Economic Analysis of Protection of Essential Fish Habitat in Alaskan Fisheries: An Analysis of Research Needs

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Abstract

This report identifies the types of information necessary to evaluate the economic impacts of Essential Fish Habitat (EFH) protection and the gaps in the existing knowledge and research methodologies necessary for this evaluation. A discussion of the basic elements of a benefit-cost analysis or a cost-effectiveness analysis for essential fish habitat protective measures (EFHPM) is used to identify the types of benefits, both use and nonuse benefits, which must be quantified and monetized if they are to be used in a formal benefit-cost analysis. There is also a discussion of the potential costs of EFHPM and how they might be measured. This discussion identified large gaps in knowledge and in valuation methodologies. On the basis of the relative importance of these knowledge gaps and the feasibility of filling them with new research, a number of potentially productive areas of research are identified. This research would improve our ability to more optimally balance habitat protection against costly constraints on fishing. The research areas identified include random utility modeling of location choice, bioeconomic modeling to estimate fishery benefits of habitat protection, non-market valuation of habitat protection programs perhaps paired with research on benefit transfer techniques, and market based systems to cost-effectively achieve specified habitat protection objectives.

Contents

Introduction

The objectives of this report are to identify the types of information necessary to evaluate the economic impacts of Essential Fish Habitat (EFH) protection, identify gaps in the knowledge and research methodologies necessary for this evaluation, and discuss research topics relating to the economic evaluation of EFH protection. The report makes recommendations of projects that are likely to be useful and feasible given informational and budget constraints.

The report also reviews the economic literature that pertains to EFH and provides a discussion of the reasons for designating EFH and the issues associated with evaluating the economic impacts of EFH protection. Although there are a variety of non-fishing actions that may affect habitat, such as nutrient releases, recreation, shipping, introduction of exotic species and wetland conversion, the focus of this report is on EFH protection measures targeted at modifying fishing practices. This includes spatial controls on fishing, gear restrictions, and other measures that regulate the activities of fishers with the intent of providing protection to EFH.

The rest of the report is organized into four major sections. The first section contains a general discussion of EFH policy, legislation and regulations. This section discusses the motivation and intent of the legislation and the general issues associated with economic evaluation of fishery management measures that might be implemented to protect EFH, henceforth referred to as EFH protection measures (EFHPM). Here it is considered that EFHPM encompasses a broad range of possible measures that may be employed to protect fishery habitat and the ecosystem of which fisheries are a part. These include but are not limited to marine reserves, defined as areas where no extractive activities are allowed, marine protected areas (MPAs) that put spatial restrictions on certain activities such as fishing for particular species or using particular gears, and general restrictions on use of particular fishing practices and gears. They may also include economic incentive or property rights based strategies designed to afford protection to EFH. The second section provides a discussion of the general approaches and methodologies that might be applied to evaluate the economic impacts of EFHPM. These include benefit-cost analysis, cost-effectiveness analysis and economic impact analysis. We also include a brief discussion of issues associated with risk and uncertainty. This section also identifies and categorizes the potential benefits and costs of EFHPM and briefly discusses the methodologies appropriate for evaluating them. Section 4 continues with a discussion of specific methodologies for evaluating certain categories of costs and benefits of EFHPM and reviews some applications of these methodologies. Section 4 also proposes a number of potentially useful research efforts related to economic evaluation of EFHPM. The final section provides a summary of findings and recommendations. An appendix reviewing literature on marine reserves and MPAs is included at the end of the report. While not all of this literature is directly applicable to EFHPM, it is likely that EFHPM will rely heavily on these types of measures and this literature is instructive since the arguments for and against EFHPM are likely to bring forth many of the same arguments brought to bear in the debate over marine reserves and MPAs.

Background on Essential Fish Habitat

In 1996, the U.S. Congress added new habitat conservation provisions to the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). These changes came in response to concerns that habitat loss was threatening the viability of many of the Nation's fisheries. The findings section of the MSFCMA states "One of the greatest long-term threats to the viability of commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats. Habitat considerations should receive increased attention for the conservation and management of fishery resources in the United States."

The MSFCMA directs the National Marine Fisheries Service (NMFS) and the eight regional fishery management councils, under authority of the Secretary of Commerce to identify and describe EFH in each fishery management plan, minimize to the extent practical the adverse effects of fishing on EFH, and identify other actions to encourage the conservation and enhancement of EFH. Section 303(a)(7) of the MSFMCA requires that fishery management councils, to the extent practical, shall minimize adverse effects on essential fish habitat (EFH) caused by fishing. In addition, 50 CFR, part 600.815 (a)(3) requires councils to prevent, mitigate, or minimize adverse effects from fishing, to the extent practical, if there is any evidence indicating that a particular fishing practice is having an identifiable adverse effect on EFH. Section 600.815 (a)(4) permits a range of management options to protect EFH including fishing equipment restrictions, time and area closures, and harvest limits.

Essential Fish Habitat is defined as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity (16 U.S.C. 1802(10)). "Waters" include all aquatic areas and associated physical, chemical, and biological properties that are used by fish. Areas historically important to fish may also be included. "Substrate" includes sediment, hard bottom, structures underlying the waters, and associated biological communities. The NMFS defines "necessary" to mean the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem.

The EFH provisions of the MSFCMA address a relatively broad concept of habitat centered on its role in supporting a healthy marine ecosystem. Rosenberg et al. (2000) discuss the intent of EFH legislation as it relates to ecosystem health. They note that EFH completes the foundation for an ecosystem approach to marine fisheries that is vital to the efforts of NOAA/NMFS to improve the health of fish stocks, ensure the long-term sustainability of fisheries and fishing communities, and develop a more comprehensive approach to fisheries management. This interrelationship between EFH and marine ecosystems is discussed in some detail in the final rule on EFH promulgated by NMFS (2002) which states:

> Councils should strive to understand the ecological roles (e.g., prey, competitors, trophic links within food webs, nutrient transfer between ecosystems, etc.) played by managed species within their ecosystems. They should protect, conserve, and enhance adequate quantities of EFH to support a fish population that is capable of fulfilling all of those other contributions that the managed species makes to maintaining a healthy ecosystem as well as supporting a sustainable fishery......Ecological relationships among species and

between the species and their habitat require, where possible, that an ecosystem approach be used in determining the EFH of a managed species or species assemblage. The extent of the EFH should be based on the judgment of the Secretary and the appropriate Council(s) regarding the quantity and quality of habitat that is necessary to maintain a sustainable fishery and the managed species' contribution to a healthy ecosystem……Healthy ecosystem means an ecosystem where ecological productive capacity is maintained, diversity of the flora and fauna is preserved, and the ecosystem retains the ability to regulate itself. Such an ecosystem should be similar to comparable, undisturbed ecosystems with regard to standing crop, productivity, nutrient dynamics, trophic structure, species richness, stability, resilience, contamination levels, and the frequency of diseased organisms.

Rosenberg et al. (2000) argue that to achieve the goals of the congressional EFH mandate, the regional fishery management councils and NMFS must consider a greater diversity of fishery-management tools, including MPAs, marine zoning, and marine reserves. They note, however, that NMFS and the councils should learn how to minimize threats without imposing undue hardships on recreational and commercial fisheries (e.g., by modifying fishing practices or gears rather than banning fishing). They state that "the legal mandates and the agency's precautionary approach to resource management mean we must proceed without delay with reasonable conservation measures for fish habitat."

Rosenberg et al. (2000) note, however, that "for habitat conservation and enhancement to contribute fully to fishery management, we must continue to sharpen our policymaking and to improve the scientific basis for decisions. It is not sufficient simply to provide general protection measures because adverse impacts to our nation's fisheries may occur." The language of the MSFCMA and the EFH rules promulgated by NMFS suggest that EFH protection is required to the extent that the lack of this protection puts the sustainability of the fishery at risk, but does not explicitly require managers to attempt to design EFH protections to optimize either fishery productivity or fishery profits. Councils are instructed to "consider the nature and extent of the adverse effect on EFH and the long and short-term *costs* and *benefits* of potential management measures to EFH, associated fisheries, and the nation, consistent with National Standard 7" (50 CFR §600.815). Councils are not, however, required to perform a formal benefit-cost analysis.

Nevertheless, the protection of EFH must be considered within the context of the rest of the MSFCMA, which requires consistency with the several national standards. From an economic standpoint, the most important standard is National Standard 1 which states that "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry." "Optimum", with respect to the yield from a fishery, is defined in the law as the amount of fish which:

 (A) will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;

(B) is prescribed as such on the basis of the maximum sustainable yield from the fishery, as reduced by any relevant economic, social, or ecological factor; and

(C) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery.

In order to be in compliance with the MSFCMA, actions taken to protect EFH must consider the corresponding benefits and costs of these actions. Unfortunately, the determination of the net economic benefits from conserving and protecting EFH is likely to be extremely complicated. In the first place there may be considerable uncertainty as to what effect protective actions will actually have on habitat and, in the case of habitat rehabilitation, how long recovery will take. Even if the physical impacts on habitat are perfectly understood, valuing those impacts remains difficult. It is not generally adequate or appropriate to directly value the features of habitat. Although protection of specific components of the habitat (e.g., marine plants and animals such as corals and sponges) may generate non-use values which might be estimated with non-market valuation techniques, these benefits are arguably not the primary focus of the EFH provisions of the MSFCMA[1]. The benefits from habitat protection in the Alaska EEZ are likely to be primarily indirect benefits resulting from increases in the productivity of fisheries, reduction of variability in production, or reduction of risk of fishery depletion or collapse.

To the extent that EFH protections eventually result in increased catches (or prevent decreases in catches) they may increase both consumer surplus and producer surplus. The effect on consumer surplus will depend on both changes in landings and the elasticity of demand, and may depend on changes in product mix that could results from EFHPM. The effects on producer surplus are even less clear since EFHPM are likely to affect both revenues and costs. Assuming relatively elastic demand, changes in revenues might be closely correlated with changes in total catches. However, factors other than total catch may influence prices either positively or negatively. For example, changes in the average sizes of fish might occur, and variation in price-based on fish size is common. For fisheries where roe or milt contributes a significant proportion of value, EFH measures that affect the proportion of catch taken around spawning will clearly have an impact on total revenues. Even if EFH protections increase revenues in the long run, short-run impacts on revenues are quite likely to be negative. In fisheries constrained by limits on Total Allowable Catches (TAC), the EFH protections may have no impact on total catches unless TACs are reduced, but for other fisheries where TACs are not constraining, EFH protections may lead to reduced catches if they increase harvest costs. There are a number of species in the North Pacific for which TACs are not constraining, so the possibility that EFH protections might reduce catches is real. To the extent that EFH protections close areas to fishing or restrict certain fishing practices and gears, they are likely to increase harvest costs. These increases may be offset in the long run if EFH increases the biomass of fish stocks thereby increasing fish availability. This is more likely for fisheries that had been overfished or where habitat damage was having substantial impacts on fishery productivity. But even where increases in productivity occur they will not necessarily compensate for increased costs.

The discussion above makes clear that the economic impacts of measures designed to protect EFH are likely to be complex and highly uncertain. A full understanding of these impacts requires knowledge of the effects of fishing practices and gears on marine habitat as well as an understanding of the degree to which and the speed at which habitat will recover from previous damage. The linkages between habitat quality and fishery productivity and stability must also be

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[1] Other legislation, the Endangered Species Act in particular, may require protection of particular animals or plants but generally does not require a balancing of the costs and benefits of doing so.

known. Next, changes in fishery productivity as well as qualitative changes in fish catches must be translated into changes in revenues and consumer surplus. The effects of regulatory measures on harvest costs must be also factored in. Finally, it is important to consider the relative timing of cost and benefit streams. Benefits from EFH protection are likely to be realized only after several years while increases in costs and decreases in catches may be immediate. Thus the discount rate that is used to compare costs or benefits incurred at different times together with predictions of when those costs and benefits will be incurred can affect the benefit-cost analysis dramatically. Resolving uncertainty in the time path of outcomes may be as important as understanding equilibrium[2] outcomes.

Economic Analysis of EFH Protection

King et al. (2002) provide a good general discussion of issues associated with economic analysis of marine habitat degradation. They discuss the three analytical tools most widely used by economists: 1) Benefit-Cost Analysis – a tool that provides a quantitative method for comparing the economic benefits and costs associated with a specific project.; 2) Cost-effectiveness Analysis – a technique used to rank the project alternatives that achieve a similar or predetermined outcome; and 3) Economic Impact Analysis – an approach that allows one to assess how the direct impact of expenditures of a project will ripple through the local economy and affect regional employment and income. In the discussion below we focus primarily on various aspects of first two of these general methodologies and on the tools and analytical techniques required to determine the net economic effect of EFHPM on the value of fisheries. EFH policies are likely to have varying economic impacts on regions and communities that will be of interest to policymakers, but the standard tools used for economic impact analysis should be applicable to evaluation of EFH without substantial modification once the direct impact on fisheries is understood[3]. Cost-benefit analysis and cost-effectiveness analyses of EFH policies, in contrast, are likely to require new purpose-built analytical tools such as models that can predict the biological and economic effects of spatial management measures (e.g., area closures and spatial gear restrictions). It is these tools that we focus on in this report.

Benefit-cost Analysis

An economic evaluation of regulatory actions to protect EFH should go beyond the identification of various benefits and costs that may result from their implementation. As Farrow (1996) notes in his discussion of benefit-cost analysis for marine reserves, "what is important is the process that draws all impacts together in a summary analysis and the delivery of a bottom line result." To do this, benefits and costs must generally be quantified and expressed in monetary terms so that they can be compared. We can not compare, in a benefit-cost framework, "biological benefits" to "economic costs" without a common metric such as money.

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[2] Not least because marine fisheries and the ecosystems that support them are constantly subject to a variety of exogenous shocks so that they are never really in equilibrium.
[3] That is not to say that the models (e.g,. IMPLAN) used for this analysis could not be improved to give more accurate predictions of impacts on communities, but that work is ongoing and not specifically related to EFH protection.

In general, a benefit-cost analysis should attempt to determine whether the proposed management measure provides greater net benefits than some alternative policy. The basis for comparison might be the status quo or might include alternative policies designed to achieve similar objectives. Quantification and valuation of costs and benefits of EFHPM such as area closures are likely to be difficult and imprecise due to uncertainty about the direct and indirect impacts of a closure and of what would happen in the absence of one. However, the difficulty of this task should not be used as an excuse to avoid it completely.

A complete economic evaluation of an EFHPM should also explore the probable distribution of benefits and costs. Even if a particular measure does have positive net benefits overall, it is likely to disadvantage some individuals and groups. Compensation of losers in some form may be necessary or advisable to ensure the acceptance and success of the measure. Measures perceived as unfairly imposed are likely to require higher compliance costs and may generate political action that will block their implementation or lead to their failure or repeal.

EFH protections are likely to generate costs immediately while benefits may not be realized for some time. There is an extensive literature on marine reserves and MPAs (see Appendix) that supports this contention. The delay in benefits versus costs should be explicitly considered when undertaking a benefit-cost analysis. The standard procedure is to compare the present value of all cost and benefit streams, which requires discounting future costs and benefits by the appropriate discount rate. There is often disagreement about what the appropriate discount rate should be, and there is no standard rate used for public policy evaluation. Harte et al. (2000) state that "no valid argument exists for zero or very low discount rates" and recommend real discount rates of between 4% and 15% as appropriate for applied fishery analyses. Their report notes that "the predominant consensus in the literature supports the use of discount rates near inflation-adjusted market interest rates (e.g., rates between 4% and 12%)." A discount rate of 7% has been used in evaluating Federal fisheries policies in some cases (NEFMC 1996).

It is not sufficient to simply identify or even quantify the types of costs and benefits that may result from implementation of EFH protections, but it is useful to categorize these cost and benefits. This will help to facilitate and organise the appropriate scientific and economic analysis for a specific proposal. It is likely that any benefit-cost analysis of EFHPM will have to make use of a number of separate valuations of different impacts, positive and negative, and then compare these.[4] These valuations may require data that is not currently available and development of new valuation methodologies tailored for application to Alaskan EFHPM.

Although EFH is geared toward ensuring the productivity and sustainability of fisheries, the debate over the merits of EFH protections is often polarized in terms of extractive and non-extractive uses. It is generally argued that appropriately designed measures such as marine reserves or MPAs will generate net benefits to non-extractive users,[5] while it is the perception of many extractive users of the marine environment (e.g., commercial and recreational fishermen) that these measures will result in a net loss of benefits to them. The two groups need not be mutually exclusive, but categorizing costs and benefits in this way facilitates economic analysis.

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[4] Economic impacts, even expectations of long-run impacts, may be revealed in values of fishing rights for fisheries with marketable fishing rights. However, even in these cases, benefits to non-extractive users will not be revealed.
[5] This in no way suggests that a poorly designed reserve might not result in a net loss of non-extractive value.

Non-extractive Benefits and Costs -- Non-extractive benefits associated with EFH may include both use values and non-use values. Non-use values are benefits derived without actually experiencing something physically or visually (e.g., someone may derive value in knowing there are wilderness areas even though they have no desire to visit them). Protecting marine habitat may generate non-use values if there are features of these areas including plants or animals that might be lost or disturbed in the absence of protections. The primary method used to estimate non-use values is contingent valuation which uses survey techniques to elicit the value a sample of individuals places on the resource being evaluated. These values are then extrapolated to a broader group to determine the welfare effects of changes in the quantity or quality of the resource. There is considerable debate regarding the accuracy of these methods and which methods provide better estimates, but these valuations have been used in public policy and in the courts. A Blue Ribbon panel of economics and survey research experts chaired by Kenneth Arrow and Robert Solow concluded that, if properly conducted under strict guidelines, contingent valuation "can convey useful and reliable information that can produce estimates reliable enough to be the starting point of a judicial process of damage assessment" (Arrow et al. 1993). It is important to keep in mind that the non- use benefits associated with EFHPM may relate more to changing the probabilities of extinction of known or unknown species or subspecies rather than losses of particular numbers of plants and animals.

EFHPM may also generate non-extractive use values associated with a viewing or recreational experience of some sort (e.g., the value of a diving experience inside a marine reserve or a whale watching cruise). Although there may be no observable market transactions for amenity values that reveal their net value, a variety of methods can be used to estimate these benefits including contingent valuation and revealed preference techniques that infer value from observing choices and actions rather than relying on respondents to provide their own valuation. The regulatory impact review (RIR) suggests that non-extractive use benefits are not likely to be significantly affected by the EFHPM being considered by the North Pacific Fishery Management Council (NPFMC), so costly quantification of these impacts may not be justified.

There may also be costs to non-extractive users. Sanchirico et al. (2002) note that non-extractive activities such as diving and snorkeling or anchoring of pleasure boats may cause damage to habitat reducing some of the non use benefits a marine reserve was meant to provide. The development associated with tourism attracted by a reserve can lead to environmental degradation that may reverse initial conservation benefits. A marine reserve that excludes fishing from a large area may also result in the disappearance of traditional fishing communities which local residents, tourists and the general public may perceive as a loss of value. Again it appears unlikely that these costs will be significant for the EFHPM being considered for Alaskan fisheries.

Extractive Benefits and Costs -- EFHPM could result in at least short-term losses to both commercial and recreational fishers displaced from their traditional fishing areas. However, EFH protection need not exclude all fishing as long as the impact of a particular activity on habitat is minimal. Consequently, they may have little effect on recreational fisheries. To the extent that EFHPM do displace recreational fisheries, they may result in a loss of value to recreational fishers. It is also possible that EFHPM could impact recreational fishers negatively if they displace commercial fishing effort into recreational fishing areas. The RIR or EFH does not

explicitly address this issue. Presumably this is because impacts of proposed EFHPM on recreational fisheries are not expected to be large.

The EFHPM are quite likely to have significant negative economic impacts on commercial fisheries, at least in the short-run. This is an increasing concern if area closures in both inshore and offshore areas are imposed. An extensive review of empirical studies of marine reserves noted that "there are no well-documented examples where marine fishery sanctuaries have been shown to provide and maintain net economic benefits for previously existing fisheries" (Ward et al. 2001). Marine fishery sanctuaries are not the only EFHPM available, but they are one of the primary tools being considered.

The benefits from commercial fishing can be divided into two primary categories, consumer surplus and producer surplus. Consumer surplus is the aggregate difference between what consumers would have been willing to pay for the seafood they consumed and the price they actually paid. Essentially this is the net benefit that consumers received from purchasing seafood rather than spending that money on the next best thing. We are not aware of any studies on the level of consumer surplus associated with Alaskan fisheries, and it is difficult to say how substantial it may be. The RIR was unable to quantify the effects of EFHPM on consumer surplus. It is probably reasonable to say that there may be some decrease in consumer surplus if catches decrease. Quantification of demand and consumer surplus for Alaskan fisheries would be useful for a variety of public policy analyses, but is probably of secondary importance for benefit-cost analysis of EFH measures. Before a more accurate understanding of the effects of EFH protection on catches is achieved, there is limited value in determining how those changes would affect consumer surplus.

Of more direct concern is the impact of EFHPM on producer surplus from commercial fisheries. Producer surplus generated by fisheries is the economic profit attributable to exploitation of the fisheries. It can be defined quite simply as the difference between revenues and the opportunity cost of inputs used in generating those revenues. This includes the costs of inputs such as fuel and labor as well as the opportunity cost of capital. The EFHPM could impact the producer surplus from commercial fisheries in a variety of ways. The most obvious impact is the effect on total catch and associated revenues. It is important to keep in mind that the percentage decrease in producer surplus is likely to be greater than the decrease in revenues. The slimmer the margin is between revenues and costs the greater will be the ratio of the drop in producer surplus relative to the drop in revenues.

As we discuss in the Appendix, the modeling literature on marine reserves and fishery area closures suggests that they may very well decrease catches in well managed fisheries and are unlikely to increase them. In general, a decrease (increase) in catch will result in a decrease (increase) in revenues and producer surplus. The EFHPM may, however, increase or preserve the productivity of fisheries and consequently have a more positive impact on catches than the literature on marine reserves suggests (see Appendix).

Reductions in revenues due to reduced catches may be partly offset if decreased supply leads to increased prices. This of course depends on the nature of demand just as consumer surplus does. Prices may also be affected by changes in the quality of product, and these changes might be

either positive or negative. For example, closures of spawning areas may reduce harvests of roe or milt which could greatly decrease the value of a given quantity of fish for some species. An area closure that prevents targeting of larger size fish and results in a smaller average individual size might lead to a decline in average price. The opposite could also occur in the long run as a result of large fish spilling over from a marine reserve or may result directly from closing areas with high concentrations of juveniles. However, if there are substantial benefits in selection of larger fish, this might be achieved more efficiently in other ways such as changes in codend mesh size (Holland 2003).

The EFHPM are likely to have their greatest impacts on the cost side of producer surplus. Since they constrain choices, they can be expected to increase the cost of taking a given amount of fish. In the short run, this is an obvious outcome. Fishermen are presumably fishing in the location using the fishing practices they find most profitable. Location preference could be because the fish caught there are more valuable (e.g., they are larger or the proximity to market allows them to be delivered in better condition). Alternatively, the location preference might be due to safety reasons or to the expectation of higher catch rates there than at alternative sites. In the latter case, reducing catch rates results in a higher cost per unit of catch. Any of these reasons suggest a loss in producer surplus following exclusion from the preferred fishing areas. The same argument can be applied to restrictions on particular fishing methods or gears.

The costs of area closures, particularly areas closed to all fishing, may actually compound over time. If the same level of total catch is maintained after an area is closed the displacement of effort from the closed area to other areas may result in crowding and gear conflicts as well as local depletion. Over time, if the closure is effective at protecting a resident fish population, the density of fish inside it will build up relative to the density outside. Obviously, catch rates would be higher and unit costs lower, if fishing were concentrated where fish are concentrated but this will not be possible if those areas are closed to fishing.

A final category of costs to consider is management and enforcements costs. Monitoring and enforcement of EFHPM may significantly increase costs both for the management agencies and for fishing vessels. For example, some of the alternatives being considered for providing increased protection to EFH in the Alaskan fisheries are expected to require expanded use of electronic vessel monitoring systems (VMS) which will require both additional expenditures on government infrastructure and personnel, and expenditures by fishers on equipment and operational costs of the VMS (Tetra Tech et al. 2003).

Both the costs and benefits resulting from EFHPM are in fact streams of costs and benefits over time. The cost and benefits streams are unlikely to be constant or simultaneous. For example, a marine reserve is likely to increase fishing costs immediately while any gains resulting from the emigration of large fish from a reserve are likely to be several years down the road. Non-extractive benefits such a fish viewing will also only be fully realized after many years. An appropriate economic evaluation of and EFHPM must take the timing of costs and benefits into account by transforming cost and benefits streams into present value terms so that they can be compared.

Cost-effectiveness Analysis

It may be easier to quantify the costs of EFHPM than the benefits. If actions to protect EFH are taken without a clear understanding of benefits, cost-effectiveness analysis rather than benefit-cost analysis may be the appropriate evaluative approach. Cost-effectiveness analysis simply attempts to find the policy that will achieve a desired outcome at the lowest cost. For example if a decision has been made to close 20% of a certain type of habitat to trawling but there is little knowledge of the relative benefits of closing one part versus another, the appropriate question is which 20% can be closed at the least cost.

Cost-effectiveness analysis is likely to be less onerous than a full benefit-cost analysis primarily because only certain impacts need be valued, and the focus is likely to be on more certain and shorter term impacts. The specific types of costs that might be considered in such an analysis were discussed above so we do not go through them again here. As with benefit-cost analysis it will be appropriate to discount costs that fall in the future relative to immediate ones.

EFH, Risk and the Precautionary Principle

Uncertainty and risk associated with irreversible events is a dominant issue in policy arguments by natural scientists advocating policies for marine management and habitat protection. Auster (2001) advocates a hierarchical approach to marine resource management. He suggests making use of MPAs and marine reserves when limited information is available on impacts of human activities and potential impacts are severe and difficult to reverse. The hierarchy he proposes is preventive approach – corrective approach – precautionary approach – precautionary principle. Thresholds based on the relative size of the footprint (percentage of total area fished) trigger precautionary closures unless enough information is available for preventive and corrective measures. Preventive measures are to be used when spatial patterns of resource use, habitat, and biological diversity are understood. They include spatially explicit gear restrictions or effort limitations. Corrective measures adjust boundaries of preventive measures when new information becomes available (this is not really mitigation but adaptation). Precautionary approaches are implemented when we know little about linkages between habitat and exploited populations particularly for sensitive or long-lived species. This approach calls for marine reserves targeted at specific habitat types or species. Finally the precautionary principle is used when "we know the least about the system and the potential for irreversible damage is high or very high such that it is not possible to predict that actions will not irreversibly damage habitats or threaten the extinction of species." Here a network of marine reserves of representative habitats might be called for. Auster (2001) suggests that this approach (with the precautionary approach or principle in the form of area closures) be invoked when thresholds based on the fishery footprint are crossed. He argues that this provides incentives to industry to help collect more detailed spatially explicit information. Of course it could also motivate strategic reporting.

Many economists also advocate conservative, precautionary approaches to deal with risk and uncertainty. As King et al. (2002) explain, "evaluating risk is an inherent component of economic analysis and is useful even when evidence regarding biological impacts is weak. The unique role of risk in economic analysis has some potentially powerful applications in habitat protection because it can be used to demonstrate that over and above any direct project costs that are being

measured; there are additional costs due to the fact that a project may also be expected to increase society's risks." King et al. (2002), in this comment, are referring to benefit-cost analysis of development projects that may have negative impacts on EFH, but the rationale applies to continuation of fishing practices that may be impacting fishery habitat and consequently posing some risk to the future productivity of the fishery or some other valuable components of the marine ecosystem. If the habitat delivers a stream of benefits with a present value of $100,000 and there is a 20% chance that those benefits will be lost if a particular fishing practice is allowed to continue, the expected value of the gross benefits from restricting the fishing practice would be $20,000. However if the public is risk-averse regarding that loss, it might require more compensation to allow this practice to continue and leave welfare unchanged. These types of considerations are particularly cogent when the potential impacts are irreversible or where recovery might take a very long time.

The regulatory impact review prepared for the EFH Environmental Impact Statement (EIS)(Tetra Tech et al. 2003) provides some discussion of this issue motivating the need for precautionary management on the basis of what they term "opportunity reservation value."

> Opportunity reservation value is defined here to mean a societal value distinct from traditional option value, the latter being an individually held form of future use value. In this instance, the value being defined may be regarded as a collective hedge against irreversible loss of some highly valuable goods or service, flowing from EFH, that has not yet been recognized. That is, ecosystems such as those that comprise EFH are enormously complex and, as yet, not well understood. EFH may provide some future consumptive use benefit that is not currently used or even identified. For example, minimizing the adverse effects of fishing practices on EFH may preserve a species of plant or animal or an ecological process that, in the future, may prove to have irreplaceable, tangible value to the world's population. Such examples already exist. Specifically, marine sponges have yielded valuable medicinal compounds for use in anti-malaria and HIV infection suppression drugs. At present, it is not known whether or how many of these potentially valuable species or functions exist and, therefore, it is not possible to place a monetary value on their future use. Retention of the option to exploit these public assets in the future clearly has some reservation value, and argues for a precautionary management approach (i.e., erring on the side of preserving these assets).

While the concept of a risk premium associated with potential but uncertain losses of marine habitat or its components is intuitively appealing, determining its existence and magnitude is likely to be problematic. The appropriate risk premium can depend on risk aversion which may be positive or negative depending on the nature of gains and losses. One source of this risk premium might be option value, a concept introduced by Weisbrod (1964). This is essentially equivalent to the concept of "opportunity reservation value" discussed above. Freeman (1993) explains that originally option value was seen to arise when an individual was uncertain as to whether he would demand a good in the future and was faced with uncertainty about its availability. "If option price is defined as the maximum sum the individual would be willing to pay to preserve the option to visit a site before his own demand uncertainty is resolved, then the excess of option price over expected consumer surplus can be called option value." Bishop (1982) suggested an alternative concept of option value where the individual's demand is certain, but

supply is not. Bishop (1982) showed that option value is positive for a project that eliminates uncertainty in supply. However, Freeman (1993) explains that formal analysis of this concept uncovered subtleties and complications that showed that option value could be either positive or negative even for risk averse individuals. Freeman (1993) is quite skeptical of the concept of option value and states that:

> "we can now see that option value is really just the algebraic difference between the expected values of two different points on a willingness-to-pay locus. Specifically, it is the algebraic difference between the expected value of consumer surplus and the state-independent willingness to pay (option price). Since these two points represent alternative ways of measuring the same welfare change, the difference between their expected values cannot be a separate component of value. Furthermore, option value cannot be measured separately; it can only be calculated if we have enough information on preferences to calculate both option price and expected surplus. And finally, as I have shown in this chapter, neither of these points on the willingness-to-pay locus has any particular claim as a superior welfare measure. I think it is time to expunge option value from the list of possible benefits associated with environmental protection."

Another source of a risk premium referred to as quasi-option value was proposed by Arrow and Fisher (1974). Quasi-option value describes the welfare gain from delaying a decision when there is uncertainty about the payoffs of alternative choices that may be resolved over time, and when at least one of those choices involves an irreversible commitment of resources. Arrow and Fisher (1974) showed that quasi-option value is not dependent on risk aversion. Freeman (1993) notes however "that it is not difficult to imagine situations where the relevant information to guide future decisions can be gained only be undertaking now at least some development" in which case quasi-option value of development, rather than forgoing development, could be positive. Freeman goes on to argue:

> "whether quasi-option value exists or whether it is positive or negative for preservation depends on the nature of uncertainty, the opportunities for gaining information, and the structure of the decision problem. Quasi-option value is not a component of the values individuals attach to resource changes. Even if individuals' utility functions were known, quasi-option value could not be estimated separately and added into a benefit-cost calculation. Quasi-option value is a benefit of adopting better decision-making procedures. Its magnitude can only be revealed by comparing two strategies where one of the strategies involves optimal sequential decision making to take advantage of information obtained by delaying irreversible resource commitments. The decision maker who knows how to use an optimal sequential decision-making strategy has no reason to calculate quasi-option value. The calculation would be redundant since the best decision is already known."

Another important concept relating to risk and uncertainty is the Safe Minimum Standard (SMS) (Ciriacy-Wantrup 1952). In the case of an activity with some positive probability of causing an irreversible impact of unknown value it may not be possible to determine the optimal level of the activity to balance against the expected value of the loss. An alternative is to set a safe minimum standard believed sufficient to ensure the loss is not incurred and not allow that standard to be

breached unless the costs of doing so are catastrophic. Ciriacy-Wantrup and Phillips (1970, p. 28) explain "here the objective is not to maximize a definite quantitative net gain but to choose premium payments and losses in such a way that maximum possible losses are minimized." This concept is intuitively appealing to many, but it has been found to be somewhat flawed from a logical/philosophical perspective. Ready and Bishop (1991) show with a simple and clear exposition that a two-person minimax game against nature cannot provide a firm theoretical foundation for the SMS approach of endangered species preservation. The problem is laid out as follows. Preservation of a species by choosing SMS involves forgoing development benefits. There is an unknown probability of a disease in the future which might be cured by a species lost if development proceeds. In an insurance game, the occurrence of the disease is uncertain but the cure is definite if the species still exists. In the lottery game, there is only a finite probability that the disease will be cured if development doesn't proceed and the species is preserved. The maximum loss in the insurance game occurs if development proceeds and the disease happens, so SMS is the dominant strategy to minimize the maximum loss. But for the lottery game the maximum loss is not developing and not having the cure for the disease anyway when it comes. The dominant strategy is develop and hope. To the extent that the lottery game is a better description of reality, it brings the logic of the SMS into question.

Despite its flaws, the SMS approach is more or less that adopted by the Endangered Species Act (ESA). To the extent that particular fishing practices or other marine activities create a significant risk of extinction of a particular species the ESA would be the pertinent legislation, but reliance on a safe minimum standard approach is less defensible for general EFH protections oriented at ecosystem protection or protection of biodiversity where there are not specific species or unique objects at risk.

The concept of a precautionary approach to marine resource management is much in vogue, but operationalizing the concept is problematic. An analysis of the arguments over risk premiums, option value and safe minimum standards suggests that a precautionary approach to EFH management is not necessarily welfare increasing. Actions to protect habitat that are costly and have highly uncertain payoffs should themselves be considered cautiously.

Economic Research Topics Relevant for Evaluation of EFHPM in the Federal Fisheries off Alaska

The discussions of the EFH EIS and RIR and the economic analysis of EFH protection above highlight a number of areas where data and research methodologies are inadequate. Many of these areas, such as estimates of operating costs of fishing vessels and improvements to economic impact analysis tools such as IMPLAN, are needed for a wide variety of regulatory analyses but need not be tailored for use in economic analysis of EFHPM. The Alaska Fisheries Science Center, the NMFS Alaska Regional Office, and the NPFMC have active programs in these areas. There are a number of research topics specifically related to economic analysis of EFHPM for which we believe new or increased research efforts are likely to be worthwhile. These include random utility modeling of location choice, bioeconomic modeling to estimate fishery benefits of habitat protection, non-market valuation of habitat protection programs, and market-based systems to cost-effectively achieve specified habitat protection objectives. The following sections discuss each of these topics in turn. The essential elements of the research and some of the

relevant literature on theory and previous applications are reviewed. It is outside the scope of this paper to provide a detailed research proposal for each of these areas, but potentially useful research is discussed in general terms.

Estimating Costs of EFH Protection

Analysis of the costs of EFHPM is likely to focus on the value of activities displaced or restricted. A first order analysis might simply look at the revenues or preferably the net profits associated with the activities that are being excluded. This approach has been applied for the RIR for the Alaska EFH EIS (Tetra Tech 2003) and was also applied by NMFS and New England Fishery Management Council staff to estimate the costs of EFH closures in the Northeast (NMFS, NEFSC 2003). As the authors of the Northeast EFH analysis note, this approach may tend to overestimate costs: "All other things being equal, this "no displacement" assumption is likely to result in an upper bound estimate of revenue impact since vessels would attempt to maintain fishing incomes by redirecting effort to alternative fishing locations."

In most cases, it is unrealistic to assume that restriction of particular activities will not result in some redirection of that activity unless there is a simultaneous reduction in total allowable catch or effort. Restrictions on fishing location choices or practices can be expected to initially reduce the profitability of fishers, but the net losses will depend on the value of the alternative choices still available. Net losses can be estimated by comparing the revenues or preferably the profits that that displaced fishers can expect to make from their next most valuable option relative to the restricted option. However determining which option would be the most profitable for an individual and what the change in profits would be is not straightforward. Even if the profitability of other areas is observable (i.e., information on catch rates and revenues in other areas is available), it may not be fair to assume that a displaced fisher can expect to generate profits equal to observed average profits in other areas. There may be learning costs associated with moving to a new area and fishers may have insufficient information to choose the most profitable alternatives. Also, if a substantial amount of effort is displaced, it may be unrealistic to assume that it can be moved to other areas without negative effects on catch rates in those areas.

Therefore it may be important to use a more sophisticated approach to modeling effort displacement and estimating the economic losses associated with that displacement. To understand longer run results it may be necessary to integrate fleet dynamics models into dynamic bioeconomic models. There is a growing literature on modeling location choice and entry-exit decisions in commercial fisheries (e.g., Bockstael and Opaluch 1983, Curtis and Hicks 2000, Dupont 1993, Eales and Wilen 1986, Hicks et al. 2004, Holland and Sutinen 1999 and 2000, Mistiaen and Strand 2000, Smith 2000, Smith and Wilen 2003). These modeling techniques can be used to predict where effort will be displaced which may be useful in modeling expected impacts on catches and costs. The models can also be used to directly estimate the welfare losses to fisheries associated with eliminating specific fishing options.

These analyses use discrete choice random utility models to model fishing decisions. Individuals are assumed to choose among several discrete alternatives (e.g., fisheries and/or areas). We observe which alternative they choose and assume that the alternative chosen is the one which generates the

highest expected utility for them. The probability that a particular alternative k is chosen can be expressed as:

$$P(k) = P\left[U_k \geq \max_{k' \in C} U_{k'}, \forall k \in C, k \neq k'\right] \qquad , \qquad (1)$$

where U represents an indirect utility function for choice k for a specific individual. Assuming U can be divided into a systematic component, V, and a random component, ε, and that utility is a function of alternative specific characteristics, q, and individual specific characteristics, s that may differ across individuals n, the discrete choice probability can be written:

$$P(k) = P\left[V_k(q_k, s_n) + \varepsilon(q_k, s_n) \geq V_{k'}(q_{k'}, s_n) + \varepsilon(q_{k'}, s_n) \forall k \in C_i, k \neq k'\right] \qquad (2)$$

Equation 2 from observed choices leads to predicted choice probabilities at the individual and aggregate level. Some behavioral assumptions are required. Typically, choices are assumed to be independent and mutually exclusive such that the ratio of the probabilities of choosing any one alternative over another are not affected by the attributes of other alternatives in the choice set[6]. In most applications to date, the random components of the indirect utility functions, ε, are assumed to be independently and identically distributed (iid) with the Gumbel (type I extreme value) distribution. Given these assumptions the parameters of the utility function can be estimated with the multinomial logit (MNL) model. Combining the arguments q_k and s_n of the utility function into a vector $\mathbf{x_k}$ with parameter vector Beta, $\boldsymbol{\beta}$, the MNL choice probabilities can be expressed as:

$$P(k) = \frac{\exp V_k(q_k, s_n)}{\sum_{k \in C} \exp V_k(q_k, s_n)} = \frac{\exp \beta' x_k}{\sum_{k \in C} \exp \beta' x_k} \qquad (3)$$

Bockstael and Opaluch (1983) appear to have been the first to use this approach to model fishing decisions in a commercial fishery. They use a random utility formulation based on individual profit and wealth maximization to model individual fisher's decisions to enter or exit the groundfish fishery or participate in other fisheries. The decision was assumed to occur on an annual basis. They found that, "while fishermen demonstrate a bias toward remaining in the same fishery, sufficient incentives, in terms of changes in expected returns and risk, are shown to elicit a response."

Dupont (1993) extended the methodology of Opaluch and Bockstael to a model of location choice in the British Columbia salmon fishery and found that fishers switch areas in response to higher expected profits and wealth but, as other models have shown, demonstrate bias toward remaining in the same area.

Eales and Wilen (1986) used the individual utility framework to model daily location choice decisions of vessels in the California pink shrimp fishery. They utilize a nested logit model that predicts first the choice between large zones and then the choice of areas within a zone. The nested logit model is a generalization of the multinomial logit model that accommodates cases where the

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[6] Increased computing power has also made other estimation techniques feasible such as multinomial probit and random parameter logit models. These models can be less restrictive in their assumptions.

random component for groups of alternatives may be correlated, a violation of the iid assumption of the multinomial logit model. In this case, it was assumed that there are unobserved characteristics common to areas within a given zone. As with the studies discussed above, Eales and Wilen's results indicate that information derived from the prior day's catches (by the fleet, not by individual fisher) were a good proxy for individual expectations about relative profitability of various fishing areas the following day.

Holland and Sutinen (1999 and 2000) extend this discrete choice modeling approach to accommodate fishery and location choice on a trip-by-trip basis in a multispecies, multi-area fishery. The resulting individual level behavioral models are used to predict aggregate temporal and spatial fishing effort distribution for the overall trawler fleet. Effort distribution predictions can be updated as the characteristics and size of the fleet and the biological and regulatory status of the fishery change. Like Eales and Wilen, they utilize a nested model structure assuming a hierarchy of decisions; first a fishery and zone choice (e.g., groundfish on Georges Bank or squid in southern New England), and then an area choice within the larger zone. Fishers are assumed to make a decision between one of 41 discrete fishery-area alternatives prior to leaving port. Each fishing alternative has an expected utility based on a vector of explanatory variables and an error term.

Holland and Sutinen specify a two-level model where individuals first choose a fishery/zone combination, j, at the highest level of the model and an area, k, within the larger zone at the lowest level. The choice probability of the multinomial logit model is now redefined as the conditional probability of area k in fishery/zone j, k|j :

$$P(k|j) = \frac{\exp \beta' x_{k|j}}{\sum_{k \in Cj} \exp \beta' x_{k|j}} = \frac{\exp \beta' x_{k|j}}{\exp J_j} \qquad , \tag{4}$$

where $J_j = Log \sum_{k \in C_j} \exp \beta' x_{k|j}$

and is known as the inclusive value for fishery/zone J and represents the composite utility of the choices within branch j.

The probability of choosing a particular fishery/zone j is:

$$P(j) = \frac{\exp(\gamma' z_j + \sigma_j J_j)}{\sum_{j \in C} \exp(\gamma' z_j \sigma_j J_j)} \qquad , \tag{5}$$

where σ is the inclusive value parameter, z_j is a vector of variables that differ across fishery and zone, but not across area alternatives, and γ_j the estimated parameter vector or the variable vector z_j. The unconditional probability of choosing a given alternative is the product of the probability of choosing the fishery/zone, and the conditional probability of choosing the area given the zone and fishery choice, $P(j) \times P(k|j)$. Commercially available software is used to estimate the parameters of this model simultaneously using full information maximum likelihood methods.

Smith and Wilen (2003) used daily logbook and landings data to estimate a location choice model for divers in a commercial sea urchin fishery in California. They utilized a nested logit specification to model first a decision to dive or not dive and a location choice decision contingent on the decision to dive that day. The utility of not diving captures the utility of leisure, work opportunities outside of fishing, and the value of avoiding exposure to unsafe diving conditions. Theirs is the first analysis to model discrete participation and location choices jointly in a commercial fishery.

Curtis and Hicks (2000) and Hicks, Kirkley and Strand (2004) use location choice models to estimate welfare losses associated with area closures. This is accomplished via numerical methods, by calculating equivalent variation (EV) for a closure of certain areas in the choice set

$$V^0(W^0 + E(\pi)^0, Var(\pi)^0) = V^1(W^0 + E(\pi)^0 + EV, Var(\pi)^0) \qquad , \qquad (6)$$

where the maximum expected utility for a choice occasion, $V^i(W^0 + E(\pi)^0, Var(\pi)^0)$, is equal to $E\{max[U(W^0 + E(\pi_j) + EV, Var(\pi_j)) + \varepsilon_j, \forall j \in S^i]\}$. Notice that EMAX is taken over the original set of fishing alternatives S to yield V0 and taken over only those sites remaining open S1 to yield V1. Therefore, EV is the amount of money necessary after the closure of some sites to hold utility at a level as if the closures never happened. The expectations operator of the maximum expected utility function is the researcher's expectation taken over εa. Hicks et al. (2004) use an iterative approach to calculate EV since some of the explanatory variables in the utility function (namely aggregate effort in each area) change when effort is redistributed after a closure: "In order to model the welfare effects and choice probabilities, we conduct an iterative analysis. We first, calculate each vessel's choice probabilities for all sites that remain open after the EFH designation. Using these choice probabilities (which are greater than zero for only those sites remaining open after the EFH designation) to allocate effort across fishing grounds, we recalculate the variables fleet and fleet-squared to update the previous thirty day's activity, so that the choice model properly accounts for the closure effects. This process is repeated until choice probabilities stabilize."

The AFSC is currently funding work on location choice modeling of vessels fishing in the walleye pollock fishery in the Bering Sea and Western Gulf of Alaska. Layton et al. (2003) have developed a variant of the random utility modeling approaches described above. They formulate the location choice model directly in terms of expected profits and allow actual observed catch to differ from expected catch. This leads to a tractable discrete/continuous model where the discrete portion relates to the choice of fishing location, and the actual catch constitutes the continuous portion. This work is in progress and results will be supplied directly to the AFSC so there is not a more extensive discussion in this report. The initial results from this work appear promising however, and, in principle, the methodology could be applied to other fleets and areas that may be affected by EFHPM. Layton, Hanie and Huppert do not use this model to estimate welfare effects, but measures of equivalent variation could be presumably derived from the model in a similar manner to Curtis and Hicks (2000) and Hicks et al. (2004).

A potential limitation of the specific approach taken by Layton at al. (2003) relates to the data used to support the analysis. Their analysis is applied to a fleet of pollock vessels with complete

fishery observer coverage. The vessels most likely to be impacted by EFHPM are bottom trawlers fishing cod, flounders, rockfish, etc. Many of these vessels are only partially covered by observers. It may be necessary to adapt the approach to use of logbook or fish ticket data which may require changing the temporal and/or spatial resolution of the model. This is not a trivial matter as it affects both the reliability of results and the degree to which specific area closures can be evaluated.

Whether or not subsequent modeling of other fleets requires changes in data sources and resolution of the choices modeled, the amount of effort required should not be underestimated. Location choice applications tend to require a great deal of time for data preparation. Designing the specific of the model such as the appropriate resolution of the choice modeled and the formulation of the explanatory variables can also be quite time consuming. It will generally be advisable to interview at least some of the decision makers (presumably skippers) to develop a better understanding of how they make their location choice decisions (Holland and Sutinen 2000).

Estimating Fishery Benefits of EFH Protection

Evaluating the economic benefits of EFH protection is a challenging task on which limited progress has been made to date. There have been a large number of empirical studies that have documented the ability of fishery area closures, marine reserves and MPAs to protect fish populations within their boundaries (see Appendix), but much less focus on their effects on habitat. There have also been a number of recent studies on the effects of fishing on marine habitat (NRC 2002). These studies have documented significant damage to habitat from certain fishing gears. From these studies of area closures and impacts of fishing, we can conclude that measures such as gear restrictions, MPAs, and marine reserves are likely to be effective at protecting marine habitat to the extent that threats from non-fishery sources are not important. Protection is likely to be extended to physical features and sessile organisms as well as some species that exhibit limited movement and are afforded protection by a closure. However, success at linking these biological outcomes to changes in fishery productivity and consequent extractive benefits has been quite limited to date.

Estimating the fishery benefits of habitat protection has and will likely continue to rely heavily on modeling. A modeling approach is necessary because it is generally not possible to directly observe the effects of habitat protection on the productivity of the fishery. It is sometimes possible to observe how individual fish utilize habitat to escape predators or for reproductive purposes, but relating changes in habitat to changes in productivity of the fish stock must rely on a statistical analysis of changes in fishery biomass and catches or on some type of analytical model or numerical simulation. Statistical approaches must isolate the impacts of habitat protection from those of other factors such as regulatory, environmental, and market changes that affect natural productivity or human activity. Because of the amount of natural variation in fishery productivity, the lack of accuracy of observations of fish stock size and growth, and the long time frames over which habitat impacts are likely to be realized, statistical approaches to evaluating benefits of habitat protection have met with very limited success to date.

An alternative is to incorporate mechanisms for habitat quality to influence natural mortality, growth, or recruitment into analytical models or numerical simulations. These models allow us to test the effects on fishery productivity of specific assumptions about linkages between habitat quality, fish mortality, and growth. In the end, the reliability of the results is dependent on the veracity of the assumptions, but sensitivity of the results to uncertainty about the assumptions can be measured. Knowler (2002) provides a good review of bioeconomic models that incorporate environmental influences on fisheries. He reviews both empirical, statistically based applications, and theoretical models. Several of these papers are discussed below.

Analytical Models -- Knowler begins by reviewing the basic static equilibrium fishery model (i.e., Gordon-Schaeffer) and discusses some of the papers that have extended that model to incorporate environmental effects. He then develops the dynamic fishery model with optimal control theory. He apparently finds little in the literature in the way of dynamic analytical fishery models, but reviews a general optimal control model of resource use by Freeman (1991) that allows the growth of the resource stock to be influenced by a variable representing environmental quality. This model does not provide a great deal of enlightenment other than to show that optimal time paths of resource use are dependent on environmental quality.

Strand et al.(unpublished) also pursue an analytical modeling approach based on optimal control theory. The authors develop a dynamic analytical model of a fishery where natural mortality and growth may be functions of habitat which is in turn affected negatively by fishing. This somewhat general biological model is incorporated in a less general dynamic economic model (notably no interaction, either biological or economic, between areas is specified – it would have complicated the first order conditions from the Hamiltonian considerably). They use this model simply to point out that optimal habitat protection will be a function of the effects it has on harvesting costs and on stock productivity. In the concluding section the authors note that operationalizing this type of model for an empirical analysis is not possible at this point. Their statement regards EFH in the Mid-Atlantic, but is equally true for almost all fisheries. "It is unlikely that this type of model will be useful for the decisions of the Mid-Atlantic Fishery Management Council anytime in the near future. Although some of the economic requirements of using it for certain species are available, the biological effects of the gear in different areas are largely unknown. Although the model helps one think through the problem and organize how the factors interact with one another, it needs the biological information to be implemented."

Statistical Approaches Based on Reduced Form Models -- Quantitative empirical studies by economists on the value EFH to fisheries can be divided into two groups. Early studies, and the majority of studies done, are statistical estimations of reduced form fishery models. Barbier (2000) reviews a number of these econometrically estimated bioeconomic models that include habitat fishery linkages. Although different applications are based on varying assumptions about the way in which habitat affects fishery productivity, the equations estimated are similar and are subject to similar problems.

The reduced form specified by Lynne et al (1981) is:

$$H_t = \beta_0 + \beta_1 E_t ln(M_{t-1}) + \beta_2 E^2 ln(M_{t-1}) + \beta_3 H_{t-1} + \varepsilon_t \quad , \tag{7}$$

where H is harvest, E is observed effort and M is a measure of habitat quantity. Despite the inclusion of lagged harvest and the fact that current capacity is represented by lagged marsh area, the basis for the model is essentially static. The reduced form assumes that the fishery is in bioeconomic equilibrium and does not include the lagged catch term. The lagged harvest term is an ad hoc addition to account for the fact that "stock adjustment may require several periods." Lynne et al. (1981) use this model to estimate the effect of salt marsh losses on the gross value of a blue crab fishery. Carrying capacity is assumed to be a function of acreage of marsh (specifically the natural log of salt marsh area). The authors find a statistically significant relationship between catch and marsh and, in a separate model, find that marsh interacted with effort shows a nonindependent linkage between effort and habitat effects. Kahn and Kemp (1985) use a similar methodology to estimate marginal damage function from destroying submerged aquatic vegetation (SAV). Although the coefficient on SAV is only marginally significant, the authors go on to derive a marginal damage function that takes into account losses to both consumer and producer surplus.

The reduced form of an explicit intertemporal formulation where the habitat affects the growth rate of the stock over time is specified by Barbier and Strand (1998) as:

$$H = qEK(M) - \frac{q^2}{r}E^2 = \beta_1 * E * Ln(Mlag) + \beta_2 * E^2 \quad , \tag{8}$$

This formulation assumes an open access fishery with zero profits (e.g., total costs are equal gross revenue). The fishery is assumed to always be in bioeconomic equilibrium although that equilibrium is shifting over time. Barbier and Strand (1998) use this model to evaluate the role of mangroves on gross revenues from a shrimp fishery in Campeche Mexico. They find a rather small effect of mangrove loss on the fishery, and that over exploitation was the main cause of welfare losses. Barbier et al. (2002) use a similar approach to estimates the welfare losses to inshore artisanal fisheries in Thailand from conversion of mangrove swamps for shrimp farming. They get significant results and estimate losses of consumer surplus between $11,000 and $408,000 per acre. The range is due to varying assumptions about elasticity of demand about which they have no information. They ignore producer surplus which is deemed to be zero as this is a classic open access fishery. A primary contribution of the paper is to illustrate how the institutional setting of open access and demand elasticity combine to affect the level of welfare losses associated with mangrove conversion.

Other applications using variants of this general approach include Loomis (1988), which looks at effects of logging on value of salmon and steelhead fishing, and Swallow (1990) which looks at an indirect effect on a shrimp fishery of wetland conversion that reduces salinity in an estuary and consequently the productivity of the shrimp fishery. Swallow shows that ignoring effects on the fishery results in too fast and too much development. Interestingly, both of these studies give relatively low values for stopping the activity that is harming the fishery.

A primary difficulty with the approaches described above, aside from having a good measure of habitat loss to use in the estimation, is that the change in habitat is likely to be unidirectional and is likely to be correlated with other variables that might be the cause of reduced catches. These might be natural environmental fluctuations or manmade. Increasing fishing effort or maintaining

too high a level of catch in the face of declining fish stocks is an obvious one. The correlation of these explanatory variables can make it impossible to identify the extent to which individual variable influenced the outcomes, particularly if the fish stock has been on a one-way trip downward.

Dynamic Bioeconomic Simulation Models -- Knowler concludes, and we concur that, although analytical models and simple reduced form bioeconomic models provide some insights, it may be profitable to begin to incorporate more realistic assumptions about biological, ecological, and economic processes into more complex dynamic models. Presumably this will require numerical simulation modeling as these models quickly become intractable to analytical manipulation.

A number of simulation modeling applications, both by economists and natural scientists, that evaluate the effects of habitat protection on fishery productivity are reviewed. Most of these applications (e.g., Mangel 2000, Sanchirico 2004) are essentially theoretical models that sidestep issues associated with estimating the functional relationships between habitat protection, habitat quality and growth or reproduction of the fish stock. They posit a stylized fishery model, where fishery growth is a function of biomass as well as habitat, and explore the impacts of closing some areas to fishing, thereby protecting habitat. Others, such as Lindholm et al. (2001), are based on actual fisheries but are essentially a numerical story telling device incorporating arbitrary assumptions and parameters about habitat functionality. Lindholm assumes that natural mortality of juvenile cod is lower in areas with structural complexity in substrate and that complexity and reduced mortality is only maintained if the area is closed to fishing. Lindholm et al. (2001) further assume density dependent mortality and movement. They go on to show, as is rather obvious from their assumptions, that if you close areas with complex substrate you will reduce juvenile mortality and increase recruitment. They argue that areas must be large to reduce the effect of density dependent mortality and movement, and that the closures should be in the complex habitat as opposed to other areas -- for obvious reasons. The authors infer that the increased recruitment will increase fishery harvests, but they don't model the fishery itself or account for the possibility that adult fish may tend to aggregate in closed areas where they can not be caught.

These theoretical and quasi-theoretical simulation approaches can be useful in clarifying the implications of some of the assumptions about how habitat may affect fishery productivity, but they tend to show that specific assumptions about functional forms and the magnitude of parameters (such as the effect of habitat quality on growth or recruitment) matter a great deal. Examples of studies that have attempted to estimate these functional relationships and incorporate them into empirically based simulation models are few. Knowler et al. (2003) is probably the best example of this approach. They statistically estimate the effect of habitat on the stock-recruitment function and introduce this into a dynamic simulation model of the fishery. The authors use a cross-sectional time series approach to isolate the effect of habitat loss on recruitment to salmon fisheries. This is possible because there are multiple discrete spawning populations and time series of returns and data on habitat loss. They estimated a linear relationship between the instantaneous average annual change in recruitment for each stream ( ) and the "habitat concerns index" (*HCI*). These results are incorporated into a modified Beverton-Holt stock-recruitment function to model the production of coho smolts (*SM*) as a function of spawner escapement (X_{t-2} - h_{t-2}) and habitat quality (*Q*). They then incorporate these results into a dynamic bioeconomic

model to calculate the benefits (in terms of the value that increased salmon catches add to producer and consumer surplus). They use this model to estimate the marginal value of protecting salmon habitat which can be compared to the cost of doing so. They also discuss the additional value of ecosystem services that habitat protection might provide.

Another example is Rodwell et al. (2002) which models habitat quality effects of area closures and effort management in a reef fishery in Kenya. The study uses a dynamic model but looks at equilibrium conditions. Though it is not completely clear, they appear to try to fit the model and its parameters with data from a fishery near Mombassa, Kenya. The main novel aspect of this model is to posit that both natural mortality and movement are related to habitat quality. Habitat quality is assumed to improve over time in the reserve and reduce natural mortality as well as, in some versions of the model, to influence migration and transfer of recruits between the reserve and open area. The study does not compare different reserves sizes; rather it looks at the effect of alternative assumptions about habitat effects with various exploitation rates while maintaining the current reserve size. In their sensitivity analysis they do compare results to those with no closure. They find that the increases in equilibrium catch and biomass with the habitat effect on natural mortality are greater when there is higher exploitation. If habitat quality also increases relative movement into the reserve, the increases in catch from habitat effects are less. The results are not very general and the model specification is quite complex and ad hoc. Unfortunately, habitat parameters are really just guesses (though some sensitivity analysis is done). They fall short of explaining the conditions under which habitat effects of a closure result in increased catches. The results from the sensitivity analysis show maximum equilibrium catches around 30% higher with the reserve than without it but requiring an exploitation rate of 0.4 instead of 0.3. This occurs when there is no habitat effect on movement – it only affects natural mortality. Very slight increases still occur with a moderate effect of habitat on movement. Otherwise highest catches are achieved with an exploitation rate of 0.3 and no closure.

Useful Modeling Research for the Alaska EEZ -- The reduced form statistical approaches to evaluating EFHPM do not appear to be particularly useful or applicable to EFHPM for Alaska fisheries for several reasons. It is not clear that there is a good measurable proxy for habitat quantity or quality and observed productivity of fisheries appears to be more influenced by climate cycles than by habitat effects. Furthermore, total catches have been dictated by regulations and are probably not closely related either to surplus production or to a bioeconomic open access equilibrium. The basic methodology used by Knowler (2002) would seem to be appropriate for offshore fishery analysis (that is modeling marginal increase in optimal catch and profit as a function of increased recruitment which is in turn a function of habitat). Unfortunately, applying a similar methodology to offshore fisheries such as pollock, cod or rockfish looks daunting if not impossible. Estimating the habitat parameter for recruitment for the offshore fish is likely to be problematic. The cross-sectional approach used for Pacific salmon was innovative, but is probably not feasible for offshore fish stocks since stock and recruitment estimates are for large stocks. Straight time series analysis is likely to be confounded by non-habitat factors (e.g., climate cycles and such). Furthermore there isn't good time series (or even current) data on benthic habitat quality. On top of all that, the stock recruitment functions for the stocks in question are likely to be very noisy and not well estimated. Still it may be useful to build these types of models to test the ramifications of reasonable assumptions about functional relationships between habitat protection and fishery productivity and stability. Even if they do not generate

believable estimates of benefits, these models can be useful for identifying critical parameters that justify increased empirical research.

Nonfishery Benefits and Benefit Transfer

While the primary motivation for EFH protection is presumably to increase productivity of fisheries and/or ensure their sustainability, there may also be other benefits that accrue as a result of conserving non-commercial species and habitat independent of their role in supporting commercial or recreational fisheries. Estimating the economic value of conserving habitat for its own sake (e.g., existence value) requires valuation using stated preference techniques (e.g., contingent valuation studies). These valuation approaches have been widely used and are finding increasing acceptance in policymaking and the courts. There is a large and expanding literature on these methods which is not practical to review here. However, as the RIR for the NPFMC EFH EIS (Tetra Tech et al. 2003) notes, there are no existing valuation studies that provide estimates of the existence value of EFH. Furthermore any valuation of the impacts of EFHPM on existence value would need to value the marginal gain or loss in habitat which may have little to do with the total existence value of a certain type or area of habitat. It may well be that the value of marginal losses of habitat relates to changes in probability of losing particular species or risks of causing an ecosystem to collapse. These increased risks may be small for initial habitat losses but might increase as more areas are lost.

Designing appropriate valuation studies will be a challenging task. Due to the lack of information about the relationship between policies and outcomes, these studies may need to focus on valuation of habitat protection programs and their objectives rather than valuing habitat itself or even damages to it. This was the approach taken in a contingent valuation of a protection program for Steller sea lions in Alaska by Giraud et al. (2002). They asked respondents to value a protection program for sea lions that would increase costs for the fishing industry as well as increase their taxes. The respondents were informed that the causes of sea lion population decline are uncertain and that the protection program would not ensure their survival but was designed to increase the probability that the population would survive. The respondents were asked whether they would vote yes on a ballot measure for such a program given that it would result in a specified increase in their taxes. A similar approach might be appropriate for valuation of a habitat protection program. However, the challenge is to effectively communicate the expected impacts of the program to elicit the value individuals actually place on those outcomes. This is tricky since the specific outcomes themselves are not well understood and the public is not likely to be familiar with the particular organisms or physical features that may be affected. Essentially they are being asked to value something they know virtually nothing about solely on the basis of information provided by individuals conducting the survey.

Nonmarket valuation studies are likely to be very expensive to carry out correctly and the results may only be applicable to the specific case. Benefit transfer techniques that allow results from one study to be extrapolated to another area can reduce the cost of valuation, but require that the characteristics of the locations and user groups are highly similar. Boyd and Waigner (2003) describe how benefit transfer, and a related technique they refer to as an indicator approach, can be used to estimate the value of a site using information from a valuation study done in another location.

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. For example, if an exhaustive analysis of recreational benefits at the study site found a benefit of $10 million, the value of the new site may be more or less, or close to the study site, depending on observed differences in the sites.

The indicator approach suggested by Boyd and Waigner (2003) is similar if less formal than standard benefit transfer techniques.

First, indicators can be statistically evaluated in order to establish a more scientific justification for their use as proxies for monetized benefit estimates. In principle, statistical analysis can determine the success of specific indicators as predictors of a known set of ecosystem benefits. What is needed is meta-analysis, taking established benefit estimates from existing studies, and testing the degree to which indicators are predictive of those benefit estimates across a range of studies and locations. The results of such an analysis would validate or reject individual indicators' predictive value, on average, across different study areas. Indicators can then be used as inputs to so-called benefit transfer studies.

Boyd and Waigner's research relates to terrestrial areas, but presumably similar techniques would be applicable in the marine environment. We are unaware of any nonmarket valuation studies of areas closely enough related to habitat in Alaskan fisheries to be used in a benefit transfer exercise, but in designing any future valuation of marine habitat for Alaska fisheries, the ability to use these values effectively for benefit transfer should be considered. This may be best accomplished by linking the valuation to indicators of habitat value for different types of habitat. However, it should be noted that the accuracy of benefit transfer techniques may be low. A Monte Carlo study by Chattopadhyay (2003) finds that, "even under extremely favorable transfer conditions, quality of transferred benefits is poor."

Market-based Instruments and Property Rights Approaches to EFH Protection

The focus of the discussion so far has been on how to estimate the benefits and costs of command and control style approaches to habitat protection such as area closures and gear restrictions. Market-based environmental management approaches that rely on economic incentives or property rights might significantly decrease the costs and increase the effectiveness of an EFH protection program. Savings would be due to added flexibility that would allow damage reduction or mitigation measures to be implemented by those with the lowest cost of doing so.

Market-based environmental management tools are often divided into tax-based and quantity-based systems. Tax based systems levy a tax on undesirable activities or subsidize desirable ones in order to achieve environmental objectives. In principle, a tax on a negative activity set equal to the marginal social cost of damage associated with that activity should result in a reduction of that activity to an optimal level. Generally this will not eliminate the activity since the marginal cost of avoiding damage often rises as the level of activity is reduced and the marginal benefits

may also decline. Subsidies for abating damaging activities can, in principle, achieve similar results to taxes. In either case, the individual begins to internalize the external cost of the damage associated with the activity.

Implementing a system of taxes or subsidies for protection of EFH might be possible but presents some serious difficulties. An efficient tax or subsidy system would require estimates of the marginal social cost of damage associated with particular fishing activities. As discussed above, determining the economic value of habitat and the marginal social cost of damage is problematic. The difficulties go beyond simply estimating the value of protecting a particular area of habitat. The marginal social cost of habitat damage in a particular location is likely to change over time. Fishing activity in an area may reduce the value of the habitat that remains there which will reduce the marginal social cost of further activity in that area. For example, it may be that most of the damage to an area might be incurred after a few passes of a dredge. Subsequent fishing in that area may inflict little additional damage. If the area is left unfished for a while and begins to regenerates, the value of the habitat will begin to increase again. Thus, an efficient tax system would require adjusting the tax according to the current state of the habitat in each location.

Quantity-based systems, such as cap-and-trade systems, set an upper limit on a measurable activity or output and create a market to allocate rights efficiently. They can achieve a cost-effective outcome though overall economic efficiency is dependent on setting the total quantity allocated at the optimal level. Cap-and-trade systems have been implemented for pollution emissions, water use and in fisheries in the form of individual transferable rights to catch or effort. Rights to shares of the activity or output are often allocated to existing users on the basis of historical use, but trading amongst users eventually leads to a more efficient distribution of use rights. There may be restrictions on trading if the location of the activity matters. Other restrictions on trading or concentration of ownership may be imposed to restrict market power or achieve social objectives. Rights may be allocated or held by individuals, companies, or cooperatives. For fisheries, these property right systems have been introduced to limit over-exploitation and to end the race-to-fish which dissipates resource rents by increasing the costs of fishing and often by reducing the value of the fish landed. They do not generally internalize other costs associated with fishing such as damage to EFH. However cap-and-trade systems for environmental bads such as volatile organic chemicals have proved effective at achieving environmental objectives cost-effectively. In some cases it may be useful to more carefully define property rights for fishery exploitation to address these incidental outcomes of exercising catch rights. This may involve bundling, attenuation, or division of resource use rights (Edwards 2003).

Defining catch rights to better protect habitat might take the form of attenuating the exercise of those rights by proscribing certain fishing methods or fishing in specific areas that results in excessive damage to EFH. Alternatively, assuming that impacts can be monitored and measured, bundling of rights might allow holders of catch rights to inflict a level of habitat impact or bycatch proportional to their catch rights. Neither approach can be expected to create incentives for habitat stewardship or result in an optimal degree or spatial distribution of avoidance of habitat damage. Edwards (2003) suggests that "Introducing habitat attributes to the bundle of property rights requires the right to exclude others from your territory" He suggests that territorial use rights (TURFs) might accomplish this. However, TURFs present both political difficulties and practical difficulties. Multiple species are likely to be found in and move through any given

location. It is probably not possible to give an individual entity exclusive right to all fishing in an area big enough to enclose all the species fished there. Holland (2004) suggests that spatial fishery catch rights in the form of spatial Individual Transferable Quotas (ITQ) might facilitate habitat protection. These catch rights would not give exclusive rights to all fishing in a particular area, but the catch rights themselves would specify where they could be exercised. This might allow Coasean bargaining approaches to habitat protection similar to land covenants. For example right holders in certain areas might agree to easements on their fishing rights precluding the use of certain gears. This may still require contracting with a number of rights holders, but it would not be possible for rights holders in other areas to move in to the covenanted areas and use the restricted gears. It is not clear, however, who would contract with fishers for these easements and who would pay for them since the benefits are essentially a public good.

An alternative to attenuation or bundling of marine resource use rights is to create a separate property right associated directly with habitat impacts. Experience in Australia with property rights for irrigation water has demonstrated the value of separating the property right for removals of water from a river or groundwater aquifer from negative outcomes such as salinity loading associated with its use (Young and McColl 2002, 2003). The authors note that negative externalities associated with water use can vary with how and where the water is used. A separation of the rights for the negative impacts of water use from the quantity of water used will tend to lead to a shift of use to users with lesser external impacts per unit of water use. Thus, a given level of abatement of those impacts can be achieved at lower cost than either reducing total water use or specifying consistent best management practices for all users. A similar approach might be taken with habitat impacts of fishing. A correctly designed approach would lead to a distribution of rights that would equate marginal costs of avoiding habitat damage across fishers thereby minimizing the cost of achieving a given level of habitat protection.

Developing, monitoring and enforcing a system of individual property rights for habitat impacts in Alaskan fisheries is likely to be difficult and costly. It may be more practical and less costly to monitor the impacts associated with a group of fishers than to monitor individual impacts. If so, it might be possible to devolve responsibility for achieving habitat related outcomes to a cooperative of fishers. The cooperative would be responsible for achieving specific outcomes or face some penalty such as fines, taxes, or a limiting of catch rights. This approach may allow more flexibility in how habitat impacts are avoided which could reduce the cost of achieving those outcomes.

Cooperatives in Alaskan fisheries have had some success at managing external effects of fishing manifested in the form of bycatch. The "Sea State" system used by the bottom trawl fleet in the Gulf of Alaska and the Bering Sea facilitates a cooperative approach to minimizing the cost of caps on bycatch of halibut, salmon, and crab. The members of this cooperative share spatially explicit information about bycatch rates and agree to avoid fishing in areas with high bycatch rates. This allows the fleet to take more or all of TACs that might otherwise be constrained by fishery closures. However, cooperative approaches such as Sea State that rely on voluntary restraint by members are vulnerable to free rider behavior that limit their effectiveness. Sea State was no exception to this and its effectiveness was limited by a lack of participation by some of the fleet (Holland and Ginter 2001). Cooperatives such as the Pollock Conservation Coop with more formal legal contracts that clearly define the rights and responsibilities of individual

members may be more effective at achieving outcomes. Such cooperatives effectively create individual rights but leave the definition, monitoring, and enforcement of those rights to the membership of the cooperative rather than government.

Economists working on market-based tools to control nonpoint pollution have explored a variety of contracts that might be useful for facilitating a cooperative approach to habitat protection. The primary characteristic of these contracts is that they impose a collective penalty or withhold a collective subsidy if a pollution abatement target is not met. The share of the penalty that the individual pays or the conditional subsidy that they receive must be higher than their individual marginal cost of abatement. Early proposals of this type of system (Meran and Schwalbe 1987, Segerson 1988) required individually tailored contracts with each participant in the system. The informational requirements were likely to be prohibitively costly unless abatement costs are homogeneous across the group. Bystrom and Bromley (1988) propose a system that allows for identical contracts or a single contract with the group even when the group is heterogeneous. The system requires full participation of all parties affecting the ambient standard being monitored and trading of and side payments for abatement activities. It effectively creates a "virtual" tradable permit system that should allocate abatement activities such that marginal abatement costs are equated across participants. Bystrom and Bromley develop a theoretical exposition of this system with an enforceable contract achieved by a subgame perfect Nash equilibrium.

Economists at Ohio State University are undertaking some interesting work that will test real world applications of these approaches. Pushkarskaya (2003) and Taylor (2003) explore variations on these group contacting systems for nonpoint pollution using an experimental economics approach. Randall (2003) describes this work which was undertaken by two doctoral candidates working under him:

> "Pushkarskya's contract solves a 2-stage generalized agency problem. The regulator offers to buy pollution reduction credits from a contracting team of NP sources. The adverse selection problem is solved via a bidding process that selects the least-cost team of NP abaters. Individual abatement targets are assigned and enforced within the team. The incentive-compatibility of this contract can be proven only if it is assumed that team members know each other's costs. While this clearly assumes greater knowledge than exists, it is reasonable to assume that NP sources in a sub-watershed (neighbors farming in a natural amphitheater) have better information about each other's costs than the regulator. Taylor's contract relaxes the assumption regarding team members' knowledge of others' costs. He, too, solves a 2-stage generalized agency problem. The regulator solves the adverse selection through an abatement procurement auction. NP sources bid individual abatement quantity and associated price; the contracting team is formed from the lowest cost bidders. Moral hazard is solved by an "all-or-nothing" team contract – all are paid if the aggregate NP abatement target is met, but none are paid in the event of a shortfall.

Experimental work on these contract forms is ongoing and moving toward actual application as Randall (2003) explains. Pushkarskaya is exploring the performance of her contract in a structured series of experiments in a laboratory setting and soon she will begin to experiment with farmers using a mobile laboratory. The research team has completed a series of focus groups using Taylor's contract with farmers, in a setting that explicitly acknowledges the uncertainty of

abatement performance. One interesting finding is that farmers tend consistently to worry more about unfavorable weather than about shirking within the group, as sources of possible failure to meet the group abatement commitment. As a result, risk averse individuals tend to exceed their contracted abatement responsibilities to ensure the target is not missed by chance. The final step in research currently planned will be a demonstration in a watershed in Ohio, where farmers have agreed to use Section 319 funds (money that typically is used to subsidize implementation of Best Management Practices) for payments to farmers as specified by Taylor's contract.

Experimental work of this type might be quite useful for studying cooperative contracts for EFH protection. This research would also be highly relevant for bycatch reduction programs and in general for cooperative approaches to fishery resource management

Conclusions and Recommendations

The MSFCMA has recognized the importance of EFH for maintaining healthy fisheries and requires fishery managers to take actions to minimize the impacts of fishing on EFH to the extent practicable. The requirement for practicability means that the benefits of EFHPM must be balanced against their costs. However, the information required to design an economically optimal set of EFHPM is probably not available for any fishery anywhere in the world and clearly is not for Alaskan fisheries. Valuation and even methods for valuation of benefits and of costs of EFHPM are inadequate.

In this report, there is a discussion of the basic elements of a benefit-cost analysis or a cost-effectiveness analysis for EFHPM. A number of different types of benefits, both use and nonuse benefits, which must be quantified and monetized if they are to be used in a formal benefit-cost analysis have been identified. There is also a discussion of the potential costs of EFHPM and how they might be measured. This discussion identified large gaps in knowledge and in valuation methodologies. On the basis of the relative importance of these knowledge gaps and the feasibility of filling them with new research, a number of potentially productive areas of research are indentified. This research would improve our ability to more optimally balance habitat protection against costly constraints on fishing. The research areas identified include random utility modeling of location choice, bioeconomic modeling to estimate fishery benefits of habitat protection, non-market valuation of habitat protection programs perhaps paired with research on benefit transfer techniques, and market based systems to cost-effectively achieve specified habitat protection objectives.

In conclusion, it will be some time before it is possible to quantify the benefits of EFHPM. Yet it will probably be necessary to ensure some level of habitat protection without fully understanding the benefits of doing so. Consequently there is a need to develop cost-effective ways of achieving a given level of habitat protection. This should go beyond developing the tools to quantify the relative costs of different sets of EFHPM. What may be most useful is to develop systems that utilize economic incentives and property rights to facilitate desired outcomes. Well designed incentive systems could substantially reduce the cost of achieving these outcomes.

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Appendix: Fishery Impacts of Marine Reserves and Marine Protected Areas

Marine reserves (which are defined here as areas closed to all extractive activities) and Marine Protected Areas (MPA) (which may restrict some but not all extractive activities in an area) are specifically mentioned in the final rule on EFH as a means of affording protection to EFH. Various types of MPAs are the primary EFHPM being considered by the NPFMC. There is a large and growing literature that focuses expressly on the impacts of marine reserves and MPAs on fisheries. Although most of this literature does not directly focus on habitat protection and the feedback of habitat quality on fishery production and stability, it is worthwhile to review it here because many of the general arguments made for or against use of marine reserves and MPAs are likely to be brought forward when considering closures designed specifically for EFH protection. In most cases, the proposed fishery benefits from marine reserves come from constraining overfishing rather than protection of fishery habitat per se, but these arguments have relevance even in cases where other management measures such as TACs are effective at constraining catch. If, as is often argued, closing even a fairly large proportion of a fishery will increase or at least will not decrease fishery yields, even when catches are already constrained to optimal levels, benefits from habitat protection may be realized without sacrifice of fishery yields. If habitat protection then feeds back to increase the potential productivity of the fish stock, yields may be increased. On the other hand if closing a proportion of the fishery reduces the surplus production that can safely be taken, EFH closures may result in fishery losses unless improved habitat quality increases the productivity of the fish stock. As the discussion below shows, the answer to this question is hotly debated and far from resolved. What is arguably more clear is that marine reserves and MPAs are likely to increase harvest costs which must be weighed against potential increases in catches.

The literature provides substantial evidence from both empirical and modeling studies that marine reserves can contribute to protection of marine species and habitat *within* their boundaries. In some cases this can yield benefits including amenity values associated with enhanced viewing opportunities inside reserves and existence value associated with preservation of threatened species or, more generally, of biodiversity. It has also been shown that, *under certain circumstances*, marine reserves can increase the benefits derived from marine fisheries. However, no studies have shown that benefits will accrue to fisheries where catches are already constrained to desired levels using output controls as is the case for Alaskan fisheries. A number of studies also identify costs associated with the creation of marine reserves and note that the distribution of costs and benefits is likely to be uneven in most cases. Costs are generally borne by excluded extractive users while benefits accrue to non-extractive users of the reserve and what it protects.

The most general conclusion that can be distilled from the plethora of empirical and modeling studies on marine reserves and MPAs is that their impacts, both positive and negative, will vary greatly depending on design and on the physical, biological and socioeconomic characteristics of the environment in which they are imposed. In addition, the biological effects of a given marine

reserve or MPA will vary across species, and the economic impacts will vary across individuals and groups. This suggests that a generic evaluation of the benefits and costs of marine reserves or MPAs is unlikely to provide general conclusions regarding their net value. It also suggests that any specific marine reserve or MPA may or may not be in the public interest, and that an evaluation of costs and benefits should be undertaken for each specific proposal.

Empirical Studies of Marine Reserves

The literature on marine reserves includes a large number of empirical studies that explore the impacts of closures on fish populations inside the closures but relatively few that document the effects on catches outside them. Though they have been used in many fisheries the impacts of marine reserves are not well understood. A study of time and area closures in over fifty fisheries in eleven countries produced little clear evidence of improved resource conservation (OECD 1997).

Nearly all of the empirical studies that document increased catches after creation or elimination of a marine reserve are isolated examples of fisheries with very different physical and socio-economic characteristics to any in Alaska. A recent comprehensive study of marine fishery reserves worldwide (Ward et al. 2001) concludes "there appear to be few well documented examples of fisheries that have been shown to benefit from the introduction of reserves. The experiences often cited in support of reserves are limited to either the recovery of stocks from a highly depleted state, using temporary closures of various forms, or involve mainly subsistence-scale tropical reef fisheries. Experiences in neither of these categories can be related directly to the world's commercial capture fisheries, and there is little documented evidence that in a well managed fishery, no-take reserves offer additional advantages to a fishery over and above those offered by better classical management techniques."

Modelling Studies Marine Reserves Impacts on Fisheries

As Holland (2002) notes, there are serious limitations to our ability to answer these questions with empirical research. Doing so rigorously would require a number of replications over long periods with comparisons to controls, and even then might only provide conclusions valid for very specific sets of circumstances. The full impacts of reserves can take many years to be realized and will be confounded by environmental and regulatory changes. Modelling studies provide an alternative approach to evaluate basic questions about how reserves of various designs in various environments might affect fisheries.

Several published modeling studies of marine reserves and closed area for fisheries suggest that a correctly sized marine reserve may increase yields in fisheries that are subject to growth or recruitment overfishing, but that little if any yield increases can be achieved in fisheries where effort is already at the level that produces maximum sustainable yield or maximum yield per recruit (e.g., Beverton and Holt 1957, Guénette and Pitcher 1999, Hannesson 1998 and 2002, Hastings and Botsford 1999, Holland and Brazee 1996, Nowlis and Roberts 1999, Polacheck 1990, Rodwell et al. 2002, Sanchirico and Wilen 1998, 1999 and 2002a, Smith and Wilen 2003).

In these models increased yields are achieved through two possible mechanisms. Fishing mortality and growth overfishing are reduced by decreasing the efficiency of effort. This is achieved by closing a proportion of fishable area which, over time, leads to an increase in the density of the stock inside the marine reserve. Catch per unit effort is reduced relative to what it would be if the fish stock or fishing effort was evenly distributed spatially or concentrated where fish are concentrated. If fishing mortality was above the maximum yield per recruit level prior to implementation of the marine reserve, this may lead to increased yield per recruit, but only if mature fish eventually leave the protection of the marine reserve and are caught. Modelling studies have shown that this requires high rates of migration out of the reserve (Polacheck 1990, Holland and Brazee 1996).

Models that incorporate a stock-recruitment function explore the possibility of increasing yields through a second mechanism. If the marine reserve protects a resident stock of fish it may allow an older, more fecund spawning stock to develop, potentially increasing recruitment. However these benefits are likely to be small unless the spawning stock is at low levels since recruitment per spawner is generally considered to be a decreasing function of total spawning stock.

Most of the models cited above compare only equilibrium catches with and without reserves. However, as Holland and Brazee (1996) show, there will generally be an initial drop in catches and it may be many years before catches return to or exceed pre-reserve levels. Thus even, if equilibrium catches are higher post-reserve, the present value of the stream of revenues over time might be lower with a reserve than without it. Holland and Brazee (1996) also show that equilibrium catches with a reserve are unlikely to exceed catches without one if the fishing mortality is already set at the level that yields the highest catch in the absence of a reserve.

Nearly all of the models and empirical studies of marine reserves have focused on open access fisheries or fisheries managed with input controls. While these studies have shown that marine reserves can lead to increased catches in heavily exploited fisheries, they have failed to demonstrate that reserves can increase either catches or profits for fisheries that are optimally managed (with either input or output controls). Furthermore marine reserves do nothing to correct the incentives that lead to dissipation of resource rents and may actually lead to increases in fishing capacity (Hannesson 1998).

Only one modeling study has examined the effects of spatial controls on efforts in an optimally managed fishery. This study by Sanchirico and Wilen (2002b) shows that the distribution of effort might not be optimal in a fishery managed with Individual Transferable Quotas (ITQ) or with a spatially homogeneous landings tax. Optimal effort distribution might be achieved with a spatially specific landings tax or with spatially specific ITQs. However, it suggests that marine reserves, which are equivalent to an infinite landings tax for that area, are generally not optimal policies in terms of economic efficiency.

It is important to keep in mind, that establishing a marine reserve is only one method of reducing fishing mortality and probably not the most efficient or effective method in most cases. Marine reserves may be the only option in some developing countries where resources to control effort or catches directly are not available. However, in Alaska, TACs provide an effective means of controlling fishing.

Although models shed some light on how marine reserves might affect fisheries, the generality of results from existing models is limited by the uniqueness of different marine systems. Comparing different marine reserve models, one can conclude that results are highly sensitive to the specifics of the biological and socio-economic systems embodied in the implicit and explicit assumptions of the models. Results from explicitly spatial models of marine reserves demonstrate that the location and shape of the marine reserve in the context of the larger fishery system is likely to be just as important as its relative size in determining outcomes (Guénette et al. 2000, Holland 2000, Walters and Bonfil 1999). This is particularly true when the fishery system includes multiple, spatially heterogeneous species and user groups. Holland (2000) shows that even when a marine reserve increases the productivity of fish stocks directly impacted by it, other biologically separate fisheries and the incomes of fishermen dependent on them may fall as a result of the displaced effort those fisheries absorb. The benefits and costs of a marine reserve must be addressed in the context of the larger system.

Most of the modeling of marine reserves has utilized single species models. Even in fisheries where it is possible to tightly control the level of fishing effort there may be a limited ability to manipulate the relative levels of fishing mortality on different species that are caught together. Consequently, any level of fishing effort may be too high for some species and too low for others. Fisheries controlled by total allowable catches (TACs) for individual species may be subject to the same problem leading to regulatory discards for species with lower TACs relative to their catchability. If there are consistent heterogeneous patterns in the distribution of different species it may be possible to tune the relative fishing mortality of species using spatial controls on effort (either marine reserves or seasonal closures). Holland (2003) explored this possibility with a linear advection/diffusion model of three intermingling groundfish stocks (cod, haddock and yellowtail flounder). The model suggests that potential equilibrium revenue gains from using area closures to manipulate the distribution of effort are quite limited. This was partly the case because redistributions of effort away from areas with the highest concentrations of cod increases the total cost of achieving a given level of combined revenues from the three stocks. Controls on nominal effort and appropriate mesh size are shown to be the most important management measures.

There are alternative and more efficient means to control the relative catch rates of different species and stocks. For example, individual quota systems provide strong financial incentives to do just this. Quota values of species which are constraining catch of other species will tend to rise reducing the net benefits accruing to individuals who catch them. This provides incentives to individual fishermen to alter the species mix of catch of their catch by changing fishing locations or other aspects of their fishing strategy.

One argument for including marine reserves as part of an optimal fishery management system is their potential utility as a hedge against management failures. However, the evidence that marine reserves are an effective tool for this purpose is equivocal. One of the most cited papers supporting the use of marine reserves as a hedge against management failures is Lauck et al. (1998); however this paper makes a number of strong and unrealistic assumptions that bring into question the generality of the results. This paper shows that, under some very specific conditions, a marine reserve combined with catch controls can achieve a given level of fish stock protection (modeled as the probability the fish stock is maintained above 60% of virgin biomass over a 40 year time horizon) with a higher average catch than a strategy that relies solely on catch controls.

The model assumes that catch varies randomly around a mean equal to the TAC. A key assumption is that, after each fishing season, the stock redistributes itself evenly over the fishable area and the reserve. This assumption is equivalent to assuming a zero diffusion rate during the fishing seasons and an infinite diffusion rate between seasons. This seems quite unrealistic and is key to the results of the model. Although the authors do not develop the point, they show that a greater level of total effort is required to achieve a given level of catch with a reserve in place. This is consistent with other models and suggests that gains in revenues are likely to be offset by increased harvesting cost.

It should also be noted that poorly designed marine reserves have the potential to increase risks in some cases. By displacing fishing effort they may increase pressure in certain areas critical to particular species. They may also serve to concentrate spawning stock biomass spatially. The changes in spatial distribution of adult animals may affect average spawning success and may alter the survivorship of larvae and juveniles by effectively changing the average oceanographic conditions and the predation they face at various pre-recruit stages. Sinclair (1988) argues that the spatial distribution of spawners and ocean conditions that impact retention of eggs and larvae may be much more important than resource limitations or predation in determining recruitment. Larson and Julian (1999) point out that if there is stochastic spatial variation in the sources of successful recruits, spatial concentration of the spawning stock may increase the variance of the stock recruitment relationship and thereby increase the risk of fishery collapse. Under these circumstances, having a number of substocks of spawners distributed spatially is acting as a hedge against spawning failures of other substocks. The value of this portfolio may be reduced by concentrating most of the total spawning stock inside the closed area.

There can be little disagreement that stock assessments are often highly uncertain and that actual catches will often diverge from TACs. This indeed suggests the need for precautionary approaches to management. But, while marine reserves may play some role in applying the precautionary approach in some cases, they too are an imperfect means of compensating for management failures and only one tool among many. Many prominent fisheries scientists argue that classical management tools, augmented with modern risk management procedures, can overcome the fisheries-management problems experienced in the past (Rosenberg et al. 1993, Mace 1996).

Conclusions

A conceptual exploration of the impact of reserves and a review of the literature suggest that few general conclusions can be drawn regarding the net value of marine reserves and MPAs. It seems clear that the total net benefits of a marine reserve or an MPA, including both extractive and non-extractive uses, could be either positive or negative depending on its design and location and the degree to which the area closure or system of closures complements or conflicts with other marine management methods.

Both the benefits of a well-designed system of reserves and the costs of a poorly designed one could be substantial. Due to the importance of this issue and the potentially large magnitude of costs and benefits that may result, a serious effort should be undertaken to assess the costs and benefits of specific marine reserves when they are proposed.

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