety of forms, reflecting diversity and variety in economics as a discipline. Moreover, decisionmakers themselves have different conceptions of what constitutes economic analysis. Some envision economics as describing the ways that jobs and incomes are affected by an investment or a regulation. Others imagine economics as being the calculation of the financial cost of taking an action. Others think of economics as the analysis of government revenues and expenditure programs. Still others think of benefit-cost analysis.

Professional economists engage in all these different forms of analysis, and economic arguments are made in all the ways described above. More specifically, there are differences among economic impact analysis, economic efficiency analysis (more commonly understood as benefit-cost assessment) and replacement cost analysis (used for measuring loss compensation). Each form of analysis has its own independent logic and computational requirements. In general, the foundations and tools of economic argument differ, depending on the kind of argument being

[4] The goal of natural resource damage law, for example, is to "make the environment and public whole" following a pollution event (15 C.F.R. §§990.53).

made. In this report, we focus on economic arguments that are intended to represent and aggregate people's preferences for different habitat conditions. We further narrow the discussion to economic analysis of the ecosystem services provided by habitat.[5] We define "habitat valuation" as the ways NMFS can represent people's preferences for such services.

3.1 Biophysical Analysis: The Foundation for Economic Argument

Habitat valuation rests on the biophysical assessments that are familiar to NMFS scientists. Ecosystem structure and functions, as described and evaluated by ecological science, generate services that people value. It is the services created by ecological characteristics that will be explicitly tied to social value.[6] The distinction between ecosystem structure and functions and ecosystem services can be defined in the following way: ecosystem services are the outcomes of ecosystem functions that yield value to people. For example, the ability of wetlands to mediate extremes of flood and drought at a downstream location is a biophysical function. A service is created if the absorbed floodwater causes less damage to buildings, roads, and agriculture or if the higher flows in limited rainfall years support a recreational fishery. Even if an ecosystem rates highly in terms of a functional characteristic, that function may not provide a socially valued service.[7] As another example, any consideration of the value of lost commercial and recreational fishing opportunities due to an oil spill in coastal wetlands must understand the role of the wetlands in fishery population dynamics.

Therefore, an important challenge for NMFS is to support ecological assessments that link well with the agency's economic assessment needs. The generic challenge is that ecological analysis must generate models and data that are useful for making economic valuation arguments. Consider one important manifestation of this challenge: ecological analysis of impacts may focus on the effects of pollution or habitat degradation on individuals (e.g.,

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[5] For an example of economic impact analysis, see Socioeconomic Impacts of Marine Reserves, http://marineeconomics.noaa.gov/reserves/welcome.html.

[6] We recognize that people may value a natural system as an end in itself and not for utilitarian service flows from the asset. Sometimes this is termed intrinsic value. See Mark Sagoff, *The Economy of the Earth,* Cambridge, 1988; and Mark Sagoff, Can We Put a Price on Nature's Services? *Philosophy and Public Policy* 17(3), 7–12, 1977.

[7] Perhaps a more direct way to make the point is to consider functionally identical ecosystems. Functional equivalence does not imply equivalent social value. Wetlands with an equivalent ability to remove nitrates from groundwater or absorb floodwater pulses will nevertheless differ in their social value because the number of people whose drinking water is purified and the number of homes protected from flooding will not be identical.

measured contaminant loads in fish tissue) of a species. Economic arguments typically require ecological analysis of effects on species populations, not analysis of individual effects. The reason population effects are the desirable endpoint for ecological analysis is that populations, rather than individuals, are what is actually economically valuable. Generally, people value the ability to observe, appreciate, fish, or hunt a population.[8] The size and condition of that population determines the value of the service the population provides. A collection of dead or sick individuals cannot be valued unless the effect of that mortality or morbidity on the overall population is known. Accordingly, improved economic analysis within NMFS may also require changes in the way ecological assessments are conducted.[9]

3.2 Representing Preferences: The Logic of Valuation

We begin the discussion of economic valuation with an admittedly simplified description of market exchange and its role in economic thinking. In a market economy, if person A owns[10] an asset,[11] then person B cannot enjoy the services of that asset without the willing consent of A. That consent usually requires a payment from B to A and the transfer of the asset or service from A to B. Exchange takes place if the compensation B offers to A is greater than the value A ascribes to the asset or service.[12] Of course, the value of the asset to B must be at least as great as the compensation paid. Economists argue that the exchange thereby reveals something of the asset's value to the parties to the exchange.[13] It is for this reason that economists use observed market prices to estimate economic value. The price of a good or service is its value to the consumers of that good (represented by the income they could have spent to secure other goods

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8 Rare or endangered species are an exception, since individual effects are closely related to the effect on the population.

9 We further discuss these modeling needs in Section 5 of this report.

10 By ownership, we mean that A has the legal authority to determine the use of and access to the services of an asset.

11 By asset, we mean all manufactured goods, services, and resources, such as land and labor.

12 More generally, we might say that A's willingness to sell is based on the "utility" A expects to forgo by no longer owning the asset, and that B's willingness to pay is based on the "utility" B expects to derive from taking ownership of the asset.

13 In this simple example, the possible exchange values are bounded by A's willingness to sell and B's willingness to pay. The actual exchange value will depend on the bargaining power of A and B. In a more competitive market, however, the exchange value will be determined by the willingness to pay of the marginal buyer.

or services). The price of a production input (cost), such as a gallon of fuel, is the value that input would have produced if dedicated to its next best use.

However, although market exchange is the touchstone of economic thinking, not all assets are exchanged in well-functioning markets.[14] When a "market failure" occurs, accurate price signals are not available to discipline and guide resource allocation. Instead, government regulations or other policies are used to make resource allocation decisions. As has been noted, it is the assertion of market failure that creates the trustee responsibility for NMFS in its habitat consultation role. The premise is that the services of habitat are not represented in market exchange, and it is thus the responsibility of NMFS to make that representation. For the representation to be an "economic" argument, the benefits people derive from habitat services must be considered in relation to the social benefits that are realized from the proposed habitat alterations. Furthermore, depending on the initial ownership rights implied by the laws, compensation may be required that would be equal to the value of the services lost. The NMFS consultation mission is to represent the value of habitat services and perhaps secure compensation for values that are lost through an alteration from those who benefit from that alteration.

3.3 Representing Preferences: Benefit Assessment

At the broadest level benefit-cost analysis is the representation of positive and negative consequences of a public action using a common monetary metric. For this reason, some professional economists seek to measure people's preferences in monetary terms in order to aggregate the preferences using the common metric—dollars. Drawing upon the logic of market exchange, professional economists advance willingness to pay (and sell) as the conceptual foundation for the value of environmental services. When focusing on environmental services, this is called "environmental benefit assessment," or "nonmarket valuation." Benefit assessment seeks to represent the value of environmental services to people so that those values can be analytically compared with values from the habitat alterations that compromise environmental services.

In fact, agencies have long weighed and translated into dollar metrics the benefits and costs of alternative actions. However, there also has been a reluctance to extend such monetary

[14] See note 1, above.

calculations to intangibles, such as environmental services gained or lost through habitat alteration or restoration. Many economists now argue that the theories, data, and analysis methods have been developed to a point where it is practical to place a money measure of value on environmental services.[15] While acknowledging that environmental benefit assessment is professionally challenging and may be costly, they argue that without such assessment, environmental services will be underrepresented in the decisionmaking process.[16]

3.4 Economic Argument and Compensation

In the laws and implementing regulations governing its trustee role, NMFS may be able to assert that the public has a right to a continuation of compromised habitat services. A corollary to this right could be that those who receive benefits from habitat alterations need to provide compensation for the opportunity costs (forgone benefits) the alteration imposes on others. Of course, this is the way that markets work. A resource is not transferred from one use to another unless the owner of the resource in its original use is paid for the ownership right. Compensation payments are the heart of market exchange and are a discipline on the demands made on resources to ensure that the resource goes to its highest and best use.

In fact, the consultation analysis and interagency bargaining most often turn on the adequacy and form of the compensation and on the reasonableness of the costs—including a judgment that the costs are not wholly disproportionate to the opportunity costs (environmental benefits forgone) from the proposed habitat alteration. Thus the burden on the trustee may be to temper its replacement requests on one case and instead seek larger-scale compensation programs that replace habitat services in the most cost-effective way. We will argue that this is best done with explicit attention to the consultation's spatial context. For this reason alone, the focus on physical replacement does not preclude the need for economic valuation arguments. As we will argue, the spatial location of the replacement may mean that one or more of the services will not be compensated.[17] We will also argue that a spatial approach is the foundation for examination and minimization of "threshold" (or cumulative) losses.

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[15] A. Myrick Freeman, The Measurement of Environmental and Resource Values: Theory and Methods, Resources for the Future, 2003.

[16] See Michael R. Moore, Elizabeth B. Maclin, and David W. Kershner, Testing Theories of Agency Behavior, *Land Economics* 77(3): 423–42, August 2001, for an example of this argument.

[17] In addition, there may be a temporal loss of services and hence value.

3.5 Benefit Monetization Tools

Economists over the past 20 years have been experimenting with methods to estimate the dollar value of nonmarketed ecological services. The development of these techniques is continuing, but their application to date is not without controversy.[18] Attempts at nonmarket valuation fall into three general categories: revealed preference, expressed preference, and derived willingness to pay.[19]

Revealed preference studies look at the price people pay for marketed goods that have an environmental component. From those prices, one can make inferences about the environmental benefits associated with the good. The use of market prices is demonstrated in Section 5, which discusses the value of habitat to commercial fishing. Another example is that the prices of homes located near an aesthetically pleasing habitat with access to recreation reflect the value of the aesthetic and recreational services realized by the homeowners.[20] Alternatively, for people who do not live near the site, recreational and aesthetic services can be valued by the time and money spent traveling to the area. These direct and imputed travel costs reveal a willingness to pay for the recreational services.[21] Differences in quality attributes can be valued if there are perceptible differences in the number, length, or cost of trips taken to sites of different quality or sites that have undergone improvements.[22] Consider another example. Sediment, nutrient, or pathogen trapping that affects swimming beach quality at a remote location can be valued if the relationship between the habitat of the remote site and the recreational quality of the beach can be estimated.

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18 Northeast-Midwest Institute and National Oceanic and Atmospheric Administration, Revealing the Economic Value of the Great Lakes, 2001: "The development of natural resource damage assessment regulations was controversial because stakeholders disagreed over what damages would be assessed, how damages would be calculated, and how damages to environmental goods and services not valued in traditional markets would be calculated."

19 For a good overview of these methods, see Freeman, note 15.

20 Hedonic analysis is used in this type of study. See, e.g., Brent Mahan, Stephen Polasky, and Richard Adams, Valuing Urban Wetlands: A Property Price Approach, *Land Economics* 76: 100, 2000. For an application of the same approach to calculating the value for filling urban wetlands, see Leonard Shabman and Michael Bertelsen, The Use of Development Value Estimates in Wetland Permit Decisions, *Land Economics* 55(2): 213–22, May 1979.

21 There is a substantial body of literature on this subject. See, e.g., Kenneth McConnell, On-Site Time in the Demand for Recreation, *American Journal of Agricultural Economics* 74: 918, 1992.

22 An important issue in travel cost studies, for example, is the definition of relevant substitutes for the sites in question. See Northeast-Midwest Institute and NOAA, note 18, at 73: "omitting the prices and qualities of relevant substitutes will bias the resource valuations."

An expressed willingness-to-pay study asks people, in a highly structured way, what they would be willing to pay for a set of environmental improvements. Contingent valuation studies and contingent ranking are examples.[23] Surveys of expressed willingness to pay are expensive and controversial, and answers to questions may be affected by the specific context.[24] The more complex the habitat change and its consequences, the more difficult the challenge of survey design and interpretation. A principal drawback to this approach is the risk that people may misunderstand the precise service being valued when undisciplined by the need to spend their own money. Also, respondents may not have fully formed preferences for the service. For both of these reasons, they may misstate their willingness to pay.[25]

In a valuation survey, the questions are structured in such a way that a particular economic calculation can be extracted from the results—willingness to pay in terms of a hypothetical amount of one's income for a change in the state of the environment. This calculation is expected to represent the nature of an individual's preferences and is a way to aggregate those preferences over individuals in a population.[26]

For many years, researchers have also derived benefits via simulation studies based on engineering analysis.[27] For instance, if we want to know the value of a wetland for reducing flood damages, we can, in principle, estimate the dollar value of real property damage due to a flood and estimate the greater likelihood such an event will occur if the wetland is destroyed.

An important characteristic of benefit monetization studies is that they need not and typically do not monetize benefits arising from the entire suite of services generated by a site. This is true because different services typically require different assessment procedures. Valuing recreational services require one kind of study, eliciting existence values requires another

[23] See Richard Carson, Nicholas Flores, and Norman Meade, Contingent Valuation: Controversies and Evidence, *Environmental and Resource Economics* 19: 173–210, 2001, for a review and defense of contingent valuation's role in the evaluation of environmental goods and services.

[24] Leonard Shabman and Kurt Stephenson, Searching for the Correct Benefit Estimate: Empirical Evidence for an Alternative Perspective, *Land Economics* 72(4): 433–49, November 1996.

[25] See Raymond J. Kopp et al., eds., *Determining the Value of Non-Marketed Goods,* Kluwer, 1997, who present a good collection of articles relating to the contingent valuation method.

[26] If a measure of public opinion is desired to support decisionmaking, other public opinion polling approaches and calculations also might be considered. For example, a survey that asked about levels of agreement with statements about options and trade-offs may be seen as a kind of "valuation" effort.

[27] Orris C. Herfindahl and Allen V. Kneese, *Economic Theory of Natural Resources,* Columbus, Ohio: Charles E. Merrill, 1974, 252–61.

approach, and understanding individual valuation of flood prevention yet another. Contingent valuation surveys can be designed to value a wider suite of benefits, but this complicates the administration and design of the survey.

In many situations, it is not feasible to implement a survey to determine the value associated with each parcel of habitat of open space, preservation of an acre of wetland habitat, or increase in catch per unit of effort for a recreational fishery. Therefore, economists have long argued for the use of benefit transfer methods as a way to avoid site-specific monetization exercises and minimize the need for costly new data collection.[28] Benefit transfer methods essentially take the benefits estimated at a well-studied reference site and relate those benefits to the benefits likely to be found at a site of interest for preservation, mitigation, or exchange. The "transfer" of the benefits is made a function of differences in the reference site and site of interest. Benefit transfer still requires data collection and careful econometric analysis, but it reduces somewhat the burden of new data collection.

4. A Discussion of Monetary Estimation of Benefits

We now discuss some of the tradeoffs associated with the use of nonmarket techniques, which are only one tool to inject economic thinking into policy debates. Our discussion is aimed at informing decisions on when and where monetary valuation techniques might be applied by highlighting some of the tool's drawbacks. Some of the critiques are obvious, such as the often high cost of valuation studies. Others are more subtle and focus on whether it is appropriate to extrapolate values that are elicited in a survey format in one (typically hypothetical) context to social values formed in real-world decisionmaking contexts.

We also present an overview of the regulatory decisionmaking contexts that are similar in spirit to NMFS EFH consultations. Surprisingly, very few of these decisions employ nonmarket techniques to date. It is beyond the scope of this study to fully explain the reasons for the lack of reliance on nonmarket techniques. Rather, we make the observation because we believe it is important for NMFS staff to understand that these measures are controversial. The lack of

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[28] For an overview of benefit transfer methodologies, see the special issue of *Water Resources Research* devoted to this subjecct, volume 28, 1992. Also see S. Kirchhoff, B. Colby, and T. LaFrance, Evaluating the Performance of Benefit Transfer: An Empirical Inquiry, *Journal of Environmental Economics and Management* 35: 75–93, 1997, and Ray Kopp and V. Kerry Smith, Valuing Natural Assets: The Economics of Natural Resource Damage Assessment, Resources for the Future, 1995, 329.

adoption of these methods by other agencies presents a risk to NMFS. Even if NMFS did undertake a valuation study, it is unclear whether the results would be incorporated into the debate on EFH consultations by other agencies, especially agencies that do not currently use these techniques.

4.1 The Costs of Monetary Assessment

Whatever the approach, nonmarket valuation studies are complex, time-consuming, and costly. Methods seeking monetary estimates of ecosystem benefits are technically challenging, fraught with dangers that may not be obvious to nonpractitioners, and require significant amounts of data collection.[29] In other words, economic valuation methods are expensive and difficult to execute, particularly by noneconomists.[30] For these reasons, NMFS should not expect relatively small budget outlays on nonmarket valuation studies to easily translate into significant policy impact. A professionally credible study costs, at a minimum, tens of thousands of dollars and can easily cost an order of magnitude more. Although economists continue to develop "rules" of benefit transfer that will allow these costs to spread out over many projects, the transferability of studies to other locations is still methodologically difficult.

In addition to the methodological difficulties, several characteristics of the EFH administrative process should be considered before valuation techniques are employed. First, the number of consultations is significant and spread across field offices and regions. Several thousand EFH reviews must be conducted by NMFS each year.[31] Second, the Magnuson-Stevens Act does not require economic analysis of impacts.[32] Third, the EFH process is only advisory in nature. NMFS does not have the authority to delay or prohibit development or some other action based on the EFH provisions. Fourth, current funding levels devoted to consultations are

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[29] For an interesting case study of real world ecological service damage assessment, see David Chapman and W. Michael Hanemann, Environmental Damages in Court: The American Trader Case, in *The Law and Economics of the Environment*, edited by Anthony Heyes, Elgar, 2001.

[30] See Carson et al., note 23, at 196: "we believe that at this point in the development of CV, the key objective in terms of methodological development should shift to trying to determine how to reduce the cost of conducting CV studies while still maintaining most of the quality of the very best studies now being conducted."

[31] Testimony of Penelope Dalton, Assistant Administrator for Fisheries, NOAA, on The Essential Fish Habitat Provisions of the Magnuson-Stevens Fishery Conservation and Management Act, House Subcommittee on Fisheries Conservation, Wildlife, and Oceans, Committee on Resources, March 9, 2000.

[32] Proponents of development often provide economic analysis in support of a project. The natural resource trustee is under no legal obligation to counter such analysis with an ecosystem benefit impact analysis of its own.

relatively small.[33] In other words, NMFS must conduct a large number of reviews on a limited budget, has no explicit mandate for economic analysis, and lacks legal authority under the EFH consultation requirement to prohibit actions or seek compensation for actions it deems ecologically undesirable.

These administrative realities, combined with the site-specific and spatially broad nature of the necessary evaluations, create a substantial challenge for the agency. The need to conduct numerous methodologically sound evaluations on a limited budget suggests that NMFS should be strategic about how it uses economic valuation arguments in the EFH consultation process.

4.2 A Methodological Critique of Nonmarket Valuation

Many economists and policy scientists question the value of valuation. For a variety of methodological reasons, they are skeptical of the possibility and desirability of ecosystem benefit assessment in many decisionmaking contexts. Some argue that not all aspects of a decision can or should be expressed in monetary terms, and that, by doing so, the contribution of economic argument to environmental decisionmaking is limited rather than enhanced. The basis for this critique is that the public decisionmaking process is a forum where appropriate values are debated and discovered. In such a setting, otherwise ill-formed preferences for different levels of environmental services are sharpened through public debate and reflected in the decisions.

Critics of nonmarket evaluation argue that preferences for the environment (or for any service) are not stable, static, or well formed—assumptions that are needed to measure society's willingness to pay for environmental goods and services. They observe that in market exchange, people engage in a preference discovery and adjustment process and argue that public choicemaking can be made to mimic this preference discovery process. Some schools within economics adopt this perspective, even though social psychologists have long accepted the proposition that preferences are not recalled as a basis for choice, but instead are constructed as people make choices. Meanwhile, philosophers have argued that revealed choices cannot be

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[33] Through the year 2000, NOAA Fisheries' EFH budgets hovered between $750,000 and $2 million for the entire EFH program, including basic habitat research. At most, roughly 40% of the total was devoted to the consultation process, including ecological research and assessment. At current levels, the amount available to economic assessment is limited. See testimony of Penelope Dalton, note 31.

interpreted as preferences except under circumstances and with assumptions that do not apply for environmental services. [34]

Explorations of the role of nonmarket evaluation for ecological benefit assessment are currently under way at the National Academy of Sciences[35] and at the Environmental Protection Agency (EPA).[36] This report does not take a position on this debate. But we believe that recognition of the debate over the value of valuation is essential, especially as NMFS sets its priorities for its future economics research and training programs.

4.3 Experience with Benefit Assessment in Environmental Decisionmaking

In this section, we review a variety of contexts in which NMFS has a significant decisionmaking or consultation responsibility. Overall, there are many occasions when it appears that ecological benefit assessment could inform decisionmaking, but assessment in the form of nonmarket (monetary) benefit estimation remains limited. The reason for the absence of such valuation estimates in decisionmaking varies by program. The agency may be prohibited by law from employing such estimates, agency leaders may question valuation for the philosophical reasons summarized above, or valuation may not be used for other reasons. However, it is unlikely that the reason is a lack of research on these techniques. For example, despite years of development and experimentation with practical benefit assessment tools, agencies —including the National Oceanic and Atmospheric Administration (NOAA) itself—have yet to deploy such tools in a widespread manner.

Our review consists of the existing and potential role of economic arguments in the two areas where NMFS and NOAA exercise maximum decisionmaking authority: critical habitat designation, and natural resource damage assessment.

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[34] See Leonard Shabman and Kurt Stephenson, Environmental Valuation and Its Economic Critics, *Journal of Water Resources Planning and Management,* September 2000, for a summary of the literature and arguments.

[35] Currently, the National Academy of Sciences is exploring methods for assessing services and the value of aquatic and related terrestrial ecosystems in a project titled “Assessing and Valuing of Aquatic Ecosystem Services.”

[36] EPA, for example, is currently convening a science advisory panel as part of “a comprehensive effort that will improve the methods used to value the benefits of protecting ecological systems and services to facilitate Agency decisions concerning the protection and restoration of ecosystems.” Federal Register 68(45): 11082–84, March 7, 2003.

4.3.1 Critical Habitat Designation

NOAA Fisheries and the Fish and Wildlife Service (FWS) list threatened and endangered species without reference to the economic impact of such a listing. However, they are also required to define critical habitat necessary for the support of such species. It is in the critical habitat designation process that economic assessment arises. Section 4(b)(2) of ESA requires NOAA and FWS to "take into consideration the economic impact, and any other relevant impact, of specifying any particular area as critical habitat." If the costs of designation exceed the benefits in a particular area, that area may be excluded from the critical habitat designation.[37] If the economic impact of restrictions (or project modifications) is deemed significant, ESA clearly states that critical habitat can be declassified. For example, ESA can bar changes to habitat when federal projects (irrigation, dam construction), federal money, or other federal actions are involved. Alternatively, the act can require project modifications designed to protect the habitat.

A recent court case brought by the New Mexico Cattle Growers Association has prompted FWS to devote more attention to the economic impact of critical habitat designations.[38] In the New Mexico case, the court ruled in favor of the ranchers, who argued that their livelihoods could be threatened if they were forced to fence cattle out of certain areas, build new stock ponds, or remove livestock from federal grazing allotments. Proponents of traditional land use practices hailed the New Mexico decision.[39] But ESA also makes clear that the benefits of a habitat designation must be taken into consideration.

[37] Exclusion is prohibited, however, if it will result in the extinction of the concerned species.

[38] See *New Mexico Cattle Growers Association v. U.S.F.W.S.*, 248 F.3d 1277 (10th Cir. 2001). The court took issue with Fish and Wildlife Service assumptions regarding the baseline for the economic impact analysis. FWS assumed that the postlisting world was the baseline, and then argued that the critical habitat designation had little incremental impact beyond the listing. Since economic analysis is not required for the listing itself, the result was, in the court's view, an excessively limited economic impact analysis: "we conclude Congress intended that the FWS conduct a full analysis of all of the economic impacts of a critical habitat designation, regardless of whether those impacts are attributable co-extensively to other causes."

[39] See "Court Ties Economic Impact to Endangered Species Areas," CNSNews.com, May 19, 2001, at http://www.newsmax.com/archives/articles/2001/5/18/180201.shtml (accessed September 10, 2003): "The decision was hailed as a significant victory for people throughout the country who say their livelihoods and culture are being ruined and their communities destroyed through deliberate misuse of the ESA by environmental extremists."

4.3.2 Natural Resource Damage Assessment

NOAA also has a role in calculating natural resource damage (NRD). Liability for events that damage resources was established under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA),[40] the Oil Pollution Act (OPA),[41] and the National Marine Sanctuaries Act.[42] The statutes create a compensable monetary liability for damage, which in turn requires calculation of the monetary value of the damage. NRDs can take a variety of forms but typically relate to changes in the condition of a habitat or species population and in the underlying ecological processes on which they rely.[43] For this reason, the NRD experience is relevant to the challenge NMFS faces under Magnuson-Stevens.[44]

OPA and CERCLA directed NOAA and the Department of the Interior (DOI), respectively, to develop rules governing natural resource damage compensation.[45] In the 20 years following this instruction, the agencies have been developing and applying damage assessment tools that integrate biophysical and economic analysis.[46] For this reason, NRD cases and assessments are a laboratory for examining the use of the benefit assessment for service-based habitat evaluation in the U.S. regulatory environment.[47]

[40] 42 U.S.C. §§ 9601–9675 (1994 & Supp. IV 1998). Section 107 of the act establishes NRD liability and authorizes federal trustees to recover damages for assessing and correcting natural resource injuries. 42 U.S.C. § 9607(f)(1) (1994).

[41] 33 U.S.C. §§ 2701-2761 (1994 & Supp. IV 1998). Section 1002 of the act establishes liability for "injury to, destruction of, loss of, or loss of use of natural resources." 33 U.S.C. § 2702(b)(2)(A) (1994).

[42] 16 U.S.C. §§ 1431-1445(a) (1994 & Supp. IV 1998). Section 1432(6) defines damages, and Section 1443 establishes liability and authorizes civil actions to pursue cost recovery.

[43] See 15 C.F.R § 990.51(c) (2000): "Potential categories of injury include, but are not limited to, adverse changes in: survival, growth, and reproduction; health, physiology and biological condition; behavior; community composition; ecological processes and functions; physical and chemical habitat quality or structure; and public services."

[44] An important difference is that NRD assessments occur *ex post*, after ecological damage has already occurred. NMFS consultations occur before damages arise.

[45] Compensable value includes "all of the public economic values associated with an injured resource, including use values and nonuse values such as option, existence, and bequest values." 56 Fed.Reg. 19760, April 29, 1991.

[46] The rules are codified at 15 C.F.R. 990 (the NOAA rules for OPA damages) and 43 C.F.R. 11 (the DOI rules for CERCLA damages).

[47] This claim is based on NRD liability's unique statutory basis and on the relatively large government expenditures devoted to the assessment of NRD damages. Compare the resources spent to evaluate the *Exxon Valdez* oil spill with the resources available for individual wetland permit decisions.

The science behind both the ecology and the economics of this kind of assessment has been the subject of much debate. Most NRD cases demand significant amounts of data regarding both biophysical conditions and demand for the services damaged. The cases tend to be complex and have not always been successful.[48] The fact that major NRD assessment is triggered only under a narrow set of conditions, most typically large pollution events to which OPA and CERCLA apply, means that lessons drawn from the NRD experience may not necessarily apply to other decisionmaking contexts, such as EFH consultation.[49]

Recently, NRD assessments have shifted from an emphasis on monetary benefit estimation to nonmonetary techniques—in particular, habitat equivalency analysis. Given the relatively significant resources available to NOAA under NRD actions, it is unlikely that this shift is due entirely to the research costs of nonmarket valuation.[50]

We now turn to the existing and potential role of economic arguments in areas where NMFS plays a consultative role with the Corps of Engineers and EPA: wetland permitting and public works approval decisions.

4.3.4 Section 404 Permitting

In the late 1960s, a series of court interpretations of the 1899 Rivers and Harbors Act required that the U.S. Army Corps of Engineers expand its review of applications to build structures in navigable waters to include not only possible obstructions to navigation, but also the effects on wildlife habitat. This judicial action was intended to bring that Corps permitting program into compliance with the requirements of the Fish and Wildlife Coordination Act of 1958 (FWCA). However, FWCA required only that the habitat effects be considered in decisionmaking; there was no mandate to protect habitat. At this point, the Corps slowly began to develop a

48 For an excellent description of the NRD process in court, see David Chapman and W. Michael Hanemann, Environmental Damages in Court: The American Trader Case, in *The Law and Economics of the Environment*, edited by Anthony Heyes, Elgar, 2001. Also see Dale B. Thompson, Valuing the Environment: Courts' Struggles with Natural Resources Damages, *Environmental law* 32(1), 2002.

49 The NRD provisions of the OPA, 33 U.S.C. §§ 2701–2761 (1994 & Supp. IV 1998), and the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), 42 U.S.C. §§ 9601–9675 (1994 & Supp. IV 1998), explicitly mandate ecosystem valuation as a response to ecosystem service losses.

50 See Carol A. Jones and Katherine A. Pease, Restoration-Based Measures in Natural Resoiurces Liability Statutes, *Comtemporary Economic Policy*, XV, October 1997; also see Monica P Medina, Just Do It, *The Environmental Forum,* July/August, 2001.

program of regulation over the filling of wetlands. Indeed, questions about the permit program arose continually. Did the jurisdiction of the Corps permit program for the navigable waters of the United States include wetlands adjacent to the water bodies? Were the effects on habitat to be only those at the immediate site of the filling, or did the effects include possible indirect (off-site) consequences? The National Environmental Policy Act of 1969 then expanded the required review of permits to environmental concerns beyond wildlife habitat and deemed a permit decision to be a “significant” federal action. However, NEPA, like FWCA, required only that consideration be given to environmental impacts and carried no substantive statement of environmental requirements.

In Section 404 of the 1972 Amendments to the Federal Water Pollution Control Act (FWPCA 72), Congress granted the Corps, with EPA oversight, the responsibility for issuing permits for placement of fill material in waters of the United States. The goal of the act as a whole was to “restore the chemical, physical and biological integrity of the Nation’s waters.” The Corps was expected to review the merits of private and public sector fill proposals and initially evidenced some intent of applying an evaluation model that balanced the benefits and costs of granting a permit. In the regulatory program, this was called the public interest review process (PIRP). However, because the time allowed for a permit decision in the vast majority of cases was only a few weeks, there was no opportunity for the permit review to be based on detailed technical evaluations. Instead, possible categories of effects were articulated, and the permit decision was based on that “laundry list.” In the early stages of the program, the Corps made a judgment on the extent of development value that might be realized by the wetlands permit applicant; if that was deemed to be “large” relative to the environmental harm, the permit was granted. Where practical and cost-effective, some permit conditions required that the applicant restore or create wetlands as compensation for those lost to the fill.

However, the Corps effort at such explicit balancing of benefits and costs was soon replaced by the 404(b)(1) guidelines, under which only the activities deemed “water dependent” would be considered eligible for a permit. Whenever it was “technically practical” to avoid the wetlands (i.e., no water dependency), the permit was to be denied. If avoidance was not possible because of the water dependency of the activity, the permit applicant was expected to minimize the impact on the wetlands, and then compensate for those effects that were not avoided after impacts were minimized.[51] The logic of this sequencing can be traced to the goals and logical structure of

[51] 33 U.S.C. § 1344 (1994).

CWA itself: the act calls not for balancing values but for eliminating the discharge of all pollutants into the waters of the United States.

The 404(b)(1) sequencing logic focuses attention on how much and what type of mitigating actions will replace the services of any lost habitat. The Corps must, as must NMFS under EFH consultations, make this decision for many relatively small permitted fills every year. In this setting, the value of the lost services is not the issue. Instead, the Corps relies on biophysical analysis for the assessment of wetland impacts that must be mitigated. Program guidance documents for the use of wetland mitigation banks emphasize functional characterization, rather than service valuation, as the relevant metric for wetland comparisons. According to federal guidelines, "[t]he objective of a mitigation bank is to provide for the *replacement of the chemical, physical, and biological functions* of wetlands and other aquatic resources which are lost as a result of authorized impacts" (emphasis added).[52] Such a focus is not surprising, given CWA's goal of restoring the chemical, physical, and biological integrity of the nation's waters. Surveys of wetland mitigation banking practice confirm the reliance of bank program administrators on relatively vague, function-based compensation ratios.[53] A 1992 study found that although a variety of evaluation methods were used, all were based on functional assessment, habitat evaluation, or simple acreage-based conversion rules.[54] Case studies of 46 banks, published that same year, showed 20 banks using functional assessment and 26 using rough acreage-based ratios.[55] Current practices are much the same.[56]

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[52] Federal Guidance for the Establishment, Use and Operation of Mitigation Banks, 60 Federal Register 58,605, 58,607 (November 28, 1995). Functional assessment itself is not a strict requirement.

[53] See John Brady, Mitigation of Damage to Wetlands in Regulatory Programs and Water Resource Projects, *Mercer Law Rev*iew 41: 893, 941–46, 1990.

[54] Environmental Law Institute and Institute for Water Resources, National Wetland Mitigation Banking Study, Wetland Mitigation Banking, U.S. Army Corps of Engineers, Institute for Water Resources, IWR Rep. 94-WMB-2, 1994.

[55] Robert Brumbaugh and Richard Reppert, National Wetland Mitigation Banking Study: First Phase Report, U.S. Army Corps of Engineers, Institute for Water Resources, IWR Rep. 94-WMB-4, 1994.

[56] J.B. Ruhl and R. Juge Gregg, Integrating Ecosystem Services into Environmental Law: A Case Study of Wetlands Mitigation Banking, *Stanford Environmental Law Journal* 20: 365, 2001. See also Leonard Shabman, Kurt Stephenson, and Paul Scodari, Wetlands Credit Sales as a Strategy for Achieving No Net Loss: The Limitations of Regulatory Conditions, *Wetlands* 18: 3, 1998.

4.3.5 Army Corps Investment Decisionmaking

Proposed investments in water infrastructure by the Corps of Engineers are evaluated according to the principles and guidelines (P&G) for water and related land resources planning. The federal objective in the P&G is to maximize national economic development (NED) while complying with all environmental laws and regulations. The NED evaluation is a conventional benefit-cost analysis. For example, in planning navigation capacity for bulk commodity movements, the P&G requires an analysis documenting an increased demand for bulk commodity transportation without the project in place, to show that additional navigation capacity is justified. In considering the environmental effects of such projects, the P&G is structured to require a kind of sequencing text so that losses are avoided, minimized, and then compensated. Although the actual practice of mitigation has been subject to critical review,[57] the point of interest here is that environmental services lost or gained from any investment are not valued in monetary terms using nonmarket valuation methods.

Recently, the Corps introduced a new mission to its investment program, termed environmental restoration, in which the evaluation and justification procedures for projects to restore rivers and aquatic habitat do not employ nonmarket valuation methods. In principle, the Corps has adopted an incremental analysis process that evaluates how costs increase with increasing levels of restoration outcomes (e.g., additional acres of restored wetlands). Incremental analysis does not provide a justification for selecting any specific restoration level. Rather, the justification rests on demonstrating the "significance to the nation" of the level of restoration recommended.[58] Plausibly, monetary measures of people's preferences for the restoration services might be calculated and reported to document significance, although such analysis is not required. Instead, emphasis tends to fall on physical habitat quality or legal and policy recognition of significance of the waters or areas where the restoration is realized (for example, the area may be critical habitat according to the Endangered Species Act).

57 U.S. Army Corps of Engineers, Scientific Panel's Report on Fish and Wildlife Mitigation Guidance, GAO 02-574, May 2002.

58 See U.S. Army Corps of Engineers, EC 1105-2-404, Planning Civil Works Projects under the Environmental Operating Principles, May 1, 2003.

4.3.6 Agriculture Reserve Programs

A variety of programs administered by the U.S. Department of Agriculture (USDA) involve planning or government expenditures for conservation.[59] There is also a growing movement to measure and prioritize subsidy payments based on the public environmental benefits of private lands.[60] For instance, all else being equal, subsidies could be directed to farms producing greater ecological benefits in the form of habitat to support rare and threatened or recreational species.

Ecosystem ranking already occurs under the Wetlands Reserve and Conservation Reserve programs.[61] Although the ranking process is unique to each state, most scoring criteria relate to such factors as habitat type, hydrology, species support, operations and maintenance costs, and the likelihood of limiting factors, such as invasive species.[62] These factors are largely biophysical in nature, with very few ecosystem service-related data used.

Although some value-based indicators are evident, the Conservation Reserve Program (CRP) places a similar emphasis on functional—as opposed to service—evaluation. The program gives sites an "environmental benefits index" ranking. The biophysical components of this ranking include measures of soil erodibility and leachability. Service indicators include the number of well-water users in proximity to the land. This approach does not simply index a land's functional characteristics; it also indexes the social value of the land's water purification function. The value of including additional service value indicators in the USDA's

59 For an overview of these programs, see National Governors Association, Private Lands, Public Benefits: Principles for Advancing Working Lands Conservation, 2001, Appendix A, 46–49.

60 National Governors Association, note 59, at 6: "Government-supported working lands conservation programs should demonstrate that they produce valuable and measurable 'environmental goods' or 'conservation commodities.'"

61 See generally Heimlich et al., Wetlands and Agriculture: Private Interests and Public Benefits, USDA, Agricultural Economic Report No. 765, 1998. The Fish and Wildlife Service also targets wetland creation and restoration through private lands partnerships and wildlife extension agreements. For overviews of these programs, see Partners for Fish & Wildlife, U.S. Fish & Wildlife Service, Partners Program, at http://midwest.fws.gov/alpena/prvprogr.htm (accessed March 29, 2001), and Partners for Fish & Wildlife, U.S. Fish & Wildlife Service, U.S. Fish & Wildlife Service Programs, at http://www.r6.fws.gov/pfw/r6pfw8c.htm#WEA (accessed March 29, 2001).

62 See, e.g., Natural Resources Conservation Service, USDA, Wetland Reserve Program (WRP) Ranking Forms, Oregon Bulletin No. OR300-2001-1, Attachment 3, 2000, at http://www.or.nrcs.usda.gov/admin/ orbulletin.html (last updated April 4, 2001).

targeting of CRP lands has been explored by the agency, though a service-intensive index is not currently in use.[63]

4.3.7 Forestry Management

The Forest Service's planning guidelines point toward a future role for ecosystem service analysis in the management of National Forest System lands and resources.[64] New management rules, finalized in 2000 but subsequently withdrawn, require "the development of information on the range and estimated long-term value of market and nonmarket goods, uses, services, and amenities that can be provided by National Forest System lands consistent with the requirements of ecological sustainability."[65] This aspiration is a fairly explicit statement of the need for ecosystem service benefit estimation in Forest Service planning.

In addition, the Federal Land Management Policy Act (FLMPA) authorizes the Forest Service and the Bureau of Land Management (BLM) to exchange federal land for nonfederal land, subject to certain conditions.[66] These conditions are that both the market value and the public benefits of the lands being acquired be comparable to or exceed the market value and public benefits of the lands being lost. Also, an environmental analysis must be completed for certain exchanges, as required by the National Environmental Policy Act.[67] The Forest Service and BLM have recently been criticized for their poor evaluation of both market value and public benefits. A Government Accounting Office review team found a "lack of documentation to support certain public interest determinations" and said that the agencies failed to show the

[63] See Peter Feather et al., Economic Valuation of Environmental Benefits and the Targeting of Conservation Programs: The Case of the CRP, USDA, Agricultural Economic Report No. 778, 1999 (discussing nonmarket benefits and valuation, and describing three valuation models).

[64] This is not a new idea, at least in academic circles. See Michael Boews and John Krutilla, Multiple-Use Management: The Economics of Public Forestlands, Resources for the Future, 1989.

[65] 65 Federal Register 67514, 67551, November 9, 2000: "The Department believes this language in the final rule requires the inclusion of commodity and non-commodity resource benefits in economic analyses, with values assigned to those benefits." The rule is currently suspended. Federal Register 27551, May 17, 2001.

[66] P.L. 94-579, October 21, 1976. The public interest determination requires the agency to "...give full consideration to better Federal land management and the needs of the State and local people, including needs for lands for the economy, community expansion, recreation areas, food, fiber, minerals, and fish and wildlife…" 43 U.S.C. 1716(a).

[67] P.L. 91-190, January 1, 1970.

preservation of public benefits.[68] Accordingly, analysis of public benefits is a current priority for both agencies.

4.3.8 FERC Licensing

The Federal Power Act granted the Federal Energy Regulatory Commission (FERC) sole authority to license and relicense nonfederal water power projects on navigable waters. FERC was established as an independent commission to ensure that dam licensing decisions were based on sound technical studies and to insulate FERC commissioners from political pressure.[69] When making a dam relicensing decision, FERC has had to decide which fish passage strategies (size and extent of fish screens and fish ladders) and minimum downstream flow requirements should be placed in a dam operator's license. FERC staff decides the required conditions based on "knee-of-the-curve" cost analysis. FERC analysts consider the cost of mitigation alternatives; costs typically include opportunity costs of forgone power production (e.g., from increasing minimum stream flow requirements) and the construction costs (e.g., fish ladders or screens). The FERC staff then estimates the environmental response to these mitigation alternatives in a natural unit, such as changes in the number of fish or habitat coverage. With comments and input from the licensee and other stakeholders, the staff (and if necessary, FERC commissioners) then judges whether the mitigation measures are "worth" the additional cost. Rarely is an attempt made to monetize the fish habitat or water quality benefits of these mitigation measures.[70] In the rare cases in which monetization of benefits occurs, it is related to changes in recreational activity—decisions closely related to observable market activity.

This closed decisionmaking process has prompted criticism in recent years and has become the object of legislative reform, legal actions, and administrative appeals.[71] License applicants as well as resource agencies and environmental groups have described FERC's

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[68] General Accounting Office, Land Exchanges Need to Reflect Appropriate Value and Serve the Public Interest, GAO/RCED-00-73, June 2000, at 28; at 4: "The agencies did not follow their requirements that help show that the public benefits of acquiring the nonfederal land in an exchange matched or exceeded the public benefits of retaining the federal land, raising doubts about whether these exchanges served the public interest."

[69] D. B. Spence, Managing Delegation Ex Ante: Using Law to Steer Administrative Agencies, *Journal of Legal Studies* 28(2): 413–59, June 1999.

[70] Moore et al., note 16.

[71] Moore et al., note 16.

decisionmaking processes as costly, conflict ridden, and (for some) inadequately attuned to the value of environmental services. In 1997, FERC approved the use of an alternative relicensing process, and in July 2003, it added the integrated licensing process.[72] In general, both processes encourage the dam operator to enter into a consensus-building process with interested stakeholders to identify the relevant studies, mitigation alternatives, and operating conditions, and ultimately to develop a mutually agreed-to license application. These processes, like the traditional process, do not require benefit estimation or nonmarket valuation.

5. Bioeconomic Modeling

In Sections 5 and 6, we provide concrete examples of assessment techniques. The examples are meant to illustrate ways in which benefit assessment requires a link between biophysical and socioeconomic data and analysis. Both examples illustrate the application of economic principles and reasoning to habitat assessment. In other words, both are forms of economic argument that could be applied by NMFS. However, the examples are also chosen for their differences.

In this section, we describe the use of integrated biophysical and socioeconomic models—what we refer to as bioeconomic models. These models are formal, mathematical, and correspondingly precise in their depiction of biophysical and economic phenomena. Models such as these form the foundation of assessments that seek monetary estimation of ecological benefits. We discuss model construction, use models to depict spatial phenomena, and illustrate the assessment of threshold effects. Ideally, NMFS might develop such modeling capability and the data needed to derive empirical conclusions from such models. In particular, the agency's mission argues for an emphasis on bioeconomic modeling and spatial modeling capability.

In Section 6, we present an alternative form of assessment—ecosystem benefit indicators. Unlike bioeconomic models, benefit indicators lack a formal depiction of relationships and do not purport to lead to monetary benefit estimates. This is both a positive and a negative. Benefit indicators avoid the pitfall of formal mathematical models, whose complexity can often alienate parties involved in a joint decision. The disadvantage of this decision process is that it is not comparable or repeatable across applications—two critical aspects of scientific inquiry. Nevertheless, as an input to decisionmaking processes typically void of economic practitioners

[72] 18 C.F.R., Parts 2,4,5,16,375,381, July 23, 2003.

or assessment, a rigorous application of any set of economic tools can be a marked improvement over current assessment practices that eschew economic valuation arguments.

Below we present an overview of bioeconomic modeling, what we mean when we discuss models, some caveats of the approach, and three stylized examples illustrating the approach. For expositional purposes, the examples describe the impact of the habitat alteration only on commercial and recreational fishing. Modeling of other services is clearly possible, at the expense of simplicity.

5.1 Overview of Bioeconomic Modeling

From an economist's perspective, habitat is a natural asset whose restoration and mitigation represent investment, and alteration and destruction represent divestment. Similar to man-made capital assets, habitats are durable, implying that levels and changes today affect the level and rate of return in the future. Decisions today therefore need to consider the current costs against the discounted future stream of net benefits. Further complicating the investment decision are the significant adjustment costs associated with mitigation and restoration—habitat regeneration takes time. For example, many restoration projects replicate natural system function only after a period of several years.[73] Even if full replication of a more natural system is assured, there is an adjustment cost associated with the interim loss in the ecological asset's function.[74]

Taking the analogy further, if the habitat or ecological function is an asset, then the rate of return on the asset is determined by the ecosystem services provided by the habitat. As valuable "commodities," ecosystem services exhibit the basic properties of any economic good. As discussed earlier, however, the economic assessment of ecological services is fundamentally more difficult and controversial.

To better understand the difference between ecosystem services (goods produced) and ecosystem functions (production technology), we define what economists refer to as a production function. Production functions describe the manner in which an output is related to the quantity

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[73] In the case of mitigation, the goal is to replace functions equivalent to those lost. If the lost habitat was already significantly altered, then restoration to a natural system may take longer than the time required to secure mitigation.

[74] In addition, uncertainties and changes in the temporal flow of ecosystem benefits are also associated with the nonbiophysical determinants of value. For example, demographic and technological changes can alter the flow of ecosystem service benefits over time. Accordingly, the value of a particular ecosystem may decline because of urban encroachment, manmade water diversions, or the discovery of substitutes for services generated by the ecosystem.

and nature of inputs used to create it.[75] For example, a production function describes the way in which a company can deploy labor and capital investments to manufacture a final product. Ecological analysis, of course, is concerned with the biological, chemical, and hydrological relationships that determine biological production.[76] An ecosystem's structure, such as size, vegetation, and boundaries, and its functional aspects, such as ability to absorb floodwater or remove contaminants from surface water, are biophysical contributors—as inputs—to the services the habitat generates. Although economic and biological systems are clearly different in important respects, both economics and ecology seek to understand the activity or productivity of systems by understanding the systems' basic components and the functional relationships among those components.

In addition to being a durable asset, the biophysical nature of ecosystems in general and habitats in particular is spatially interdependent. For example, ecological science emphasizes the importance of habitat connectivity and contiguity to the productivity and quality of that habitat (e.g., measured through species diversity or richness).[77] Terms like connectivity and contiguity are inherently spatial and refer to the overall pattern of land uses, surface waters, and topographic characteristics in a given region. Species interdependence and the need for migratory pathways are additional sources of spatial phenomena in ecology.[78] Because the ecological production function depends on the characteristics of the overall landscape,

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75 See *The New Palgrave: A Dictionary of Economics,* 1998, at 995: "The traditional starting point of production theory is a set of physical technological possibilities, often represented by a production or transformation function. The development of the theory ... leads to input demands (and output supplies) constructed from an explicit consideration of the underlying technology."

76 There is also a long history of integrated economic and biological production function analysis in agricultural and natural resource economics. Among other things, agricultural studies show how substitution of one farm input for another (e.g., land for fertilizer, tractors for man-hours) affects production levels, or how landscape characteristics affect yields. For a general overview, see *Introduction to Agricultural Economics,* edited by John Penson et al., Prentice Hall, 1999.

77 R.F. Noss, Indicators for Monitoring Biodiversity: A Hierarchical Approach, *Conservation Biology* 4(4): 355–64, 1990; R.H. Gardner, R.V. O'Neill, and M.G. Turner, Ecological Implications of Landscape Fragmentation, in *Humans as Components of Ecosystems: Subtle Human Effects and the Ecology of Populated Areas,* edited by S.T.A. Pickett and M.J. McDonnell, Springer-Verlag, 1993, 208–26; E.J. Gustafson, Quantifying Landscape Spatial Pattern: What Is the State of the Art? *Ecosystems* 1: 143–56, 1998; C.L. Richards, L.B. Johnson, and G.E. Host, Landscape-Scale Influences on Stream Habitats and Biota, *Canadian Journal of Fisheries and Aquatic Science* 53 (Suppl. 1): 295–311, 1996.

78 C.H. Flather and J.R. Sauer, Using Landscape Ecology to Test Hypotheses about Large-Scale Abundance Patterns in Migratory Birds, *Ecology* 77(1): 28–35, 1996.

economists need to understand an area's spatial characteristics to assess benefits.[79] In the case of EFH analysis, the challenge involves assessment of terrestrial, freshwater, estuarine, and marine "landscapes."

Understanding the relationship of inputs to outputs is a partial, but incomplete, step toward valuation. As has been noted, people's preferences are the basis for valuation. Individuals' and society's collective preferences determine the value of an additional unit of a habitat's environmental function. Just as surrounding economic conditions affect preferences for, and the value of, a port approach channel, so too will a habitat's setting—local land use configurations, related human activities, and demography—affect its environmental service value. All seagrass beds are not of equal value. The preferences that underlie demand for ecosystem services depend on a variety of factors.

First, valuation requires an understanding of input and output scarcity and the ability to make input and output substitutions. All else equal, greater scarcity or rarity of a habitat type and a relative lack of both natural and man-made substitutes increase the value of an ecosystem service. For example, the value of the only seagrass acre is greater than the value of the millionth. The cost and practicality of replacing the service of a particular habitat with some technological substitute (e.g., a fish hatchery) also affects the habitat's value. Because the value of ecosystem services is inversely related to resource scarcity, valuation requires a "marginal" analysis of ecosystem benefits, which examines the way in which a service's benefits vary with the aggregate level of the service available.[80]

Second, valuation requires an assessment of complementary goods and services. Many ecosystem services become valuable, or increase in value, when enjoyed in combination with other goods or services. For example, a forest generates recreational and aesthetic services only when there is access to it. Consequently, roads, trails, and adjoining navigable waters are complementary goods to the forest's recreational and aesthetic services.

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[79] See Nancy Bockstael and E. Irwin, Economics and the Land Use-Environment Link, in *The International Yearbook of Environmental and Resource Economics 2000/2001,* edited by T. Tietenberg and H. Folmer, Kluwer, 2000, 1–54.

[80] See R. David Simpson et al., Valuing Biodiversity for Use in Pharmaceutical Research, *Journal of Political Economy* 104: 163–64, 1996, describing the importance of marginal analysis in the valuation of preserved species and lands as a source of pharmaceutical discovery.

5.2 Principles of Bioeconomic Modeling

In this section, we discuss bioeconomic models that can be developed to measure the magnitude of benefits arising from habitat. First, we provide a short discussion of what constitutes a model and some of the caveats associated with bioeconomic modeling. The discussion of models is illustrated using three stylized examples that relate habitat values to changes in fish abundance when fish are commercially harvested. Of course, there are additional values from recreational fishing and nonuse values that need to be taken into account.

The examples used here provide critical insights into habitat value measurement. They also illustrate the need to understand ecological and economic context, to explicitly model spatial processes, and to consider the presence of threshold effects. To preview our findings, we will argue that a continuing regional or large-scale effort is essential to fully manage habitats, where the analysis considers intertemporal and spatial aspects simultaneously and is continually refined as new scientific information becomes available.

5.3 Modeling and Some Caveats

The EFH consultation mandate is to have federal agencies consult with NMFS whenever an action may alter a designated habitat and thereby affect fish populations. The concern is not just for the immediate effects in the designated habitat areas but also encompasses the functions of nursery and spawning areas, food chain effects, and cover. All of these factors need to be considered in establishing the effects of an alteration on commercial fishing and sport fish populations.

The starting point to assess these changes is the underlying biophysical system (which ecological science evaluates) because it is the basis of the underlying production function that generates ecosystem services. Historically, characterizations of species have long been considered a relatively good indicator of other ecosystem features. For example, most early attempts at ecological functional assessment focused on animal populations and were based on field surveys (or, more recently, remote sensing) to detect the presence or absence of certain species.[81] Population surveys are now often supplemented or replaced by methods focused on

[81] See Walter E. Westman, Ecology, Impact Assessment, and Environmental Planning, 1985; see also Jan Bakkes et al., An Overview of Environmental Indicators: State of the Art and Perspectives National Institute of Public Health and Environmental Protection, Environment Assessment Technical Reports UNEP/EATR, 1994.

more permanent biophysical and landscape structural characteristics—characteristics that signal the suitability of sites as habitat.[82] And increasingly, assessment focuses on habitat function analysis, or measures of the ability of a habitat structure to provide support for selected species.

The recent and growing trend to measure the interrelatedness between habitats and species diversity and abundance is one of the reasons that bioeconomic modeling is well suited to help quantify or predict habitat benefits. Our focus on bioeconomic modeling also follows from the recommendations of King et al.,[83] that to improve the use of economics in habitat protection, NMFS should develop and employ bioeconomic models, especially with respect to quantifying cumulative impacts. Bioeconomic analysis, which has been the basis for fishery management for more than 50 years,[84] is especially good at analyzing dynamic or intertemporal problems.

More recently, spatial bioeconomic models have been developed to understand the role of biological and economic connectivity of subpopulations.[85] These spatial and intertemporal bioeconomic models are used to investigate patterns of exploitation and spatial management approaches, such as marine reserves and spatially explicit limited-entry systems or individual fishing quotas. With some minor modifications, these models can be adapted to address issues associated with habitat management. As we will discuss below, however, the models require substantial data and knowledge of ecological processes that are not currently well understood.

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[82] Under one widely used approach, a habitat suitability index (HSI) and the total area of habitat are used to quantify habitat. U.S. Department of the Interior, Fish and Wildlife Service, Habitat Evaluation Procedure (HEP), 1980. The use of HSIs still requires the analysts to choose a limited number of species for attention. In effect, the species chosen for the analysis at the outset will determine the resulting HSI score.

[83] Dennis King, Douglas Lipton, Ivar Strand, and Katherine Wellman, A Role for Economics in NMFS Habitat Conservation Activities, Report to the NMFS, Office of Habitat Conservation, Habitat Protection Division, December 2002.

[84] Some examples of bioeconomic analysis are A.D. Scott, The Fishery: The Objectives of Sole Ownership, *Journal of Political Economy* 63: 116–24, 1955; M.B. Shaefer, Some Considerations of Population Dynamics and Economics Relation to the Management of Marine Fisheries, *Journal of the Fisheries Resources Board of Canada* 14: 669–81, 1957; Colin W. Clark, *Mathematical Bioeconomics. The Optimal Management of Renewable Resources*, 2 ed., John Wiley, 1990; V.L. Smith, Economics of Production from Natural Resources, *American Economic Review* 58: 409–31, 1968;V.L. Smith, On Models of Commercial Fishing, *Journal of Political Economy* 77: 181–98, 1969; and James E. Wilen, Bioeconomics of Renewable Resource Use, in *Handbook of Natural Resource and Energy Economics*, edited by A.V. Kneese and J.L. Sweeney, Elsevier, 1985.

[85] See Sanchirico and Wilen, Bioeconomics of Spatial Exploitation in a Patchy Environment, *Journal of Environmental Economics and Management* 37,129-150, 1999.

Before discussing the design of bioeconomic models, it is important to clarify the role that models can play in policy analysis and to elaborate on the definition of a model. First, models in the broadest sense are necessary to inform decisionmaking—we think individually and collectively through models. Models can run from simple, intuitive notions to highly complex mathematical representations. Whatever the complexity of the model, a model is at its basic level an abstract representation of the phenomena of interest. For collective decisionmaking involving multiple stakeholders, it is necessary to develop a common model—or suite of models—accessible to (or at least accepted by) all stakeholders for the purpose of asking and answering "if-then" questions that will inform the actions of those authorized and responsible for decisionmaking.

Models are a combination of art and science, and the relative contribution of each depends on the question being addressed. More technical models can be used to make quantitative (measured) predictions of outcomes of interest, such as the change in the population of a particular year class of fish as a result of a habitat alteration (see Example 1). It should come as no surprise that as a model's complexity increases, so too can its predictive powers decrease. For example, it is more difficult to predict the impact of habitat alteration on a fish population than to predict the impact of food availability in that population's nursery because habitat changes involve an additional causal linkage in the habitat–food supply–population chain. In these settings, the art of modeling is to devise a framework that is not so complex that it reduces the ability to empirically apply it to a particular place or prediction of interest. As a general rule of thumb, however, model prediction uncertainty grows with the number of interrelationships represented.

Some of the limits on building a complex model are clear. Complexity comes at a cost in money and time. If research or human resources are limited, then it may be necessary to rely on simpler models. Less obvious are data limitations. A complete conceptualization of a system may be written down, but the data to provide evidence of formal relationships might not be currently available. Therefore, the magnitude (and maybe direction) of the causal influences may not be known. Although this can be addressed somewhat with Monte Carlo techniques, if not carefully done, the result could be overparameterization (relative to the data) of the model design.

5.4 Models for Habitat Valuation

In this section, we elaborate on the important features that need to be incorporated into a bioeconomic model to manage habitat investment decisions. In particular, we discuss both the temporal and spatial dimensions of ecological management. Finally, we offer three stylized examples of bioeconomic model to provide further intuition on how such models can provide information useful to NFMS in evaluating the benefits and costs of habitat management.

5.4.1 Time and Space: Dimensions of Analysis

Durability and adjustment costs imply that permit decisions today will affect decisions tomorrow. Therefore, NMFS consultation comments should not only reflect today's costs and benefits, but also the future impact of habitat conversion. For example, it is possible that the effects and consequences of incremental changes in habitat can alter significantly a coastal region's function (nursery area suitability, species mix and carrying capacity, nutrient sinks and cycling) as support for consumptive and nonconsumptive recreation and commercial harvest. Representing all of these effects is the difficult challenge that faces NMFS.

The practice of treating each decision as independent over time is equivalent to assuming a myopic decision rule. Under most conditions, myopic decision rules can lead to too much habitat alteration relative to decision rules that take into account the durability and adjustment cost characteristics of the asset.[86] Another way to think about myopic decision rules is that they implicitly assume that the level of habitat today does not affect the level tomorrow. A myopic rule would be reasonable if habitats (e.g., seagrass beds, coral reefs, wetlands) were abundant resources or able to regenerate rapidly, but many habitats are relatively scarce and take years to develop.

The social determinants of service benefits also depend upon the landscape context in which those services arise.[87] In fact, the consumption of services often occurs off-site. Habitat support for recreational and commercial species, sediment and nutrient trapping and recycling to support clear swimming water, flood damage reduction, and aesthetic enjoyment are all services typically enjoyed in a larger area surrounding the habitat in question. To ignore or minimize the importance of off-site factors misses much that is central to a complete valuation of benefits. How scarce is the service? What complementary assets (e.g., trails, docks) exist in the

[86] The intuition for this stems from the literature on optimal intertemporal fishery management. See Clark, note 84, for more discussion.

[87] See Geoffrey Heal, Gretchen Daily, Paul Ehrlich, James Salzman, Carol Boggs, Jessica Hellmann, Jennifer Hughes, Claire Kremen, and Taylor Ricketts, Protecting Natural Capital Through Ecosystem Service Districts, *Stanford Environmental Law Journal* 20, 2001; and Nancy Bockstael, Modeling Economics and Ecology: The Importance of a Spatial Perspective, *Amer. J. Agr. Econ.* 78: 1168, 1996.

surrounding landscape that enhance the value of a service? These questions relate to the overall landscape setting and are, accordingly, spatial in nature.

The EFH mandate extends to adverse effects arising from habitat degradation regardless of the location of the effect.[88] Moreover, water quality management and wetlands management decisions are increasingly made in a spatial context, such as a watershed plan. A pertinent current example of this approach is the Clean Water Act's Total Maximum Daily Load Program, which relates ambient water quality to pollutant sources across an entire watershed. Coastal zone management and special area management plans are other examples of a spatially oriented regulatory approach to planning and management for wetlands. Endangered Species Act (ESA) habitat conservation plans clearly involve landscape-related planning and analysis.

A regional-scale approach is an important component of any examination of ecological conditions and associated environmental services because it can provide information about larger patterns generated by the accumulation of decisions at a finer scale. This is akin to the notion of "cumulative effects." NMFS is clearly interested in the cumulative impacts of land use, construction, and water management decisions as they pertain to EFH. Accordingly, the impact of individual decisions on the pattern of resource use and habitat over a larger scale is fundamental to assessment. For example, from a biophysical standpoint, interactions across a marine-estuarine-freshwater system may take place over very broad spatial scales. Particularly in the case of anadromous species, habitat impacts may have to be evaluated over areas hundreds of square miles in size.

According to NMFS guidance, "any reasonable attempt to encourage the conservation of EFH must take into account actions that occur outside [that habitat], such as upstream and upslope activities that may have an adverse effect on EFH. Therefore, EFH consultation with NMFS is required by federal agencies undertaking, permitting, or funding activities that may adversely affect EFH, *regardless of its location*" (emphasis added).[89] From an economic standpoint, the services created by species and habitats can be equally wide ranging. Commercial and recreational service impacts, for example, can affect individuals and communities far from the proximate cause of habitat degradation.

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[88] See note 89.

[89] EFH Consultation Fact Sheet, available at www.nwr.noaa.gov/1habcon/habweb/efh/msa2.html (accessed March 24, 2003).

To summarize, the following characteristics of habitats imply that benefit analysis should be both spatial and intertemporal:

- Habitats are a durable stock with adjustment costs.
- Biophysical production and the social determinants of service benefits depend on the landscape context in which those functions and services arise.[90]
- Habitat value depends on scarcity and complementary goods and services.

In the remainder of the section, we illustrate with examples of how bioeconomic models can be used to help predict both the direction and the magnitude of social benefit changes associated with habitat alteration, mitigation, and restoration decisions.

5.4.2 Bioeconomic Models and Habitat Valuation

As we discussed earlier, the fishery support services of habitats are not directly traded in a market and therefore the value of the habitat for this (and other) services is not reflected in market prices. We also noted that value depends on the spatial context within which the habitat exists. To illustrate how bioeconomic analysis can help elucidate habitat values, we present three examples. The first example does not consider time or space explicitly. However, it illustrates the way in which habitat alterations' effect on commercially caught fish abundance can be used to determine the social costs of the alterations. The example also highlights how quickly habitat valuation can become complex—that is, the example will illustrate how valuation depends not just on underlying biophysical relationships, but also on the economic and institutional setting. The second example focuses on spatial factors and discusses the effects of connectivity patterns and habitat heterogeneity on habitat valuation. Finally, we introduce an example with threshold effects to illuminate their role in the assessment of costs associated with habitat alteration decisions.

5.4.2.1 Example 1: Habitat effects on fish abundance

Bioeconomic models of a fishery begin with biological processes associated with the stock of fish. These processes depend on many environmental factors, including water temperature, currents, nutrient upwelling, and habitat quality and quantity, along with the life-

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[90] For a more detailed expression of this perspective, see Nancy E. Bockstael, Modeling Economics and Ecology: The Importance of a Spatial Perspective, *American Journal of Agricultural Economics* 78: 1168, 1996.

cycle characteristics of the species. Across the marine seascape, significant heterogeneity exists in environmental conditions and species characteristics (e.g., vertebrates and invertebrates). For example, some marine species grow slowly and can live more than a hundred years and at a depth of approximately 1,000 meters (e.g., orange roughy), but others live for only a year (e.g., squid) or occupy rocky intertidal zones (e.g., red sea urchin).

With such heterogeneity, it is not surprising that modeling the growth characteristics of populations can take many forms, depending on the species under consideration.[91] For the most part, economists have been content to assume that the exploitable fish stock is enumerable in terms of either total weight or numbers, and that the principal birth, growth, and death processes can be adequately described by a small number of parameters. These "lumped parameter" models include, for example, the widely used Shaefer model,[92] in which biomass growth is characterized by the intrinsic growth rate and the carrying capacity of the environment. A logistic model is the foundation for the bulk of fisheries economics literature and is utilized here.

Let X denote the biomass level of a fish population whose interaction with its ecosystem gives rise to a biological production function, F(X). Employing the logistic growth model, which is illustrated in Figure 1, yields the following properties of F: $F(\underline{0})=0$, $F(K)=0$, $F_x(X_{MSY})=0$, $F_{xx}(X)<0$, $0\leq X_{MSY} \leq K$, where subscripts on F represent the derivative with respect to the variable. The critical points are as follows: X_{MSY} is the population size associated with maximum sustainable yield (MSY),[93] and K is the carrying capacity for the population.

Carrying capacity is defined by habitat and population characteristics, taking into account such factors as food availability and the existence of predators. We make this explicit by assuming K=K(H), where H is an index of habitat quality and quantity. We assume that as the quality and quantity of habitat increase (H↑), so will the ability of the environment to support a greater abundance of species (K↑).

The change of the fish stock from one period to the next (t to t+1) is

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[91] See Wilen (note 84) and Clark (note 84) for more information on population biology in economic models.

[92] See note 84.

[93] For current fishery management, MSY is both a target level for fish stocks and a measure from which to judge how current management systems are performing. When a population is sustained at MSY, the off-take that can be harvested without changing the population level from year to year is maximized. It is important to point out that MSY is a biological, not economic, concept. The economic optimal off-take differs depending on economic and institutional conditions.

$$X_{t+1} - X_t = F(X_t, H_t) \equiv rX_t(K(H_t) - X_t) \tag{1}$$

where r is the intrinsic growth rate of the fish stock. Equation (1) presents, in a very simplified manner, the biological mechanisms that describe how populations change over time. At low population levels, populations grow relatively fast because of an abundance of food, but as the population approaches its carrying capacity, the growth rates decrease as competition for limited resources increases. The lower growth rates could occur if individuals are exerting more effort in finding and competing for food than in propagating. In equilibrium, the population settles at the carrying capacity, a point where births are directly offset by deaths.

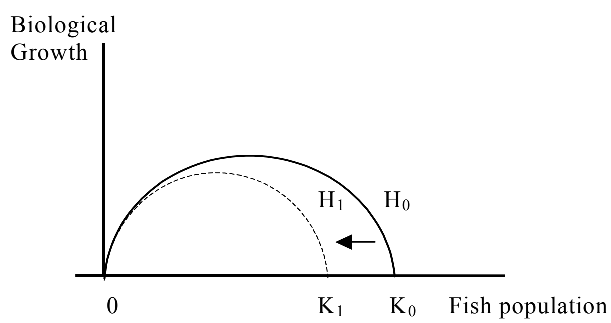

Figure 1: Biological growth function with a reduction in habitat (H_0>H_1)
Note: The two curves differ only in the level of the carrying capacity of the environment (solid line has higher K because of the higher quality of habitat).

In Figure 1, we illustrate the growth function with two different levels of habitat, H, where H_0 is larger than H_1. With H_0, the carrying capacity is larger, as illustrated in the top curve. The biological growth function with H_1 is illustrated with lower carrying capacity and lower curve. We now ask, how can we value the change in habitat from H_0 to H_1? This change might occur from either a one-time permitting decision or from a history of decisions to allow habitat alteration (cumulative impacts) when no mitigation is secured.

To illustrate how we can measure the change, we use standard assumptions made in marine resource economics.[94] First, we assume that the fish stock is exploited (C_t) and the catch level is a function of both fishing effort and level of the fish stock, C(X,E).[95] In addition, we assume that greater levels of effort imply higher catches for each level of fish stock and that higher fish stocks imply higher catches for each level of fishing effort. Second, we also assume that the price of fish (p) is exogenous—that is, the current fishery (fish stock and area combination) is not large enough to have an effect on the supply of fish in the marketplace. Finally, we assume that we are in a sustainable equilibrium where the catch is just offsetting the natural growth process in each period (X_{t+1}-X_t=0=>F(X,H) = C). Under these assumptions, we can map the growth function into the total revenue curve, where the price of the fish can scale up or down the growth function (Total revenue=p*C=pF(X,H)). We assume that total costs are simply equal to a cost per unit of fishing effort times the level of fishing effort (TC=wE, where w is the cost per unit of effort).[96]

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94 For more information, see National Research Council, Sharing the Fish: Toward a National Policy on Individual Fishing Quotas, Washington: National Academy Press, 1999.

95 Equation (1) with the catch rate is $X_{t+1} - X_t = rX_t(K(H_t) - X_t) - C_t(X_t, E_t)$.

96 This implies that all fishermen are homogeneous—for example, there are no differences in skill levels. Of course, fishermen are heterogeneous, with different skill levels; highly skilled individuals are often referred to as highliners. We will discuss the implications of the simplifying assumption on our results as we proceed through the example.

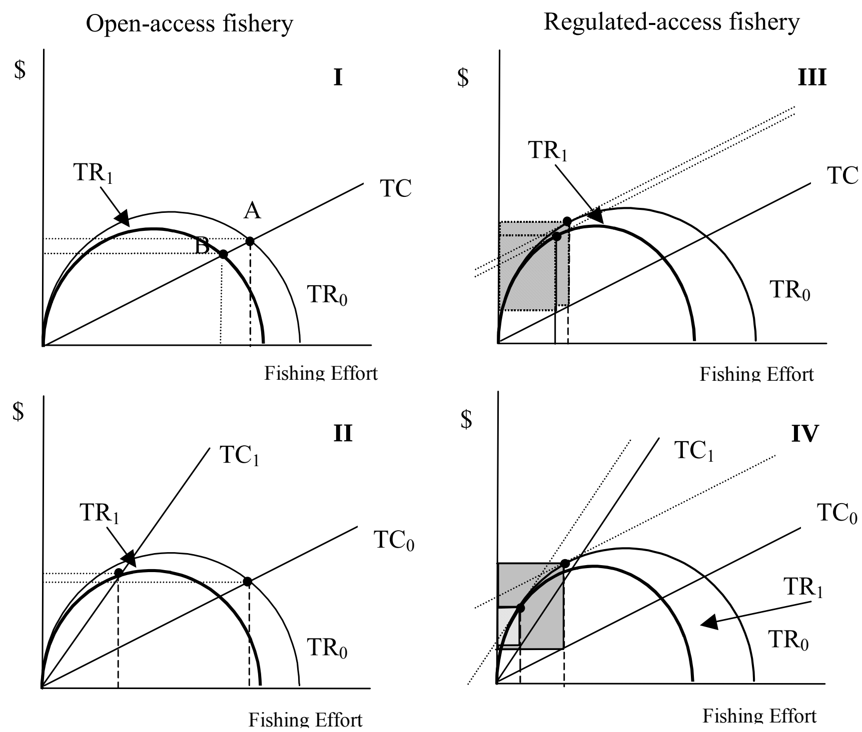

Figure 2: Examples of changes in habitat value with varying institutional structures

We represent the bioeconomic equilibrium in Figure 2, where the natural production function of the fish stock is essential for representing the total revenue curve. In panel I, the competitive equilibrium where entry and exit is freely permitted is illustrated by the intersection of the total revenue curve with the total cost curve (point A). The total revenue curves, TR_0 and $TR_{1,}$ represent the total revenue curves with the initial level of habitat (H_0) and the new level of habitat (H_1), with $H_0>H_1$. In a competitive equilibrium, we see in panel I that the habitat alteration results in a lower total revenue curve per unit of fishing effort, and in this example, total cost per unit of effort does not change. The new competitive equilibrium is at point B.

What is the cost (forgone benefit) associated with this change in habitat? The cost is the reduction in economic profits in the commercial fishery due to the decrease in fish abundance.

Under the assumptions of this example, in particular homogeneous fishermen[97] and an open-access institutional structure, total economic profits are zero before and after the change in habitat. Therefore, there is no equilibrium change in the economic condition (economic profits) of the fishing enterprises.

Although there are no long-run changes in economic profits of the fishing industry, there are still changes in the industry structure. The equilibrium point B depicts a lower level of fishing effort, which can be interpreted as an exit of vessels from the fishery. With this exodus, local employment levels could be affected. The Regulatory Flexibility Act requires that these changes, especially with respect to small business, be documented. In this case, additional costs of the habitat reduction will be the lost wages, early retirement of vessel capital, and lower output of processing plants—all of which can affect the regional economy. The restructuring and displacement of the fishing industry could be large, as we show in panel II, where total revenue declines and total costs increase ($TC_0 \rightarrow TC_1$ as $H_0 \rightarrow H_1$). Total costs per unit of effort could increase if the reduction in habitat reduces the "fishable" habitat, exacerbating or even creating congestion externalities.[98] Other factors that could increase the cost per unit of effort after habitat alterations include the displacement of fishing effort from areas close to the ports to fishing grounds farther offshore.

It is important to note that although the cost associated with the restructuring and displacement of the fishing industry is very important from a socioeconomic perspective, these are not efficiency costs because the displaced resources can be dedicated to an alternative use. The efficiency cost in this example is the dissipation of the social surplus (rents) associated with the limited supply of the resource both before and after the reduction in habitat. The efficiency

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97 Because we are assuming homogeneous fishermen, there are only resource ownership rents present in the fishery—that is, if the resource can be appropriated. With heterogeneous fishers, there is also the potential for factor rents associated, for example, with different skill levels. In this case, the fishermen at the margin earn no economic profits, but all of those with higher skill levels do earn positive rents.

98 A congestion externality results when fishing vessels interfere with one another during search or capture activities, increasing the costs of fishing for each vessel in the fishery. For example, Boyce (2002) describes the Bristol Bay salmon fishery in Alaska, where all fishing is regulated to occur within one mile of the mouth of the Naknek River. Given the area restriction and the seasonal nature of the fishery, fishermen end up sitting idle in queues and employing gear technologies to reduce entanglements, both of which increase their costs. John R. Boyce, Comment: ITQs and Externalities, *Marine Resource Economics* 15(3): 233–44, 2002. See note 101 for a discussion on how implementing rights structures defined over area and species combination can eliminate both congestion and stock externalities.

costs are due to the overinvestment in harvesting capital and labor under open-access settings relative to the socially optimal level.[99]

In panels III and IV, we illustrate that the institutional context is critical to benefit assessment. We assume in panels III and IV that the fishery is managed with some form of regulated access, such as limited-entry systems, individual fishing quotas, or cooperatives. We also assume that the system is able to maintain the profit-maximizing output level (i.e., maximizing resource ownership rents and rents to effort), or in other words, the regulations are working efficiently. The profit-maximizing output level for the fishery is reached when effort in the fishery is at a level where the marginal revenue product of effort equals marginal effort cost, or when the slopes of the total revenue and total cost curves are equal. We can find the equilibrium level by shifting up the total cost curve until we find the point of tangency between the two curves.

In panel III, the total profit (i.e., rents to the owners of the rights to the fish harvest) at the optimal equilibrium level is shown by the shaded rectangles. The solid gray rectangle corresponds to the total profit at the initial habitat level, and the rectangle with the crosshatching corresponds to that with the new, altered habitat. The economic value (or loss) associated with the change in habitat in the fishery is simply the difference in the size of the two rectangles (levels of total profit). If, for example, the fishery were operated with a licensing limited-entry system, the license price (L) would be equal to the present discounted per unit value of effort in the fishery. And the total profit in the fishery would be the license price times the level of fishing effort operating (Profit=L*E).[100] The loss due to the habitat change would simply be measured by the difference, $L_0E_0–L_1E_1$, where the subscripts correspond to habitat levels. In this setting, there is a change in economic profits because the fishery is managed in such a way as to ensure positive long-run economic profits. The reduction in profits is larger in panel IV than in III

[99] Since both the open-access and socially optimal effort levels will change when the habitat is altered, it is not clear *ex ante* whether the efficiency costs will be larger before or after the habitat is altered. This does not imply that, from an economic perspective, we are better off with more habitat alteration. For example, the lower catch could reduce the consumer surplus when the price of fish is responsive to the supply (e.g., demand for the fish is downward sloping). In this example, we assume for simplicity that the price of fish is exogenous to the fishery where the habitat degradation occurs.

[100] We continue to assume that fishermen are homogeneous. When fishermen are heterogeneous, the license price will equal the present discounted profits of the fishermen on the margin. Fishermen with greater skills will be earning higher rents than the license price. If the licenses are stackable (fishers can purchase more than one), then we would expect the fishery to consist of the low-cost or highly skilled fishermen in the long run.

because of the increase in the costs per unit of effort due, for example, to congestion externalities.[101]

In addition to the loses in economic profits in the examples illustrated in panels III and IV, there are the same adjustments that go along with industry restructuring—such as reduced employment in the fishing and processing sectors, boat mechanics, and so forth—that existed when there were no changes in economic profits. If these resources have no alternative use that provides equivalent income, then the difference in returns in fishing and the next best alternative income is a lost value associated with the habitat loss. Typically, it is assumed that after a transition period, the displaced resources are fully employed in other endeavors.

The data and science needed to measure the changes in the fishing industry as illustrated in Figure 2 include the following: vessel prices (dock prices), fishing effort costs, a measure of fishing effort, description of the catch production function, biological production function F(X,H), and the biophysical production function relating habitat quality to the carrying capacity (K(H)).

5.4.2.2 Example 2: Spatial analysis of habitat relationships

In a featureless or uniformly homogeneous landscape or seascape, a bioeconomic analysis that included aggregate or system-wide production relationships between habitat and marine life populations would suffice to measure and value changes. However, habitat is not homogenous across many of the scales at which permitting decisions are made. Furthermore, habitat is not homogeneous across the scales at which habitat, physical, and species interactions occur. Capturing these broader biophysical and socioeconomic effects requires taking a seascape or regional perspective. Only in this broader context will it be clear how removing a piece will affect the functioning of the whole ecosystem.

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[101] It is important to point out that in the presence of congestion and stock externalities, two instruments are required for the efficient solution. For example, an individual fishing quota (IFQ) right is typically defined in terms of share of the total catch, and the creation of this right addresses the stock externality associated with common pool resources. If there are spawning aggregations or low-cost areas to catch fish, these areas could be subject to the hotspot effect. It is also possible that there are still congestion externalities, such as those in the Bristol Bay salmon fishery (see footnote 98). To address these, the IFQ rights could be defined based on area and fish combinations, where the areas are defined at such a scale as to eliminate the congestion effects. It could also be that another instrument, such as vessel participation fees, could be used in addition to the IFQ management structure. See, for example, John R. Boyce, Individual Transferable Quotas and Production Externalities in a Fishery, *Natural Resource Modeling* 6(4): 385–401, 1992.

Ecologists and economists increasingly focus on the role of space in biological systems and the manner in which spatial characteristics affect fundamental processes. Of particular importance are notions of resource patchiness and heterogeneity, biophysical linkages, and dispersal mechanisms connecting patches. This research has led to a new class of bioeconomic models that are composed of a group of subpopulations distributed across a set of patches linked by dispersal processes.[102]

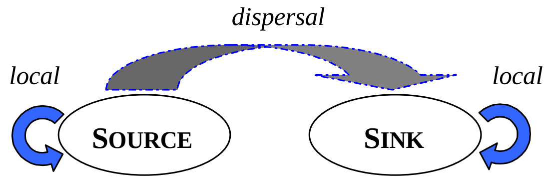

Figure 3: Source-Sink Ecological Structure

Note: The arrow linking the source and sink patches represents dispersal, and the arrows feeding back into the patches represent local production.

This example expands the above analysis by introducing another patch or fishing ground into the analysis. We can then investigate how habitat alterations in one area ripple throughout the system and affect uses in other areas. Understanding the connectivity or ripple effects is critical for assessing the total, not just the local, costs associated with habitat changes. In this example, the mechanism behind the ripple effects is the dispersal of biomass (fish populations) from one area to the next.

To simplify the analysis, we focus on a source-sink ecological structure in which the biomass is flowing in a unidirectional manner from the source area to the sink.[103] The system is

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102 See, for example, James N. Sanchirico and James E. Wilen, Bioeconomics of Spatial Exploitation in a Patchy Environment, *Journal of Environmental Economics and Management* 37: 129–50, 1996; and J.N. Sanchirico and J.E. Wilen, A Bioeconomic Model of Marine Reserve Creation, *Journal of Environmental Economics and Management* 42: 257–76, 2001.

103 Although it is not uncommon to characterize marine systems as sink-source systems, there is some debate about whether the notion applies directly. The notion of a sink was originally developed to depict more closed terrestrial systems in which local reproduction needed augmentation from outside sources to offset mortality. Marine systems are typically more open, and local populations are almost always replenished via transport mechanisms from parent populations located elsewhere. For many populations of marine organisms, the offspring of parent populations located at a particular place replenish other locations, and the extent to which they do so relative to recruitment

depicted in Figure 3. There are many other possible configurations we could discuss, such as those in which dispersal depends on relative densities, but this structure is sufficient to illustrate the importance of taking a regional, as opposed to local, perspective to habitat valuation.

In what follows, we discuss the formulation of this two-patch source-sink ecological system, in which local growth processes in each patch are logistic. Let X be the sink and Y the source population, and assume that price per unit of output is equal across the two areas. We also allow for differential biological productivity of the two patches. The equations defining the growth functions, including patch catch rates, are as follows:

$$\begin{aligned} X_{t+1} - X_t &= r^X X_t(K(H^X) - X_t) + bY_t - C(E^x, X) \\ Y_{t+1} - Y_t &= r^Y Y_t(K(H^y) - Y_t) - bY_t - C(E^y, Y) \end{aligned} \qquad (2)$$

where the superscripts define the patch-specific variables (E) and parameters (r, K). For example, r^x is the intrinsic growth rate in patch x, and can be different across the patches because of local oceanographic and ecological factors. H^i is habitat quality index in patch I (I=x,y), and this can also differ either because of biogeographic characteristics or because one area has experienced more habitat alteration.

As evident from Equation (2), the change in biomass in the source area (Y) from one period to the next is a function of the biological growth ($r^Y Y_t(K(H^y) - Y_t)$) net the biomass that migrates to the sink area (bY) and the catch in the area ($C(E^Y,Y)$). The change in biomass from the sink is also a function of local growth ($r^X X_t(K(H^X) - X_t)$) plus the dispersal from the source (bY) minus the catch ($C(E^Y,Y)$). The amount of biomass leaving the source area in any period is bY, and exactly that amount arrives in the sink area, where b is the dispersal rate. This is equivalent to assuming that there is no natural mortality during the dispersal process. In a no-fishing equilibrium (catch rate is zero), the source patch population will be maintained, with positive net growth balanced by emigration; in the sink patch, negative net growth will be augmented by immigration.

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supplied to their own parent population determines the relative role of the location as a sink. This is why understanding larval transport mechanisms and the fate of larvae is critical to sensible policymaking about habitat alteration decisions in marine systems. See H.R. Pulliam, Sources, Sinks, and Population Regulation, *The American Naturalist* 132: 652–61, 1988.

It is important to point out an asymmetry in the system due to the source-sink structure, stemming from the unidirectional dispersal pattern. This will have important implications for assessing habitat alterations in the source or sink area. To foreshadow our results, we will show that habitat in the source area should not be traded off one-for-one with habitat in the sink area. The reasons will become clear when we explore the effects of changes in habitat in each area.

We continue to assume that total revenue and cost functions have the same functional forms as in Example 1, but we allow the total costs to differ because of heterogeneity in patch-specific cost parameters (e.g., w^x and w^y would replace w in Example 1). For example, the marginal cost with respect to effort could be higher in the source area because of local oceanographic features that preclude certain gear types along with differences in transportation costs to and from the port. Also, we will focus only on the regulated-access fishery, where the optimal fishing effort level is obtainable. The results for the case of the open-access fishery follow directly from Example 1—that is, under our assumptions, there is no change in equilibrium profits before and after the change in habitat.

We ask three questions: (1) What is the cost associated with habitat alteration in the source area? (2) What is the cost associated with habitat alteration in the sink area? And (3), what is the implication of these different costs and effects for mitigation where changes in either the source or sink are mitigated by changes in the other area? Each question is taken in turn.

Suppose a permitting decision will reduce the quality of the habitat in the source area (H^y). As in Example 1, the reduction in habitat quality in the source area will shift down the total revenue curve. But this also reduces the equilibrium population level in the source area and therefore affects the level of biomass that disperses to the sink area. The lower influx of biomass from the source area shifts down the total revenue curve in the sink area. The total value associated with a change in habitat in the source area is the sum of the loss in profits in the source and sink area. Focusing only on the local effects of the habitat alteration would lead to underestimating the costs by an amount equal to the losses in the sink area. We represent these changes in Figure 4. Notice that these panels are similar to those in Figure 2, except now we have two rather than four.

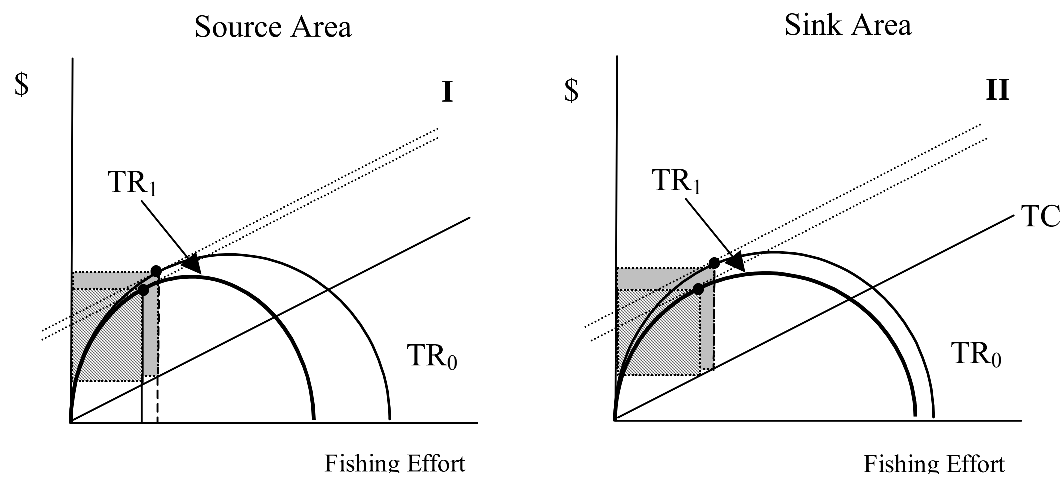

Figure 4: Measuring Habitat Changes in the Source Area

Now suppose that there is a change in the sink area. We illustrate this example in Figure 5, below. In this case, the local perspective, which is measuring value only within the habitat that is directly affected, will not underestimate the changes because there are no ripple effects beyond the sink. This is the direct result of the source-sink formulation and is not applicable to situations where biomass is flowing between patches based on relative densities. The change can be measured, as in Example 1, by the difference in the area of the two rectangles that represent the reduction in profits in the fishery.[104]

What will determine the magnitude of the changes in habitat value in this source-sink system? The short explanation is that it will depend on all of the components of the models, in both absolute and relative ways. To be more specific, it is a function of both the mechanisms and the magnitude of the effects along the casual chain from the source to the sink. The biological chain of events is (1) habitat alteration affects habitat quality in the source; (2) habitat quality affects the source population size; and (3) the size of the source population affects the level of

[104] It is possible that the local perspective could underestimate the total, even without biomass dispersal connecting the two patches. For example, in Figure 4, if the reduction in the fishing effort in the sink area simply shifted into the source area and affected the cost of fishing there (raising the total cost curve), there would be a reduction in profits in the source area from a reduction in habitat quality in the sink area. But in our example, we assume for simplicity (and to make the point that ecological asymmetries will affect habitat value) that the fishing costs in the source patch are unaffected.

dispersal to the sink. The magnitude of the reduction in habitat quality on the source population will affect the reduction in the total revenue curve in the source, and the reduction in the level of biomass dispersing to the sink will affect the size of the shift in the total revenue curve in the sink area. The magnitude of these changes depends on the biological parameters in each area, the costs of fishing in each area, the dispersal rate, and also the habitat and environmental conditions in the patches prior to the habitat alteration in the source.

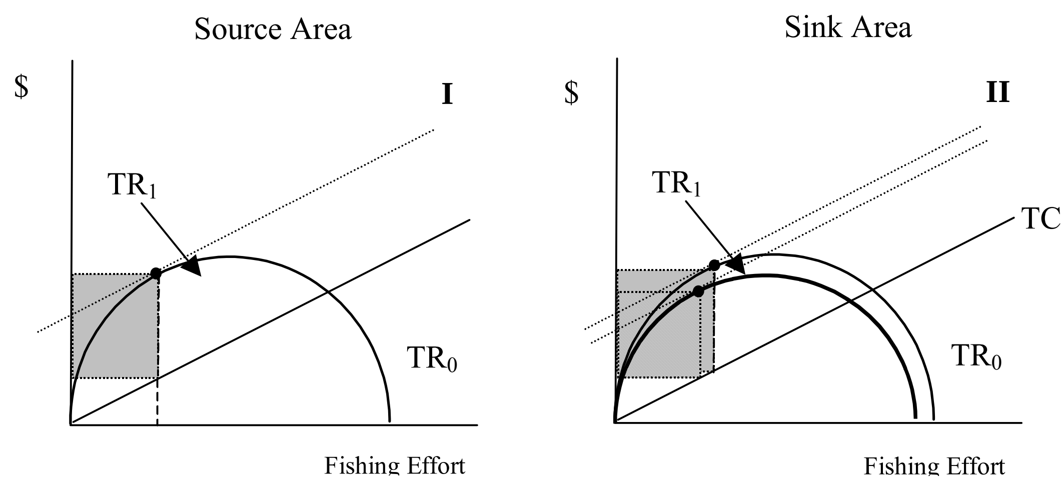

Figure 5: Measuring Habitat Changes in Sink Area

Although this example is stylized, it illustrates that simply adding a spatial dimension to the question of habitat valuation is not a marginal change. Rather, it increases the complexity and data requirements significantly. But as we discussed in Example 1, much of this information is becoming increasingly available to researchers. Furthermore, although the dispersal of larvae and adults (and juveniles) in the marine environment is complex and poorly understood, there is optimism among marine scientists that improvements in genetic analysis, mark-and-recapture methods, otolith geochemistry, and oceanographic models of currents, gyres, and coastal upwelling processes will lead to a better understanding of these processes.[105]

Our third question concerns mitigation strategies. Consider first the case of habitat alteration in the source area. Recall that the total revenue in the sink will fall (shift toward zero)

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105 Otoliths are ear bones on fish that are inert and, via chemical analysis, provide a natural way to determine the environment in which fish live, which leaves a chemical signature. Mark-and-recapture studies use artificial tags to determine dispersal pathways.

because of the lower amount of spillover. Mitigating this change by increasing the habitat quality in the sink area will, however, shift out the total revenue curve in the sink. These two effects counteract each other, implying that it is more likely that additional habitat would need to be set aside in the sink (e.g., a greater than one-to-one compensation ratio), everything else being equal. On the other hand, offsetting a change in sink habitat with mitigation in the source might lead to a mitigation ratio less than one, because some of the loss in sink habitat is also mitigated by the additional dispersal from the source area. This example illustrates that the appropriate mitigation ratio depends not only on the economic and ecological characteristics of the habitats, but also on the connectivity across the patches.

5.4.2.3 Example 3: Threshold effects

Cumulative impacts were illustrated in Examples 1 and 2 implicitly through changes in the stock of habitat quality. In those cases, the equilibrium values associated with disinvestment in habitat resulted in populations above zero. However, this is not always the case, especially when there is a threshold in the population growth function. A threshold could exist if, for example, there is a minimum viable size of the population below which finding a mate becomes difficult. If the population falls below this level, then the species goes extinct.[106] A series of habitat alterations and the corresponding cumulative loss could drive the fish population level below the threshold. The collapse of the fish population will, of course, result in a collapse of the fishing industry and a potentially irreversible loss in that habitat value.

[106] There is a growing and significant body of literature on terrestrial systems and protected areas; researchers are trying to model the critical size for different populations, where across species the characteristics that matter are dispersal and reproductive success at small population sizes.

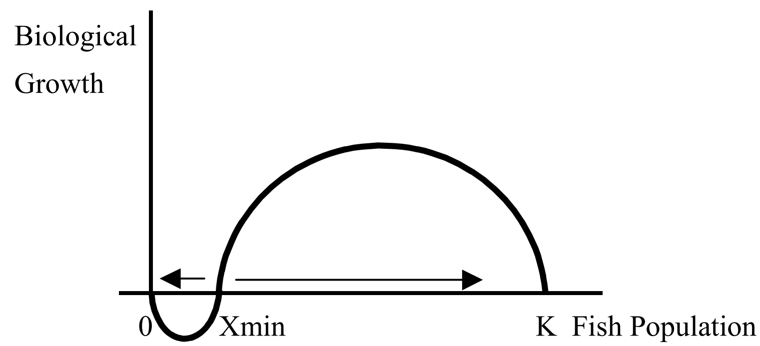

Figure 6: Population Growth in the Presence of a Critical Minimum Threshold

Example 3 includes a threshold effect and illustrates the difficulties involved in valuing habitat when critical thresholds in the population level exist. Before we graphically illustrate the example, however, we generalize Figure 1 and Equation 1 to include critical values. Equation 1 can be altered to include the minimum population size as follows:

$$X_{t+1} - X_t = F(X_t, H_t) \equiv r(X_t - X_{\min})(K(H_t) - X_t) \tag{3}$$

where X_{min} is the minimum viable population size. Upon inspection, if X_t is less than $X_{min,}$ the population level will decrease from period t to t+1, and if it is greater than $X_{min,}$ the population level will increase.

We graphically represent a growth function with a critical minimum viable population size in Figure 6. The arrows represent the fact that if the population is above the minimum critical level, it will grow without fishing until it equals its carrying capacity. The arrow to the left of the critical level illustrates that if the population level falls below, the population will go extinct. How fast this occurs depends on the biological characteristics of the population.

Appending a catch function to Equation (3), we derive a total revenue curve that is a function of the biological growth function. Unlike in Examples 1 and 2, the total revenue curve does not return to the axis at high effort levels. Instead, there is a critical effort level (E_{max}) that corresponds to the critical population level. If effort exceeds this level, then the population is driven below X_{min} and goes extinct. Unlike the previous figures, we also include a line representing the equilibrium population levels that correspond to the equilibrium fishing effort levels. When the fishing effort level is equal to zero, the fish population level equals its carrying capacity. At high effort levels, the fish population decreases. Including the information on equilibrium population levels illuminates the way in which habitat changes can drive a population to extinction.

In Figure 7, we again graphically represent the solution where a habitat alteration reduces the population level in the case of an open-access resource. We also consider only one patch. Later we discuss the impact of more than one patch, building on our results in Example 2. And although not illustrated, the intuition developed for the regulated access setting continues to hold. In the top panel of Figure 7, the equilibrium effort level is found at the point where total revenue equals total cost. The resulting equilibrium population level, which is above the threshold level, is also illustrated with the point along the fish axis.

Now suppose that the level of habitat decreases (H_0 decreases to H_1). This will result in a downward shift of the total revenue curve, which is illustrated in the bottom panel. The TR curve with H_0 is also illustrated with the dashed line. The change in habitat lowers the carrying capacity and reduces the distance between the critical threshold and the carrying capacity. We do not assume that X_{min} is affected by a change in habitat, because X_{min} is typically considered to be a function of the species characteristics as opposed to the environmental conditions of the area. The change in the potential total size of the population that can be supported by the new lower level of habitat is illustrated by a shorter axis in the bottom panel.

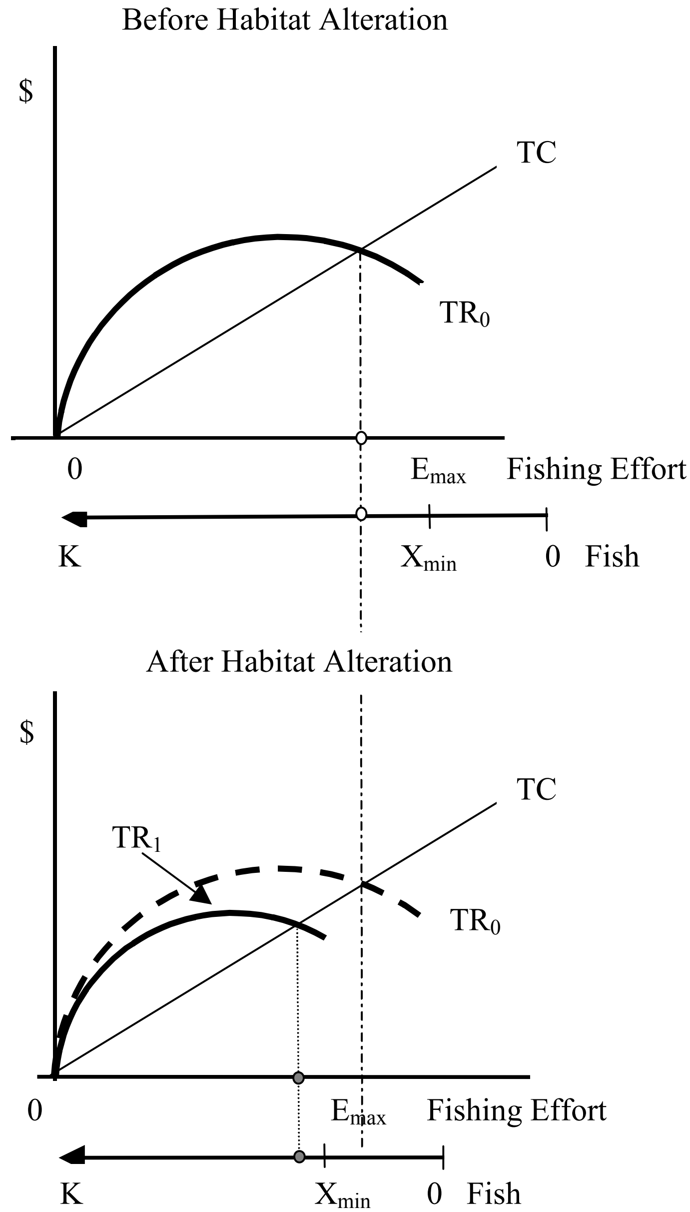

Figure 7: Habitat Alterations and Biological Thresholds

What are the implications of habitat changes and thresholds under our assumptions? First, at the equilibrium, the population is positive, implying that the current level of habitat is sufficient to maintain a population (bottom panel). Recall that there is no efficiency cost associated with the change because we are operating under an open-access regime with homogeneous fishermen (see footnote 90). However, the fact that the population level is closer to the threshold could be worrisome because another exogenous shock to the system could drive the population level below X_{min}.

Second, we choose this example because it illustrates the transition from one equilibrium level to another. As we have constructed the example, the level of fishing effort before the habitat alteration is greater than the level of E_{max} after the alteration. If fishing effort is slow to respond, it is possible that the fishing effort will be too high in the initial periods after the alteration and will drive the population level below X_{min}. On the other hand, if fishing effort responds quickly to the losses (by exiting almost instantaneously from the fishery), then the probability of extinction during the transition is lower, everything else being equal. The speed of response of the fish population to the changes in the habitat will also affect the probability of extinction during the transition. In fact, it will be the relative speed between the fishing effort and fish population response that will be the critical determinant of whether the population level goes below X_{min}.

Adding a spatial component to this analysis, as we did in Example 2, will introduce additional richness to the model, but the intuition derived without thresholds still stands. There is one slight difference, however. Because of the spillover of fish from the source area, the population in the sink habitat is less likely to go extinct even in the presence of the threshold. For example, if the population drops below X_{min} in period t, fish will be dispersing to the sink in period t+1, potentially pushing the population level above X_{min}. Therefore, small and potentially reversible habitat alterations in the sink are unlikely to lead to extinction, the greater the flow of biomass from the source. On the other hand, driving the population in the source to extinction via changes in the habitat could lead to extinction across the entire system. Therefore, just as we discussed earlier, permitting decisions need to be developed within a regional plan that encompasses the relevant ecological and economic scale of the problem.

In many cases, the level of the threshold (along with many other parameters) is unknown, and therefore NMFS is faced with making consultations and decisions under uncertainty. In general, the nature of the uncertainty and learning over time will affect whether a more precautionary approach or one that allows greater extraction than under certainty is followed. We can make some suggestions for how to approach habitat alteration from the large body of literature on exhaustible resource economics. This literature has for the most part focused on how to deplete a resource (here, habitat) over time when the stock of the resource is of unknown size.

When learning occurs during extraction, uncertainty in the stock size implies a slower rate of extraction than what would occur under certainty.[107] On the other hand, Pindyck (1980) shows that the resource might be exploited at a faster rate than under certainty, when the effects of uncertainty fluctuate continuously and stochastically over time.[108]

In this example, the uncertainty is associated with irreversible damages (collapse of fish stock), which adds another layer of complexity. In the context of greenhouse gases, Narain and Fisher (forthcoming) show that uncertainty increases the importance of early action, when damages are irreversible. They also show that endogenous risk—that is, the likelihood that damages are affected by the stock of the pollutant (in our case, the cumulative effects of habitat alteration)—also tends to promote early action.[109]

Spatial interdependence has also been shown important for irreversible decisions made under uncertainty. In the case of tropical forest management, whether ecological irreversibilities occur is a function of the spatial configuration of forested plots to provide seed sources. Albers (1996) finds that the value of waiting is higher with certain spatial patterns than with others, and that without taking this structure into account, tropical deforestation is more likely.[110]

Given the characteristics of habitats, it is likely that a precautionary approach is justified where the changes in habitat quality are lower with uncertainty than without. It is important to point out that this does not imply that no habitat is ever to be altered. One precautionary approach is to have the mitigation currency be in biophysical production units rather than on a per-acre basis. That would maintain the biophysical productive capacity of the habitat, even though habitat alterations are permitted. The spatial pattern of alteration and mitigation will also matter, as illustrated in Example 2 and the work on tropical deforestation.[111] In fact, a habitat mitigation

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107 See, for example, R.J. Gilbert, Optimal Depletion of an Uncertain Stock, *Review of Economic Studies* 46: 47–58, 1979.

108 Robert S. Pindyck, Uncertainty and Exhaustible Resource Markets, *Journal of Political Economy* 88: 1203–25, 1980.

109 Urvashi Narain and Anthony Fisher, Irreversibility, Uncertainty, and Global Warming: A Theoretical Analysis, *Environmental and Resource Economics,* forthcoming.

110 Heidi J. Albers, Modeling Ecological Constraints on Tropical Forest Management: Spatial Interdependence, Irreversibility and Uncertainty, *Journal of Environmental Economics and Management* 30: 73–94, 1996.

111 *Ibid.*

decision that reproduces biophysical production in another area might actually increase the level of production.[112]

5.5 Model Implementation

It should come as no surprise that developing empirical models for the concepts illustrated by these examples is not an easy task. First, data on fishing costs are limited at best, but there are current efforts at NOAA fisheries to improve the quality and quantity of the information. Second, fishing effort is sometimes difficult to quantify, depending on the fishery involved. For example, for a fishery using dive gear, it would be the number of dive hours, but with a trawl fishery, it would be a composite index of fishing power (vessel length, crew size, horsepower, and so on). Third, we often do not have models of fishing abundance that are readily available, but that too is changing. Finally, the science on measuring the biophysical production function, especially with an explicit consideration of sources and sinks, is being developed and estimates are coming on line, but the science is still in its infancy.

The data issues are manageable on a longer time horizon and can be accelerated with increases in interdisciplinary project funding (we discuss the appropriate scale of these efforts below). In the meantime, NMFS could represent benefits and costs based on the expected parameter levels and provide consultations based on the expected values. Ideally, uncertainties and the sensitivity of the results to such uncertainties should be assessed. This can be accomplished using Monte Carlo techniques, where each parameter is assigned a probability distribution. Such a method was discussed in the King et al. (2003) report.[113] Essentially, one can assign current best-guess probabilities to each parameter estimate, where a lower probability implies that one is surer of the estimate's true value. One then draws from a distribution of the different parameters levels to derive a schedule of expected changes in value. The simulated "empirical" distribution of outcomes can then be assigned a mean and standard deviation. It is desirable, of course, to revise these parameters based on learning about the ecological and economic systems that occurs over time. Therefore, there are benefits to ensuring the funding for long-term studies and model development and refinement in support of the habitat alteration decisions.

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112 See, for example, James N. Sanchirico, Additivity Properties of Metapopulations: Implications for the Assessment of Marine Reserves, *Journal of Environmental Economics and Management*, In press, 2004.

113 See note 83.

6. Benefit Indicators

In Section 4, we discussed nonmarket benefit estimation—what we refer to as benefit monetization. We argued that because these methods are complex, expensive, and controversial, they are best applied to relatively large-scale decisions where there is an expressed demand for such evaluations and where the necessary dollars for assessment are available. In this section, we describe an alternative: ecosystem benefit indicators. A motivation for the use of indicators is that economic benefit assessment will be most powerful at the field office level when it conveys a commonsense appreciation of benefits, rather than a potentially intimidating and controversial reliance on monetary estimation procedures. Indicators have drawbacks of their own, as we will discuss. But benefit indicators are an additional tool NMFS could consider to motivate recognition of habitat and ecosystem service value.

As we will argue, indicators signal underlying value and can thereby help structure agency deliberations over ecological benefits. But indicators are not a calculation of actual dollar values. Because of this, indicators are just as controversial as nonmarket valuation. Monetary values are the *sine qua non* of economic assessment because they provide a common metric by which otherwise noncomparable actions or outcomes can be compared.[114] We discussed these issues in detail in Section 4.2. Nevertheless, we argue that economic principles and proxies for value can serve as a useful input to regulatory deliberations.

There are several advantages to indicator systems. First, they can be carefully structured to help decisionmakers organize their thinking about valuation, communicate ecosystem service benefits, and support landscape analysis and the communication of spatial interdependencies to decisionmakers and stakeholder groups. Second, indicators may be less expensive to generate, allowing for landscape assessment of multiple services at large scales. Third, because they rely less on econometric statistical techniques and more on geographic information system (GIS) analysis, ecologists and other noneconomists may have an easier time assembling indicator-type results. These properties may allow for the diffusion of benefit assessments to small-scale field office permit reviews and other consultations. By design, indicators employ existing GIS data to highlight areas where ecosystem services are likely to be generated and areas where services are

[114] It is important to point out that this does not imply that economists believe everything needs to be quantified in dollars. If another metric could be agreed upon by all parties in an exchange, then the use of monetary assessments would not be necessary. But the transaction costs associated with the decision process regarding the acceptable metric are often very high, especially when there are many consumers and producers over time and space.

likely to be particularly valuable. The goal of such a system is a relatively inexpensive benefit characterization that noneconomists can apply in the field. Third, indicators can complement the development of benefit transfer studies involving nonmarket valuation. Indicators are quantitative measures of site characteristics and can therefore be used to calibrate benefit transfers.

Benefit indicators, bioeconomic models, and nonmarket valuation share a foundation in the principles of economic assessment. For example, both focus on the services generated by ecosystems and both require ecological analysis as a fundamental input. The economic principles discussed in Section 5.1 will be mirrored by a similar discussion in Section 6.3 below. For an indicator system to provide economic insight, it must be rooted in the same core economic principles. For this reason, the indicator system is explicitly organized around such concepts as demand, scarcity, substitutes, complementary inputs, and risk. A goal of the indicator system is to educate decisionmakers about the importance of these concepts to their resource allocation and trustee decisions.

It should be noted, however, that ecosystem benefit indicator methods are in a developmental phase. NMFS should be aware that the methods described below are not a standard approach to economic assessment, but rather a proposed method motivated by assessment challenges facing agencies like NMFS and made possible by new types of spatial data and GIS analysis tools.[115]

6.1 Overview

In the same way that conventional "ecological indicators" use observable site characteristics to signal a site's biophysical qualities, benefit indicators can also characterize an ecosystem's social value over space and time—as well as the potential for threshold effects. As noted earlier, ecological indicators are themselves indicators of social value. A site's ability to

[115] One of this study's coauthors (Boyd) and one of the companion study's coauthors (King) are advocates of this approach. For a detailed explanation of this method, including case studies, and the motivations for benefit indicator evaluation, see James Boyd and Lisa Wainger, Measuring Ecosystem Service Benefits: The Use of Landscape Analysis to Evaluate Environmental Trades and Compensation, Resources for the Future Discussion Paper 02-63, April 2003; James Boyd, Dennis King, and Lisa Wainger, Compensation for Lost Ecosystem Services: The Need for Benefit-Based Transfer Ratios and Restoration Criteria, *Stanford Environmental Law Journal* 20, 2001; James Boyd and Lisa Wainger, Landscape Indicators of Ecosystem Service Benefits, *American Journal of Agricultural Economics* 84, 2002; and Lisa Wainger, Dennis King, James Salzman, and James Boyd, Wetland Value Indicators for Scoring Mitigation Trades, *Stanford Environmental Law Journal* 20, 2001.

support diverse species, for example, is not only an indicator of biological condition, but also an indicator of its social value. But there is a much wider range of benefit indicators that can be brought to bear on ecosystem evaluations.

Ecosystem benefit indicators are made possible by the increasing availability of spatial datasets and GIS software. GIS data are now widely available and relatively cheap. They are also inherently suited to the analysis of spatial interrelationships at varying scales. As we have argued, spatial interactions are the key to assessing social benefits. Accordingly, GIS analysis allows for quantitative, spatial assessment of habitat. Spatial landscape data can be depicted visually and often yield a particularly intuitive assessment of the way in which a site contributes to conditions in the broader landscape. When GIS data are analyzed and organized around economic principles, they become a potentially powerful tool for the analysis of ecosystem benefits.

An indicator approach to ecosystem benefit assessment can be broadly outlined as follows:

First, collection and assembly of base landscape data coverages, including both biophysical and socioeconomic datasets.

Second, identification of the ecosystem services of interest in the site evaluation. If a "total service" analysis is desired, a wide range of service flows will be identified and qualitatively characterized.

Third, construction or manipulation of base data coverages with GIS tools and software to calculate specific indicators that relate to the provision of the services of interest.

Fourth, organization and selection of indicators on the basis of their ability to signal ecosystem benefits, service by service. Some indicators—typically biophysical indicators of ecosystem function—relate to the underlying production of the ecosystem; others relate to the demand for, and value of, the service.

Fifth, benefit indicators can be selected, combined, and translated into benefit "hotspot" maps at different spatial scales. Hotspot maps represent the ultimate goal of an indicator analysis. Watersheds, counties, regions, or states can be spatially characterized on the basis of areas likely to generate valuable ecosystem services.

We now describe the indicator method in more detail. In particular, we describe the types of data available and the way in which economic principles can be applied to the derivation and interpretation of ecosystem benefit indicators.

6.2 Benefit Indicator Data

An advantage of an indicator-based system is that new data need not be generated for a study or site assessment. The federal government, states, counties, and even localities can provide a huge array of GIS data in readily accessible form. Land use, demography, public infrastructure, and biophysical data are widely available. Vast amounts of the socioeconomic data are centrally distributed through the Census Bureau and are aggregated by census block or block group, city, census tract, county, state, and the nation as a whole. Federal and state databases, for example, cover census data by census tract, road networks, parks, historic sites, aquifers, and topography, among other things. Regional economic databases are available from several sources, such as the USDA National Agricultural Statistics Service and the U.S. Bureau of Economic Analysis, for evaluating the economic characteristics of counties, congressional districts, and other units of various sizes.

Indicators relating to the following kinds of data can be constructed: population and other demographic data; housing and commercial and industrial buildings; important sites, such as schools and museums; the value of homes, buildings, and infrastructure; public and private water supplies; biophysical functions; riparian and coastal characteristics; rare and endangered species; recreational opportunities; land cover; future land use; watershed land cover; floodplain characteristics; and roads and trails.

Note that most of these data characterize the terrestrial rather than the marine landscape. Spatial biophysical and economic data on the marine environment are less available. Therefore, the use of indicators may be limited to services that are primarily determined by terrestrial characteristics (including surface water). Note, though, that services associated with habitat on which NMFS consults are often dependent on terrestrial characteristics. To get a flavor of the spatial data types available, consider Table 1.

Table 1: Spatial data types

	Demographic		*Land Use and Land Cover*
	Total population		Cropland and pasture
	Population density		Livestock operations
	Households		Developed land cover
	Children		Non-ag natural land cover
	Income		Natural and pasture areas
	Educational attainment		Upland forest
	Racial composition		Aquatic preserves
			Protected lands
	Real Estate		
			Infrastructure
	Housing units		
	Median housing value		Major roads
	Median rent		Trails
	Commercial units		Permitted wells
	Value of Commercial Units		Private drinking wells
			Density of Private Wells
	Biophysical		
			Public Sites
	Watersheds		
	Wetlands		Recreational areas
	Floodplains		Schools
	Elevation		Culturally important sites
	Seagrass beds		
	Invasive species		*Planning*
	Rare and endangered species		
			Future development
			Infrastructure development

The table lists the broad types of GIS data that—at least in many areas—are already available for analysis, though there are differences in the quality and depth of the databases. To be clear, however, basic GIS datasets provide only the starting point for evaluation of ecosystem benefits.

First, raw GIS data are only occasionally adequate in and of themselves. Population density data from the census, for example, are directly available and useful in their own right. More typically, GIS maps need to be constructed, or extracted, from primary datasets. This may mean that mapped features for an ecosystem benefit analysis are a subset of a larger set of features depicted by an available GIS coverage. In other cases, data will need to be merged to derive an indicator of interest. For example, to learn the percentage of riparian wetland in proximity to a specific habitat, NMFS will have to merge datasets and perform calculations on the compound dataset. Such GIS analysis permits a wide variety of spatial calculations, including distances between two points or areas; the presence of a certain feature, or the number of features, within a specified distance; the percentage of an area that has a particular characteristic; or the density of that characteristic. It can also be useful to measure the connectivity of a certain feature with other landscape features.

Second, the data must be assembled and analyzed in a rigorous way that reflects both ecological and economic principles. The accessibility of benefit assessment to noneconomists and cross-disciplinary communication between economists, ecologists, and other stakeholders will be fostered by tools that can unite existing data with fundamental valuation principles and support the necessary public debate over relative values.

6.3 Using Economic Principles to Design and Interpret Benefit Indicators

A single biophysical function may generate several socially valuable services. For example, a wetland's ability to filter nutrients leads to improved water quality. Improved water quality can—among other things—improve contact recreation (swimming, wading), reduce drinking water treatment costs, and improve the abundance of species that support active and passive recreation. These different services should be assessed separately and may require different indicators. For this reason, we emphasize that a service-by-service indicator analysis be conducted.

Although NMFS is typically concerned with services that flow from marine and anadromous species populations, habitat analysis should not focus exclusively on these services. The total social benefits of a site arise from a potentially much larger set of service flows. Preserving a habitat for species support (which is the agency's primary trustee function) can generate significant nonmarine species benefits as well. Examples include aesthetic benefits, drinking water improvements, flood prevention, and terrestrial recreation. In a consultation context, these additional benefits are also pertinent to the overall decision.

For each service, the following economic principles should guide the development and interpretation of benefit indicators.

6.3.1 Capacity Indicators

The first step is to develop what can be called capacity indicators. Capacity indicators are based on biophysical data (e.g., data relating to floodplains, land type, watersheds, habitat, aquifers) that speak to an area's ability to provide a function capable of yielding services. These indicators are most akin to conventional ecological indicators. Note though, that these indicators will depict conditions in the landscape beyond the particular parcel in question. Like the more socioeconomic indicators to follow, they will be mappable characteristics of the surrounding landscape or region.

6.3.2 Service Indicators

Service indicators serve as a proxy for the demand for, or benefits of, a particular biophysical function. Service indicators can be based on both socioeconomic and biophysical data. For example, the abundance of a particular land type—a biophysical measure—helps define the scarcity of a function and thus the benefits of a service provided by that function in a particular location. Within this general category of indicators, it is also useful to distinguish four distinct subcategories:

1. *Primary demand indicators.* Ecosystem functions yield beneficial services only when there is demand for services. Demand for services arises when the ecosystem provides an amenity or helps avoid a harm. For an amenity (e.g., aesthetic enjoyment) to be provided, proximity to populations that benefit is a necessary condition for demand.[116] For a harm to be avoided, there has to be the possibility of harm (e.g., flood risks or water contamination) and a population disadvantaged by it.

2. S*carcity and substitutes indicators.* Because scarcity increases the value of a service, indicators of scarcity and the availability of substitutes are important to an analysis of benefits. Scarcity indicators relate to the local prevalence of other similar habitats or biophysical functions. Substitutability indicators measure the abundance of other habitats, land uses, infrastructure, or biophysical functions that can provide similar services to those generated by the habitat in question.

116 The only exception is the existence value of species, where demand does not depend on proximity.

3. *Complementary input indicators.* Some services can be enjoyed only if accompanied by complementary landscape characteristics or infrastructure. This is particularly important for recreation, where access is a crucial determinant of the ability to enjoy the service. The presence of roads or trails or adjacent beaches or parks can be a benefit-enhancing (or necessary) complement to the enjoyment of recreational services.

4. *Demographic composition indicators.* Demand for services may be related to income,[117] racial composition, education levels, or other social characteristics. Also, these characteristics may be important to distributional analysis of ecological impacts (e.g., environmental justice).

6.3.4 Indicators of Risk and Changed Future Conditions

These indicators are similar to those described above, except that they relate to landscape characteristics likely to arise in the future. They can be based on both socioeconomic and biophysical data. Ideally, the analysis of benefits is dynamic. Current landscape conditions speak to the benefits provided by a site today. But the site's value is also a function of the benefits it will generate in the future. Future benefits depend on risks to the functions provided by the site. Analysis of risks addresses questions such as whether or not a habitat's functions will continue to be provided in the future. Medium- and long-term hydrological, biological, and chemical changes can degrade future habitat function. Encroachment by invasive species and sea-level rise are examples of conditions that can threaten or alter a site's existing biophysical function. Demographic conditions can also change over time. Land use, population, and infrastructure changes, such as road construction or planned water diversions, will affect the benefits generated by sites in the future. If information on likely changes is available, it should be included in the benefit analysis.

6.3.5 Service Areas and the Scale of Assessment

GIS analysis permits characterization of the landscape at a variety of scales. The scale of the analysis will in part be determined by the decisionmaking context (if the analysis, for example, is to support regional planning). But from an economic standpoint, the scale of an

[117] See a recent collection of essays on this topic in *Economic Journal* 107, 1997.

assessment should be based on the "service area" associated with a particular habitat. In general, the relevant service area depends on the service in question and differs for individual indicators of that service's benefits. The appropriate area is sometimes determined by physical constraints, and other times by demographic constraints. Boundaries are needed to define the likely users of a service, the areas where access to a service is possible, and the areas where services might be scarce or have substitutes. Physical boundaries vary according to the nature of the biophysical function that gives rise to a particular service. Consider an example ecosystem service: floodwater retention. Here, floodplain and watershed boundaries place natural physical limits on the relevant area used to determine benefits.

Depending on the service being assessed, different scales should be used. For example, a local neighborhood may be the relevant scale for analysis if only a local population benefits. This would be the case, for example, with certain aesthetic benefits, such as the enjoyment of scenic vistas or open spaces that require ownership, access, or adjacency. Watersheds and floodplain boundaries should define the scale in cases where surface water movement is at issue.

When recreational benefits are at issue, the service area is not always easily defined. Knowledge of demographic factors, access via roads and trails, the determination of substitute sites for recreation, and local preferences is needed to determine the relevant service area. These issues are frequently confronted in econometric analyses of recreational benefits.[118] A major methodological issue in any recreational benefits study is the determination of the appropriate "choice set" facing anglers, hunters, hikers, and birders.[119] Choice sets are the set of substitutes for recreation at the site in question. Defining these sets appropriately is important because they describe the relative scarcity of recreational services provided at a particular location.

6.4 Examples

The following examples illustrate the kinds of indicators that can be calculated and the way in which they can be applied to the analysis of benefits.[120] The following examples are

[118] See Smith and Kopp, note 28.

[119] For a good collection of studies that address this issue, see the special issue of *Marine Resource Economics* 14(4), 1999. Also see the National Survey on Recreation and the Environment, 2000.

[120] For more detailed analysis of indicators, including calculated and mapped indicator analyses and application of indicators to a wider range of services, see Boyd and Wainger (2003), note 115.

indicators useful to the characterization of benefits from improved aquatic recreation services, especially contact recreation.

If the particular form of recreation is contact recreation, service flows arise from water quality improvement. Assuming that on-site biophysical analysis shows the habitat capable of filtration or sequestration of nutrients and other pollutants, we first look for capacity indicators that signal an ability to improve water quality.

6.4.1 Illustrative Capacity Indicators

Capacity indicators are based on biophysical data and include both on-site characteristics and characteristics of the larger landscape. On-site assessment of things like habitat and hydrology is relatively conventional. We stress the importance of landscape characteristics and the spatial determinants of capacity. The following types of landscape indicators will signal a site's capacity to provide service flows:

- watershed delineations;
- topography; and
- soil types.

These indicators can help determine whether or not a site is in a hydrological setting to receive polluted runoff. In some cases, a biophysical function is possible only if there is sufficient connectivity or contiguity to complementary ecosystems. Chemical processes associated with nutrient cycling may require a mix of ecosystem types or a minimum surface area exhibiting necessary hydrological and biological characteristics. For example, wetlands filtering nutrients in riparian zones have been shown to have a greater ability to prevent nutrient deposition than wetlands farther inland.[121] For this reason, off-site characteristics, such as those listed below, may be important to the determination of service capacity:

- neighboring land uses and land cover types;
- contiguity indicators (measures of physical connection between land types); and
- proximity to surface water, wetlands, riparian zones.

[121] Lowrence et al., Water Quality Functions of Riparian Forest Buffer Systems in the Chesapeake Bay Watershed, Chesapeake Bay Program Nutrient Subcommittee, Technical Report, EPA 903-R-95-004, 1995; Correll et al., Nutrient Flux in a Landscape: Effects of Coastal Land Use and Terrestrial Community Mosaic on Nutrient Transport to Coastal Waters, *Estuaries* 15(4): 431–42, 1992.

Assuming there is a demonstrated capacity to provide the service—based on the above types of data—the analysis then shifts to indicators of the social benefits of that service.

6.4.2 Illustrative Service Indicators

Service indicators can be both social and biophysical. For the service we are considering here—improved aquatic contact recreation—the focus is on the social benefits of improved water quality. If there is to be demand for improved water quality, a necessary condition is that there be a water quality problem. Indicators of impairment include the following:

- proximity to impaired water segments; and
- swimming advisories.

These indicators relate to the primary demand for the service. The more significant the disamenity, the greater the social benefits of mitigating that disamenity. For example, the more a site is impaired by runoff, the higher the benefits of filtration and sequestration. One indicator of potential runoff impairment is the proportion of a watershed covered by impervious surfaces, a known risk factor for aquatic habitats.[122] Similarly, if the water quality concern is related to nutrients, the prevalence of upslope agriculture will be important:

- percentage of upslope impervious land cover;
- percentage of upslope land cover in cropland and pasture; and
- concentrated animal feeding operations in vicinity.

Because mixing occurs in surface waters, impairments arising from sources throughout the watershed may also be important:

- percentage of impervious land cover in watershed; and
- percentage of agricultural land cover in watershed.

To calculate these kinds of indicators, physical data layers (topology, watershed boundaries) must be integrated with social data layers (land use data on roads, development,

[122] Impervious surfaces create greater runoff volumes and shorter runoff times, leading to more pollution and warmer surface water deposition. See Soil Conservation Service, Urban Hydrology for Small Watersheds, USDA, Technical Release 55, 1975.

agricultural operations). GIS analysis allows for this kind of data integration and the calculation of such indicators.

Also of importance to primary demand is the number of people benefiting from a water quality improvement. Indicators of this include the following:

- local and regional population; and
- levels of tourism.

Most people who travel to aquatic areas for recreation engage in boating and other activities on the water. Accordingly, very local demand conditions (e.g., within a mile of a site) may be of less relevance than regional population numbers and tourism. For other services, such as drinking water improvements, indicators should be more location-specific, since the benefiting population will be in close proximity to the site.[123]

Demand for services is also a function of the abundance or scarcity of the services being provided by the site. Scarcity and substitute indicators are therefore important to assemble as well:

- percentage of same ecosystem type in vicinity or upslope;
- percentage of substitute ecosystem type in vicinity or upslope; and
- percentage of watershed's riparian and coastal areas in wetland.

These kinds of indicators can be calculated at different spatial scales, depending on the biophysical functions in question. For example, if the site in question is a wetland, scarcity indicators will relate to the presence of similar wetlands in the vicinity, subwatershed, or watershed. This kind of calculation can be made using spatial land cover data. Similarly, substitutes for wetlands—including different types of wetlands—can also be assessed. Natural land, such as forests or scrubland, might substitute for certain functions provided by wetlands (though when it comes to water quality improvements, these types of land cover will be only weak substitutes). Again, the presence of the substitutes can be assessed at different spatial scales. Because riparian and coastal wetlands are particularly valuable as both aquatic habitat and contaminant buffers, their scarcity is of particular importance to surface water quality. Since

[123] This is related to the issue of relevant service areas. The concept is discussed in more detail in Section 6.3.5.

surface water quality can be affected by impairments across a fairly broad landscape, analysis at a country or regional level may also be appropriate.

If aquatic recreational benefits are to be considered true benefits, people must be able to actually enjoy such recreation. Complementary inputs—in particular, access to waters—are an important signal of likely benefits:

- proximity to private and public beaches;
- proximity to shoreline parks;
- proximity to boat ramps; and
- proximity to navigable waters.

Forms of accessibility are an important complementary input to recreational enjoyment. Water quality improvements generated in close proximity to such access can have higher recreational value than benefits produced where access is difficult or impossible.

The composition of groups likely to seek out recreational facilities may also be important to an understanding of benefits. The *demographic composition* of the user pool can be assessed:

- age, income, and race of local or tourist population; and
- proximity to schools and community facilities.

In some cases, demand may be related to demographic variables such as age and income. If environmental justice concerns are relevant, income and racial composition are relevant. Some resources provide educational opportunities, in which case age distribution and the presence of schools and community centers can be assessed.

6.4.3 Indicators of Risk and Changed Future Conditions

A site's benefits are also a function of future demand conditions and risks to the site's biophysical functions. Consider first the risks to biophysical function:

- presence of exotic species in vicinity;
- proximity to significant nutrient runoff concentrations;
- elevation; and
- location in floodplain.

Exotic species invasions, in some cases encouraged by the introduction of nutrients, can significantly alter functions, including habitat support, aquifer recharge, and nutrient cycling. The presence of invasive species and significant nutrient or other pollutant concentrations can degrade or destroy functions normally associated with an ecosystem. For example, invasion by woody exotic species can lower water tables and allow saltwater intrusion into estuarine and riparian ecosystems. Significant nutrient and pollutant concentrations can fundamentally alter the chemistry of a site and thereby lead to both habitat and hydrological changes. Low-elevation sites, and particularly low-elevation coastal sites, are more vulnerable to future sea-level rise and other major hydrological changes. In extreme cases, flood events can lead to changes in ecosystem functions.

In the social realm, future economic, demographic, and cultural conditions can alter a site's ecosystem benefits. Changes in land use, or changes in any of the indicators described above, will imply changes in benefit flows. With recreational benefits, predictions of future growth in population or tourism will clearly affect the benefits arising from improved water quality.

Planning studies are inherently speculative. Nevertheless, planning projections, including zoning studies, population projections, and water supply plans, are of direct relevance to an understanding of a site's benefits. Zoning analysis in particular can identify the degree to which existing land use configurations are a good guide to future land use patterns.

That brief set of examples is meant to illustrate the kinds of data available to make ecosystem benefit assessments. Note that the calculation and interpretation of such indicators do not result in a dollar estimate of a site's social value. Instead, they foster a quantification of characteristics that—all else being equal—add to or detract from a site's social benefits. These characteristics can be mapped and aggregated to identify areas that are likely to be service-rich.

It deserves emphasis that this is only a partial list of the kinds of indicators that can be developed. An assessment of recreational or commercial fishing services, for example, could include such indicators as proximity to aquatic preserves and seagrass beds, endangered and threatened species observations, and many others. Flood protection, drinking water, open space, and species existence services demand some similar, but many different, indicators from those described above. As noted earlier, a wide variety of such indicators are calculable using existing data in most areas of the United States.[124]

6.5 The Interpretation of Indicators: Some Important Caveats

Indicators should not be oversold as an assessment tool. In particular, indicators are not an end in themselves, but a way to help organize and structure interagency and stakeholder deliberations over relative social values. Landscape indicators should be viewed as a qualitative evaluation tool that directs deliberations over value, not as a formal means of generating benefit estimates. We envision benefit indicators as a complement to more conventional ecological indicators. In particular, we see them as a way to stimulate and expand thinking on the part of decisionmakers from purely biophysical characteristics to the socioeconomic value of those characteristics. Because indicators are a form of quantitative data, however, they must be used and interpreted with care.

Decisionmakers should resist the temptation to combine or aggregate indicators to arrive at a "summary number." A composite index may appear 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. But a composite number should be resisted for both technical and decision process reasons.

The aggregation of indicators is a highly technical and statistical process with numerous pitfalls. If combined, indicators must be appropriately weighted and scaled.[125] First, consider scaling issues. Unless the relationship between an indicator and the benefit it describes is linear, interpretation of the indicator requires subtlety. In the example provided earlier, we described biophysical and social landscape data and argued that the data can tell us something about the unobservable thing we want to describe—the social benefits created by habitat. Unfortunately, the nature of the correspondence (i.e., the relationship) between an indicator and the thing we care about (benefits) is not always clear. Scaling issues relate to the "shape" of the underlying relationship being described by an indicator. For instance, it is tempting, but usually incorrect, to

124 See Boyd and Wainger, note 115.

125 There is a substantial body of literature in economics relating to the construction of indices, mostly related to the construction of price indices. See Roy Allen, *Index Numbers in Theory and Practice,* MacMillan, 1975.

look at two columns of indicators and think that a site with twice the score creates twice the benefit. This interpretation is natural because, consciously or not, there is a tendency to view indicators as being linearly related to the thing being indicated. When a relationship is linear, twice the score means twice the benefit. But it is probably more common for an indicator to be nonlinearly related to its subject. For example, if one recreational site is found to have double the trail access of another, it is probably more valuable, but not twice as valuable, as a recreational site. Recognizing this limitation is simply recognizing a basic principle of economic logic—the law of diminishing marginal utility.

Second, aggregation requires that individual indicators be weighted based on the indicator's contribution to benefits. For example, construction of a "water quality benefits index" would require merging several indicators (relating to primary demand, substitutes, complements, etc.). Unfortunately, this requires a great deal of information regarding the relationship of the indicators to benefits. This kind of information is usually unavailable. Moreover, an aggregate index achieves institutional validity only if it is constructed with a transparent methodology so that the weights applied in aggregating the indices are fully understood and accepted. Indicators can be statistically evaluated to establish a more scientific justification for their use as proxies for monetized benefit estimates.[126] But the determination of appropriate weights thrusts the decisionmaker back into the world of technical analysis (e.g., of biophysical linkages and effects) and value trade-offs (the relative values people place on different characteristics).

Also, to arrive at a composite score, it is not just the indicators within each service that must be aggregated. The benefits of the individual services must also be assigned weights. Aggregation at this level is particularly inadvisable because this form of weighting substitutes for the explicit decisionmaking expected of agency decisionmakers and the public. Individual indicators allow for an unambiguous ranking of sites only under special circumstances. One circumstance is that the sites are identical in all respects but one. But in general, some sites are 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 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? This is a question to be addressed by interagency deliberations, not through reference to a relatively opaque calculation.

126 See Paul Murtaugh, The Statistical Evaluation of Ecological Indicators, *Ecological Applications* 6(1): 132–39, 1996.

Accordingly, aggregation should be resisted, except perhaps as a way to quickly identify sites likely to be low-benefit or high-benefit outliers. If aggregation is used, the logic of the aggregation should be made clear and open to inspection. Clearly, indicators can be more formally manipulated and applied, but such actions undermine an essential purpose of indicator development, which is to structure and inform interagency and stakeholder deliberations over relative values. Aggregation, by its nature, truncates the information provided to the decisionmaker. It also can introduce a false formality that obscures underlying trade-offs. For these reasons, we do not advocate the use of indicators as a means to generate a number for use in consultations, litigation, or regulatory rulemaking. Rather, indicators should be assembled and carefully discussed in a decisionmaking context that is sensitive to their limitations.

Although we caution against the use of indicators as an end in themselves, an indicator analysis such as those we presented can serve as useful inputs to more formal (and methodologically complex) benefit studies. As we noted earlier, indicators can serve as an input to benefit transfer studies. Also, indicators could be used as an input to studies designed to more explicitly elicit preferences. Ultimately, if the goal is to poll the public for its preferences, there is no substitute for direct survey information designed to measure and represent those preferences. Various statistical methods can be applied to the results of paired preference surveys to arrive at relative service weights or rank orderings of services.[127] One particular method of eliciting preferences across various attributes is the use of conjoint analysis. Conjoint analysis presents a group of subjects with "objects" or "goods" composed of multiple quality attributes.[128] The subjects are then presented with a set of scenarios designed to reveal their preferences for these different attributes. This is done by varying the levels of the attributes and evaluating the

[127] Paul Smith and John Theberge, Evaluating Natural Areas Using Multiple Criteria: Theory and Practice, *Environmental Management* 11(4): 447–60, 1987.

[128] See Ric van Poll, The Perceived Quality of the Residential Environment: A Multi-Attribute Evaluation, Chapter 3, Ph.D. dissertation, University of Groningen, 1997 (available at http://www.ub.rug.nl/eldoc/dis/science/h.f.p.m.van.poll/); Kevin Boyle, Thomas Holmes, Mario Teisl, and Brian Roe, Assessing Public Preferences for Timber Harvesting Using Conjoint Analysis: A Comparison of Response Formats, *American Journal of Agricultural Economics*, forthcoming; F. Reed Johnson, William H. Desvousges, Lisa L. Wood, and Erin E. Fries, Conjoint Analysis of Individual and Aggregate Environmental Preferences, Triangle Economic Research, Working Paper 9502, 1998.

subjects' responses statistically.[129] The kinds of indicators we use in our study could be thought of as indicators of these different quality dimensions. Conjoint analysis of ecosystem benefit indicators could form the basis of a statistically based, indicator-based assessment method. The method is a procedure for quantifying the degree to which specific sites meet each objective (the provision of specific service benefits) and combining the results in a way that is logically consistent with the values of the decisionmakers.[130] Typically, this kind of analysis relies on expert groups and stakeholders to make technical and value judgments.[131]

6.6 An Application: Watersheds and Fill Permitting under Section 404

A minor but growing share of compensatory mitigation required by the Clean Water Act Section 404 permit program is now being provided by commercial mitigation banks and in-lieu-fee mitigation programs. The increased willingness of federal regulators to accept off-site compensation provided by these mitigation specialists, particularly for minor impacts authorized under general permits, has been driven by advances in wetland science and restoration technology, and by recognition of the widespread failure of "project-specific" mitigation implemented by permit recipients.[132]

This movement of mitigation practices toward off-site compensation has advanced the no-net-loss regulatory goal. But the process may be moved further toward the NMFS mission if requested compensation actions are supported by carefully developed habitat priorities, informed by benefit hotspot analysis.

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129 Paul Green and V. Srinivasan, Conjoint Analysis in Marketing: New Developments with Implications for Research and Practice, *Journal of Marketing* 54: 3–19, defining conjoint analysis as "any decompositional method that estimates the structure of a consumer's preferences, given his or her overall evaluation of a set of alternatives (objects) that are prespecified in terms of levels of different attributes."

130 See Ralph Keeney and Howard Raiffa, *Decisions with Multiple Objectives: Preferences and Value Tradeoffs,* John Wiley, 1976; Thomas Saaty, *The Analytical Hierarchy Process: Planning, Priority Setting, Resource Allocation,* McGraw-Hill, 1980; and M. Granger Morgan, Baruch Fischoff, Lester Lave, and Paul Fischbeck, A Proposal for Ranking Risk within Federal Agencies, in *Comparing Environmental Risks: Tools for Setting Government Priorities,* Resources for the Future, 1996, 111–47.

131 Miley Merkhofer and Ralph Keeney, A Multiattribute Utility Analysis of Alternative Sites for the Disposal of Nuclear Waste, *Risk Analysis* 7: 173–94, 1987.

132 National Research Council**,** Securing Compensatory Mitigation under Section 404 of the Clean Water Act, National Academy Press, 2001.

6.6.1 The Logic of Threshold Effects Supports No Net Loss

Threshold effects can best be addressed through the consultation process if NMFS emphasizes the replacement of lost services and if replacement can be achieved with a high degree of certainty. However, threshold levels (along with many other parameters) are likely to be unknown and therefore uncertain. As we noted in Section 5, the literature in exhaustible resource economics suggests that uncertainty in the stock size implies a slower rate of extraction than would occur under certainty. Also, we showed that recognition of spatial interdependence is important when thinking about decisions made under uncertainty. Given the characteristics of habitats, it is likely that a precautionary approach is justified where the changes in habitat quality are lower with uncertainty than without. This said, the conclusion is not that no habitat is ever to be altered. Under a precautionary approach, the mitigation currency would be in biophysical production units so that the biophysical productive capacity of the habitat is maintained, even though habitat alterations are permitted. Third, the spatial pattern of alteration and mitigation will also matter. In fact, there is a possibility that a habitat mitigation decision that reproduces biophysical production in one area may actually increase the overall level of production.

6.6.2 A Few Principles for Watershed-Oriented Mitigation

The location, mix, and quality of existing wetland habitats within many watersheds may be the accidental result of uncoordinated historical land and water development. From an area-wide perspective, these remaining wetlands may not be the best for securing desired habitat services. But which wetlands in the landscape are most important, and which should be the focus of compensation practice? The specific answer to these questions varies by location.

From a whole-watershed perspective, priority wetland functions often may be best advanced by off-site and out-of-kind compensation. This does not mean that wetlands lost to fill permits always should be replaced off-site and out-of-kind. Rather, the decision about mitigation location and type could follow from a reasoning process that encompasses the habitat services and their values. This may mean, in the case of wetlands, that the services might be segmented for purposes of guiding compensation location and design. Consider a hypothetical fill permit in which wetland regulators are primarily concerned about the expected loss of local hydrological function if the habitat function is secured by moving the compensation wetlands elsewhere in the landscape. In this case, the compensation decision would appropriately consider any state or local stormwater management controls (e.g., ponds) that would be required at the development project. To the extent that such controls effectively replicate the hydrological function provided by the filled wetland, the habitat mitigation required for the fill permit could be located and designed according to habitat requirements.

6.6.3 Setting Priorities for Watershed-Oriented Compensation

Some people agree in principle with the area-wide perspective but view it as unworkable until formal watershed plans have been developed. Highly technical and data-intensive watershed plans that identify priority wetland types and specific parcels for restoration and protection are costly to develop and can engender more controversy than agreement. However, plans may consist of principles that are derived from "watershed thinking," and the principles might then be used in case-by-case consultations on permits.

Watershed thinking is a less structured approach to priority setting—one that relies on the informed, best professional judgment of regulators and other stakeholders in watershed management. In fact, in-lieu-fee programs are already using such an approach across the country, where program administrators and Army Corps regulatory staff jointly select the types and locations of mitigation actions that serve their understanding of watershed priorities for wetland restoration and protection. This is a low-cost process for guiding decisions on compensation actions that would best serve watershed priorities in consideration of what was or would be lost by fill permits. This type of watershed priority setting, augmented with advisory contributions by staff of other federal agencies, nongovernmental conservation entities, and the local scientific community, might be sufficient to implement a watershed-oriented compensation strategy.

However, NMFS capabilities to direct and influence this process might be enhanced by investment in the modeling approaches described in previous sections. Therefore, additional development of bioeconomic analysis tools should be an NMFS priority. Developing large models might not be cost-effective for each permitting decision, but model development is cost-effective if taken on a much larger scale and then used to support a sequence of future consultations. Ideally, given the current scientific understanding of the fundamental ecological and economic process, the models and the parameter levels would be constantly refined as new information is gleaned.

The indicators approach would be wedded to the models developed, with whatever level of sophistication is possible given budgets and available human resources. For example, conventional biodiversity hotspot analyses identify high-priority conservation areas based on biophysical criteria, such as the density of threatened species in a given area. Benefit indicators can be used to construct benefit hotspot maps to serve a similar function. Benefit indicators can also be used to identify zones that are more or less likely to be ecosystem service-rich. In turn, these maps can be used by regulators and planners to quickly assess the relative desirability of mitigation activities, restoration, and habitat preservation, as well as the consequences of landscape alterations due to development, infrastructure construction, and other activities that can potentially degrade ecological functions. By their very nature, hotspot analyses convey spatial priorities.

The method involves “zoning” the landscape on the basis of the benefit indicators described earlier. Indicators can be used to identify areas where particular services can be eliminated from consideration (e.g., because they do not meet conditions necessary for the existence of the service). In other words, the capacity to provide a service can first be mapped. Then, service indicators can be mapped to identify benefit-rich areas. The presence of overlapping benefit zones can in principle quickly identify areas where landscape conditions reinforce the existence and scale of service benefits. Potential risks to functions and services can also be mapped, informing the analysis regarding likely future social benefits. Note, however, that all the caveats associated with indicators (Section 6.5, above) must be applied to benefit hotpot mapping as well. Benefit hotspot maps are not an end in themselves and should not be presented as finished products with all simplifying assumptions suppressed. Rather, such hotspot maps should be constructed by stakeholders and decisionmakers themselves as a means to collectively assess benefits and the biophysical and social characteristics of the landscape.

Benefit hotspot analysis is a potentially effective way to convey basic valuation principles—thereby fostering interaction between economists, ecologists, and stakeholder groups involved in ecosystem decisionmaking.

7. Conclusion

Habitats and the services they provide are part of our nation’s portfolio of natural capital assets. Like many of the components of this portfolio, habitats and the services they generate are not traded in markets. This makes it difficult to assess the value of their services and considerably complicates regulators’ investment and disinvestment decisions, especially when the alternative use has easily measurable values. NMFS staff members are confronted with these difficulties daily when they attempt to fulfill their trustee responsibilities in habitat consultations.

In this report, we discuss in detail bioeconomic and ecosystem indicator approaches to habitat value assessment. Although the approaches are discussed independently, it is not hard to imagine that multiple tools, such as bioeconomic modeling, ecosystem indicator approaches, and other benefit assessment tools could be used simultaneously across different regions or within the same region on different aspects of one consultation. Whatever method is chosen for a particular problem, it is important that the analysis be carried out carefully and with a level of rigor that is greater than or equal to the magnitude of the decision. In other words, we do not advocate spending thousands of dollars analyzing a decision that would cause only hundreds of dollars in damages.

Bioeconomic analysis, which has been applied to fishery management for more than 50 years, is a more structural approach, where the specifics of the structure depend on the setting. The analysis is built upon mathematical relationships that depict the nature of the ecosystem production function and the services it provides. All models are stylized, and therefore modelers need to make important decisions regarding scope and scale. The resources needed to carry out the analysis are a function of these decisions. With all the caveats considered, bioeconomic models are very useful in understanding the incremental loss in habitat cover (such as seagrass beds), incorporating biophysical currencies in mitigation decisions, and undertaking risk assessments of different habitat alteration plans.

The other technique we highlight is an ecosystem indicators approach, which is relatively new and builds upon the habitat scoring methods employed by ecologists. The approach capitalizes on advances in data presentation made available by GIS software and the growing availability of data in electronic form. This is an important feature that can bring down implementation costs. By overlaying multiple maps of the relevant ecological and socioeconomic indicators, researchers can identify potential "hotspots." Researchers could develop criteria, maybe through an expert consultation process, on what level of hotspot classification would trigger opposition to habitat alteration permits.

In addition to discussing indicator and bioeconomic approaches, we provide guidance on why and how economic analysis and arguments should be applied to decisions surrounding essential fish habitat designations. In what follows, we summarize the major points regarding the role of economic analysis in habitat valuation and our recommendations for the scale and scope of investment in habitat valuation research.

Use of economic arguments in habitat alteration decisions

- Qualitatively or quantitatively, the socioeconomic benefits and costs of decisions that affect the condition of aquatic habitat are critical for rigorous decisionmaking and fulfilling the trustee responsibilities of NMFS.

- Habitat valuation rests on the biophysical assessments that characterize ecosystem structure and functions that generate the services valued by people; the services created by ecological characteristics determine social value and not the function per se.
- Threshold effects and the uncertainty surrounding the tipping point can lead to taking a go-slow approach, but this is not equivalent to restricting all habitat alteration projects.
- Mitigation currencies need to take into account biophysical characteristics of the habitat, including position in the surrounding landscape or seascape, rather than simply a measure of coverage, such as acres of seagrass beds.

Recommendations for investment in economic research

- NMFS should seek partnerships with other agencies to invest in multiple-year interdisciplinary efforts on a regional scale to understand the temporal and spatial nature of ecosystem production functions and the services that are provided.
- NMFS should seek partnerships with other agencies to invest in a portfolio of research methodologies, such as bioeconomic modeling and ecosystem indicator approaches, to further the development of widely acceptable techniques that can be applied in the consultation process.
- NMFS should seek partnerships with other agencies to invest in research that compares and contrasts the costs, practicality, and decisionmaking utility of different benefit assessment tools, including nonmarket valuation, by applying them to a common problem and geographic location.
- In the near term, NMFS should target investment in benefit assessment research to decisionmaking contexts where there is demand for and receptivity to the results of such research.