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# The Evolution of Non-Market Valuation of U.S. Coastal and Marine Resources

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# The Evolution of Non-Market Valuation of U.S. Coastal and Marine Resources


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## 1. INTRODUCTION

In the U.S., the National Oceanic and Atmospheric Administration (NOAA) is charged with the management and protection of the nation's coastal and marine ecosystems. As resource agencies like NOAA create policy, make regulations, or establish protected areas in these times of increasing calls for more efficient governance, understanding how these decisions impact human welfare, positively or negatively, and the relative magnitude of these impacts is increasingly important. In the U.S., this is institutionalized by policies and executive orders that require federal agencies to consider the full range of benefits and costs these actions would have. For example, the federal government requires the evaluation of the benefits and costs associated with federal regulatory actions through a benefit-cost analysis (BCA) (Executive Order 12866; OMB Circular A-94). BCA is a common tool used by analysts to evaluate the economic efficiency of policy alternatives and requires a complete and accurate accounting of both benefits and costs.

When the policies under evaluation involve natural resource and environmental goods and services, as is often the case with NOAA activities, the benefits are often challenging to measure. This is particularly true when the affected goods and services are ones with passive use, or nonuse, value, which is related to the components of total economic value unrelated to the use of the good or service (Krutilla 1967).[1] In these cases, researchers often turn to non-market valuation methods.

Non-market valuation is concerned with measuring the demand and value of goods and services in the absence of formal markets from which signals of value can be ascertained. In particular, these non-market valuation methods have been developed to measure the total economic value (TEV), defined as the sum of the use and nonuse value of the good or service. OMB Circular A-4 recognizes the importance of non-market valuation techniques for measuring both use and non-use values. There are two types of non-market valuation approaches: revealed preference and stated preference methods. Revealed preference methods use data about people's behavior to infer the value of a non-market good or service (e.g., Bockstael et al. 2007), while stated preference methods use information provided directly from individuals, usually from carefully-constructed survey questions, that reveal their values (e.g., Mitchell and Carson 1989; Bateman et al. 2004). Travel cost models and hedonic price models are examples of revealed preference approaches. The contingent valuation method is the most well-known stated preference approach.

Arguably, the modern era of non-market valuation applied to U.S. coastal and marine resources began with publication of the NOAA Panel on Contingent Valuation (Arrow et al. 1993), commonly referred to as the Blue Ribbon Panel. In addition to specific recommendations about the conduct of contingent valuation studies, the Panel concluded that passive use values were a meaningful component of welfare losses or gains and that contingent valuation provided useful

[1] An extensive literature (e.g., Smith 1987; Kopp 1992; Bishop and Welsh 1992; Cummings and Harrison 1995) points to the importance and challenges of including these values in BCA when present. Despite the challenges, it is recognized that a full accounting of the benefits and costs is necessary, including both "tangible and intangible benefits and costs" (OMB Circular A-94).

information on their magnitude. The Panel also correctly predicted that the controversy surrounding measurement of passive use values would not disappear. Two recent updates of the state-of-the-art, one favorable (Carson et al. 2014) and one antagonistic (Hausman 2012), prove the prescience of the Panel.

Another major touchstone for the advancement of non-market valuation adoption was the publication of the initial Millennium Ecosystem Assessment (2005). The report provided consistent and useable definitions of ecosystem services, distinguished market and non-market services, and made the link with measurements of human well-being. An ecosystem services approach and the concurrent need for valuation, in general, is being broadly adopted by federal agencies, for example, see the Federal Resource Management and Ecosystem Services Project as part of the National Ecosystem Services Partnership at Duke University[2], or a specific example for the Army Corps of Engineers (Murray et al. 2013).

This paper provides an overview of how the current state of both stated and revealed preference non-market valuation has evolved since the NOAA Blue Ribbon Panel report and publication of the Millennium Assessment, specifically applied to U.S. management of coastal and marine resources, and particularly from the perspective of NOAA. We discuss the development of the current state of application of non-market valuation in four areas: marine and coastal damage assessments, marine protected resources, recreational fisheries, and coastal management. We find that there has been significant progress in the development of non-market valuation tools applied to marine and coastal resources, although their use and application in policy making has not been as robust. Recommendations are made to encourage greater adoption of results from these studies into policy decisions.

## 2. NON-MARKET VALUATION IN NATURAL RESOURCE DAMAGE ASSESSMENT

The Natural Resource Damage Assessment (NRDA) process seeks to make the public whole for damages to natural resources resulting from contamination events, including oil spills, heavy metals, and polychlorinated biphenyls (PCBs), as well as for impacts resulting from vessel groundings on coral reefs. The assessment process is supported by underlying Federal legislation—the Oil Pollution Act (OPA), the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), or the National Marine Sanctuaries Act—depending on the specific impact under evaluation. Various Federal (for example, NOAA, National Park Service, Fish and Wildlife Service) and State authorities are typically involved in the various NRDA phases, with "trustee" responsibilities over differing resources. Damages are typically to ecosystem services for which there are no explicit markets such as recreation, aesthetic and cultural

[2] http://sites.nicholasinstitute.duke.edu/ecosystemservices/research/nesp-frmes-project/


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values. As there can be an interest in monetizing these losses to establish the required settlement for damages, the use of non-market valuation has a long history in the NRDA field.

Review of selected cases with publicly available information on the NOAA Damage Assessment, Remediation, and Restoration Program Web site (http://www.darrp.noaa.gov/) revealed use of a range of valuation approaches for assessment of impacts to coastal and marine resources (see Table 1).

*Table 1*. Use of Non-Market Valuation in Selected Coastal and Marine Natural Resource Damage Assessment Cases

| Year of Draft or Final Restoration Plan | Case | Type of Contamination | Valuation Method Used | Reference |
|---|---|---|---|---|
| 1994 | Exxon Valdez | Oil | Contingent valuation | Carson et al. (1991) |
| 1997/2000/2003** | Tampa Bay | Oil | Travel Cost | Florida DEP & NOAA (2000) |
| 2001 | Lavaca Bay | Mercury | Travel Cost*** | TX General Land Office et al. (2001) |
| 2001 | New Bedford | PCBs and other hazardous materials | Travel Cost | McConnell (1986) |
| 2001 | American Trader | Oil | Benefit transfer | Chapman and Hanneman (2001) |
| 2002 | Chalk Point | Oil | Benefit transfer | Byrd et al. (2001) |
| 2005/2012** | Montrose/PV Shelf | DDTs and PCBs | Contingent valuation | NOAA (1994) |
| 2009 | Athos Spill | Oil | Benefit transfer | NOAA et al. (2009) |
| 2012 | Cooper River, M/V Ever Reach | Oil | Travel cost (Shrimp baiting) /benefit transfer (shellfishing - from shrimp baiting study) | South Carolina Department of Natural Resource et al. (2012) |
| 2012 | Cosco Busan | Oil | Benefit transfer (boating and fishing)/Travel cost (shoreline use) | California Department of Fish and Game et al. (2012) |
| 2014 | Bouchard | Oil | Benefit transfer/License Demand analysis* | NOAA et al. (2014) |

*Based on license demand
**Multiple phases for restoration plans
***Random utility model used but travel cost not explicitly included

These have included contingent valuation, travel cost, and benefit transfer methods, the process of applying existing value estimates in the literature to a scenario for which they were not originally intended (see, for example, Johnston and Rosenberger 2009). To our knowledge choice experiments have not yet been applied to estimation of NRDA settlements. Benefit transfer and travel cost approaches have been applied more frequently than primary contingent valuation studies. Despite this, perhaps the most well-known application to date of non-market valuation for NRDA in the United States was for the Exxon Valdez oil spill; the contingent valuation study developed for that case estimated a WTP to avoid another similar oil spill of $2.8 billion (Carson et al. 1992).

When reviewing the valuation methods used, there does not appear to be a general pattern toward the use of one valuation approach or another in NRDA for coastal and marine areas; instead, the choice of valuation method may be driven by specific case considerations. Scope of the damage, and potentially impacted natural and human communities, may be an important consideration for the type of method selected. As indicated by the table, the majority of the assessments where dollar values were calculated used some form of benefit transfer estimates based on existing economic valuation literature. These transferred values were often combined with on-site estimation of the levels of visitation and changes in visitation resulting from the contamination event (e.g., Chalk Point, Bouchard, Athos). If relevant value estimates were not available or appropriate, site specific studies have been conducted for specific recreational categories (e.g., shoreline use in Cosco Busan, shrimp baiting in Cooper River-M/V Ever Reach). Benefit transfer estimates have also been adjusted to take specific considerations of the population under evaluation into account (e.g., boating values used for Cosco Busan, which were adjusted to reflect uniqueness of San Francisco Bay).

Though ecosystem service valuation has become a focus of many valuation efforts reported in the literature, most of the monetary valuation approaches employed in NRDA do not attempt to place a monetary value on all of the services provided by an impacted ecosystem. Importantly, the majority of the values derived in cases included in Table 1 focused on the loss of recreational ecosystem service (or human use) benefits; whereas,the contingent valuation studies tend to bebroader and attempt to measure total value of the resource base. This more extensive approach may not be appropriate when there are a clearly defined set of impacted user groups. This does not, however, indicate that other ecosystem functions or services are not accounted for in NRDA. For most of these cases, a separate non-monetary approach, termed habitat equivalency analysis (HEA), has been used to estimate the loss of ecological services, which can be services performed by a natural resource for the benefit of another natural resource, not solely for humans (the commonly used definition of ecosystem services). HEA uses a combination of site-specific ecological data and expert judgment based on the literature to determine the losses in ecological services from a contamination event and potential gains from restoration (Thur 2007). Cases will often therefore, where needed, have an evaluation of impacted ecological services in addition to impacted human use services.

While these cases typically assess the value of losses from a contamination event, the monetary value of the gains from the proposed restoration is rarely assessed using similar economic approaches. This may be a consequence of the challenges and costs associated with developing surveys that focus on specific restoration actions (one case may have several restoration alternatives) as well as the challenge of linking habitat changes (if manmade structures are not proposed) to ecosystem service benefits. Ongoing development of ecosystem production function models linking changes in ecological condition with changes in ecosystem services may be beneficial in this regard.

## 3. NON-MARKET VALUATION FOR PROTECTED RESOURCES

In the U.S., the National Marine Fisheries Service (NMFS) is responsible for conserving, protecting, and recovering marine species listed under the Marine Mammal Protection Act (MMPA) and Endangered Species Act (ESA). These species are referred to as protected species herein. At present, this group includes 125 marine mammal species located worldwide and managed under the MMPA, as well as 122 threatened and endangered marine species listed under the ESA. Twenty-eight of the marine mammal species protected under the MMPA are also listed as threatened or endangered under the ESA and include numerous cetacean (whales) and pinniped (seals and sea lions) species. The ESA-listed protected species also include sea turtles, fish, invertebrates (corals and abalone), and a single plant species (Johnson's seagrass). Recently, 20 new coral species were listed as threatened under the ESA (50 CFR Part 223). Two additional coral species (elkhorn and staghorn coral) were listed as threatened in 2006.

Economic values for protected species, generally measured in terms of willingness to pay (WTP), primarily arise from the non-consumptive value people place on them. Use values, such as the value one would obtain through the harvest or consumption of the species, are theoretically prohibited in the case of threatened and endangered species, and are likely a small component of the value placed on MMPA protected species (e.g., as bycatch in target fisheries). Thus, the total economic value of protected species are generally believed to be primarily comprised of non-consumptive values, including existence or bequest value arising from nonuse motivations and non-consumptive use value, such as the benefits from viewing or learning about a species.

Valuing protected species has been an active research area in non-market valuation for several decades.[3] Until recently, all of these studies have relied upon contingent valuation (CV) methods to measure WTP. CV methods are differentiated by the way they elicit WTP. Respondents are commonly asked to state their maximum WTP (an "open-ended" CVM question), choose the amount they are willing to pay from a list of values (a "payment card" CVM question), or accept or reject a specific amount (a "referendum", or discrete-choice, CVM question). Open-ended questions have been criticized as lacking incentive compatibility and leading to biased WTP estimates (e.g., Arrow et al., 1993; Hanemann, 1994; Carson, Flores, and Meade, 2001), but provide direct measures of WTP and do not require estimation. The payment card CV question similarly does not require estimation, as respondents select a value from the proffered amounts to

[3] This is reflected in three meta-analyses that have been conducted summarizing the extant literature. Loomis and White (1996) and Richardson and Loomis (2009) provide useful summaries of species valuation studies conducted in the U.S. through the early 2000s, while Martin-Lopez et al. (2008) conducts a similar analysis but includes studies conducted outside the U.S. The studies covered by the latter study suggest this research is most active in the U.S. (65% of all studies), though there are considerable numbers of species valuation studies conducted in Europe, Australia, and Canada (Martin-Lopez et al. 2008).

represent their WTP, although estimation approaches are sometimes used (Cameron and Huppert 1989). However, the range of values presented to respondents may influence WTP (Rowe et al. 1996). The referendum format is generally believed to be incentive compatible (Carson and Groves 2007), and was recommended in the NOAA Blue Ribbon Panel Study (Arrow et al. 1993). However, a drawback of this approach is it requires a larger sample size (Alberini 1995) and is sensitive to the amounts respondents are asked to accept or reject (e.g., Green et al. 1998).

Studies involving the estimation of WTP values for threatened and endangered species began in the early 1980s. These early studies focused primarily on terrestrial species. For example, Brookshire, Eubanks, and Randall (1983) used an open-ended CV question in a survey of licensed hunters to value the option price and existence value of grizzly bears, a threatened species, using stamps for future hunting as payment vehicles. Using referendum CV questions, Bowker and Stoll (1988) estimated the value of preserving the whooping crane, an endangered species, to both users and nonusers of the Aransas National Wildlife Refuge, where the majority of the population is found. A third early study by Boyle and Bishop (1987) estimated the total economic value of the bald eagle and a small fish species, the striped shiner. The bald eagle was listed as a threatened species at the time of the study, but the striped shiner was not a listed species. This provided a useful contrast for examining WTP differences between a well-known and iconic species and a relatively unknown one that is not endangered. They used a referendum CV question in a survey of Wisconsin residents, and their empirical results suggested that nonuse values for bald eagles may be much larger than those for the more obscure and non-listed striped shiner.

The earliest effort to value protected marine species was a government study by Hagemann (1985). In this study, a mail survey of California residents was used to estimate the value of bottlenose dolphins, California sea otters, Northern elephant seals, gray whales, and blue whales. Of these, only the California sea otter (threatened), gray whale (threatened), and blue whales (endangered) were listed species at the time the study was conducted. Respondents to the survey were asked to indicate their WTP for a protection fund to preserve existing population levels of each species in a payment card CV format with a follow-up open-ended CV question. The utility of the estimated WTP values for policy was limited, in part, due to the small samples used to generate welfare estimates, the low response rate (21% overall) that raised questions about the representativeness of the sample, and the fact that only California households were sampled. These limitations preclude the extension of value estimates to the larger U.S. population unless it is assumed that preferences for these marine mammals are identical outside California. Pate and Loomis (1997) provide evidence that preferences for wetland and wildlife protection in the San Joaquin Valley in California are different for respondents who live further away, suggesting that the assumption of identical preferences for non-target populations is not prudent.[4] A recent study by Johnston et al. (In press) provides evidence that WTP values for threatened and endangered marine species may not adhere to the distance-decay hypothesis explored by Pate and Loomis

[4] This portability issue is a trait this study has in common with other marine mammal valuation studies, specifically, Samples and Hollyer (1990), Loomis and Larson (1994), and Solomon, Corey-Luse, and Halvorsen (2003), which all estimate values for survey populations at a sub-national scale.


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(1997), but nevertheless show that geospatial heterogeneity of these values are present and potentially important.

During the 1990s, several additional CV studies were conducted to estimate public WTP values for protected marine species. Samples and Hollyer (1990) conducted a study to understand public values for two endangered species--humpback whales and Hawaiian monk seals. Information about how much money or time respondents would be willing to donate to preserve these species was collected in an in-person survey from a small stratified sample of Oahu (Hawaii) residents based on age, income, and gender. Several survey versions were employed that differed in the order the species were valued and whether respondents were told that only one or both species were threatened. The study used open-ended CV questions to elicit WTP values. As with the Hageman study, the welfare estimates from this study were based on very small samples (each between 53 and 72 responses) and were for a limited geographic sample. Another study from this period estimated the value Massachusetts households place on preserving Atlantic salmon[5] (Stevens et al. 1991). In this study, a mail survey of 1,000 households was used to collect responses to two CV questions, an open-ended CV question and a referendum CV question. Response rates were low (only 169 respondents were included in the estimation) and protest responses were not removed from the data for model estimation, a practice that has become standard practice in modern applications (Carson, Flores, and Meade 2001).

To assess whether WTP for gray whale increases is invariant to the size of the increase, Loomis and Larson (1994) undertook an in-person intercept survey of whale-watchers and a household mail survey in California. Using open-ended CV questions, the questionnaires asked respondents how much they would be willing to pay into a special protection fund that would be used to increase the gray whale population by 50% and 100%. The intercept survey targeted visitors at four whale-watching locations, while the mail survey was sent to a random sample of California households. Overall response rates were much higher than those achieved by Hageman (1985), Stevens et al. (1991), and Samples and Hollyer (1990), with 71.3% (672 respondents) of the intercepts yielding completed surveys and 54% of the household surveys (519 respondents) being completed and returned. They found users willing to pay more for population increases than non-users, and values for the larger population increase were found to be significantly greater, indicating preferences that are consistent with economic theory.

Another study from the 1990s investigated the role of uncertainty on WTP for preserving the loggerhead sea turtle, a threatened species, in a mail survey of North Carolina households (Whitehead 1992, 1993). The survey achieved a response rate of 35%, or 225 individuals responding to the survey, and included responses to a referendum CV question that asks respondents to value the preservation of the species via a “Loggerhead Sea Turtle Preservation Fund.” Uncertainty is introduced by asking for the respondent’s subjective supply uncertainty with respect to the probability of loggerhead sea turtles going extinct in the next 25 years.

[5] The Gulf of Maine population of Atlantic salmon is currently listed as endangered under the ESA, but at the time of the study, was not an ESA-listed species.

Additionally, demand uncertainty is introduced by questions about the probability of visitation to a North Carolina beach in order to observe or photograph threatened or endangered species. This question acts as a "proxy for sea turtle demand uncertainty since sea turtles are one of the most conspicuous and publicized of the coastal threatened and endangered species in North Carolina." (Whitehead 1992, 982). He found that WTP was sensitive both to the subjective assessment of the risk of extinction (and hence the implicit effectiveness of the protection program valued) and the probability of viewing or photographing the species in the future.

During the early 2000s, contingent valuation remained the primary method employed to value marine protected species. For example, Giraud et al. (2002) estimated the economic value of Steller sea lions to Alaskans and the overall U.S. population.[6] The questionnaire used in this study asked a referendum CVM question that involved voting for a measure that would create an "Enhanced Steller Sea Lion Recovery Program", but would lead to an increase in federal taxes to the respondent's household. Surveys were mailed to a stratified sample of U.S. households, Alaska households, and households living in Alaska boroughs that contain Steller sea lion critical habitat. The overall response rate was 63.6%, with a 51.16% response rate from the national sample. WTP estimates for the different samples were not statistically different, although the mean values were larger for the U.S. population relative to the local and state sample estimates. Solomon, Corey-Luse, and Halvorsen (2004) valued the endangered manatee. A mail survey was sent to a sample of households in Citrus County (Florida) drawn from phone books and stratified by gender. The survey achieved a 36% response rate. Respondents were asked to indicate their WTP in donations to a fund to protect manatees under the counterfactual that government protection of manatees in Florida was removed. A modified payment card CV question was asked, and WTP was estimated using a sample size of 297. Like other studies discussed above, the small sample and low response rate preclude extrapolating the results to the population (in this case, households in Citrus County).

Bell et al. (2003) conducted a mail survey of households living within 30 miles of five estuaries in Oregon and Washington to collect information on their preferences for enhancement of local coho salmon populations. At the time of the survey, several coho salmon populations had been listed under the ESA or were in the process of being listed. In the survey, they ask CV questions that present enhancement programs that double or quadruple the salmon run sizes in the estuaries in Washington State, and programs that would result in the de-listing of the coho salmon populations in Oregon. Respondents were asked a referendum CV question that presents the opportunity to vote yes or no on a local coho enhancement program. Only one enhancement program, presented as either a high or low level of enhancement, was presented to each respondent. For the Washington estuary programs, the low level was a doubling of current runs and catch, while the high level was a quadrupling of current runs and catch. For the Oregon estuaries, the low level was defined as preserving the local coho population to avoid extinction and the high level would achieve that level plus increase allowable catch by a specified amount per year. One thousand surveys were administered in each of the five estuary areas, with response rates varying

[6] See also Turcin (2002) and Turcin and Giraud (2003).


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from a low of 49% to a high of 62%. There were 2,209 completed surveys returned, and 2,006 included responses to the CV questions, although protest responses reduced the sample sizes by 4-10%, depending upon the area. The results suggested that WTP was not sensitive to the different enhancement levels and, unsurprisingly, high income households are generally willing to pay more for local coho enhancement than lower income households.

The studies reviewed thus far have relied upon CV methods. These methods, though capable of producing reliable estimates of passive-use (or nonuse) values, can be limited in scope and application. For example, in many CV studies on protected marine species respondents face an all-or-nothing choice between paying for a state of the world in which a species or a group of species experiences a large discrete change or paying nothing and maintaining the status quo.[7] The discrete change is usually expressed in terms of preventing extinction, decreasing the risk of extinction or degradation, increasing the population size, or enhancing the status of the population (e.g. down-listing a species from an *endangered* to a *threatened* status or declaring a species *recovered*). Several artifacts of this approach inhibit its management application. First, the estimated value can only be applied to changes identical to those described in the CV survey. Unknown or uncertain biological/ecological conditions, as well as unintended actions, may lead to different population outcomes to which the estimated WTP value is not applicable. Alternatively, if the program is not administered in the way described in the CV survey, the WTP value may not be applicable to the extent respondents who had indicated support for the program were indicating support for the outcome (e.g., preservation) and means to achieve the outcome (e.g., specific management actions described). Second, if the change involves aggregate species it is generally not possible to obtain individual species values, which may be needed for actions such as recovery planning.

Motivated in part by this limitation, non-market valuation researchers began utilizing the stated preference choice experiment (CE) approach. The approach, though relatively new to the valuation of environmental goods, has a long history in the marketing and transportation fields (Louviere 1992). CEs are grounded in Lancastrian consumer theory (Lancaster 1966) and specify that the utility one has for a good can be decomposed into individual, separable attributes. In a CE, each attribute takes on a range of levels, typically set by the researcher with attention to feasibility and policy needs. Experimental design plans are used to generate different combinations of attribute levels into an alternative, and respondents are shown choice sets that consist of two or more alternatives and asked to choose their most preferred alternative (and in some cases least preferred) from the set. Choice responses are then used to estimate a preference function that depends on the levels of the attributes. A fully detailed explanation of the CE approach can be found in Adamowicz et al. (1998).

[7] While willingness-to-accept compensation for extinction or degradation of a species population is sometimes the more relevant welfare measure, empirical and experimental evidence has pointed to the use of willingness-to-pay welfare measures in stated preference surveys ( Hanemann 1991; Arrow et al.1993; Adamowicz et al. 1993; Mansfield 1999)

In a CE, the economic value for changes to attributes of a choice alternative can be obtained in a straightforward fashion. For example, if the choice is between competing threatened and endangered species protection programs that differ in the resulting population level of a species, the marginal value of changes in population can be derived directly from the estimated preference function. This makes CEs particularly attractive as a flexible means of estimating the economic benefits resulting from a wide range of policy scenarios. CEs also allow marginal values for other attributes related to species protection to be estimated. For example, scientific uncertainty about projected biological outcomes can be incorporated directly into the preference function, or respondents can make explicit trade-offs among different species under varying conditions to provide information on preference ordering when multiple species are in question. In addition to being more flexible for policy than the traditional CV approach, some research suggests that CEs can reduce yea-saying (accepting a bid amount regardless of one's preferences) by eliminating the all-or-nothing choice faced in the more traditional CV study (Ready et al. 1996; Brown et al. 1996; Hanley et al. 1998) and identify the potential for embedding (insensitivity to the amount of good provided) by building in direct tests of scope (Alpizar et al. 2001).

Most of the CE research involving threatened and endangered marine species has been confined to North America. To our knowledge, there have been four of these CE studies conducted in the U.S.[8] The first was an unpublished study by Layton et al. (2001) that used a stated preference choice experiment approach to value Pacific salmon enhancement in the Pacific Northwest. The second focused on the western stock of Steller sea lions (Lew, Layton and Rowe 2006), an endangered mammal found primarily in the Gulf of Alaska and Aleutian Islands. A CE survey was administered to samples of Alaska households and non-Alaska U.S. households and responses from it were used to estimate non-consumptive values for enhanced protection of the western stock of Steller sea lions. Departing from the traditional CV approach, the CE design of this study resulted in WTP values for a range of policy scenarios involving Steller sea lion population increases and ESA status improvements. In addition, the authors examined the effect of scientific uncertainty about Steller sea lion population trajectories (absent additional management actions) on WTP values. Results show that, for small enhancements in protection, value estimates are sensitive to baseline population trajectories but WTP differences become insignificant for relatively larger protection enhancements.

The third study was a multi-species CE study conducted with U.S. households to estimate the value of recovering or downlisting eight marine species listed as threatened or endangered under the ESA (Wallmo and Lew 2012). Species include the loggerhead sea turtle, leatherback sea turtle, North Atlantic right whale, North Pacific right whale, Upper Willamette River Chinook salmon, Puget Sound Chinook salmon, Hawaiian monk seal, and smalltooth sawfish. The CE design allowed respondents to make explicit trade-offs among species-specific improvements, enabling a preference ordering among the eight species. The survey was administered to a web-enabled panel of households constructed to be representative of the U.S. population. Results suggest that WTP

[8] Two additional CE-based species valuation studies have been conducted in Canada (Rudd 2009; Boxall et al. 2012).

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values are highest for recovering whale species and lowest for recovering the two salmon populations, and that WTP values are significantly different between many, but not all, paired species comparisons. Using a subset of these eight species (Puget Sound Chinook salmon, Hawaiian monk seal, and smalltooth sawfish), Lew and Wallmo (2010) test for scope sensitivity, finding that respondents are generally willing to pay more to protect more species and to attain greater improvements in the status of the species.

In a fourth study, Cameron et al. (2012) estimated WTP values for Klamath River fish species in a study that sampled at three geographic strata: households in the Klamath River basin area, California and Oregon households (excluding the Klamath river basin area), and U.S. households (excluding California and Oregon). The study estimated values for increasing the populations of wild Chinook salmon and steelhead trout and values for reducing the risk of extinction for shortnose and Lost river suckers and coho salmon. In addition, the study examined two CE methodological issues including learning and ordering effects.[9] Estimated WTP values varied depending on the geographic strata; however, values for reducing extinction risks for coho salmon were generally larger than extinction risk reductions for the two sucker populations or population increases for Chinook salmon and steelhead trout. Results suggest no evidence of learning or ordering effects in the survey.

Additionally, several CE studies are currently underway at the National Marine Fisheries Service. One is a national level study conducted using a web-enabled panel of U.S. households that will estimate WTP values for recovering or downlisting eight ESA-listed species–the hawksbill sea turtle, southern resident killer whale, humpback whale, southern California steelhead, central California coast Coho salmon, black abalone, Elkhorn coral, and Johnson's seagrass. The second study, conducted as a mail survey of Alaska households, will estimate WTP values for reducing the extinction risk of the Cook Inlet Beluga whale and improving its ESA status. The study will also examine differences in WTP held by rural and urban Alaska households.

During the last 10 years CEs have been increasingly used for the non-market valuation of threatened and endangered marine species. As noted above, the approach can offer more flexibility than the traditional CV method, making it easier to evaluate a suite of policy alternatives. Additionally, in contrast to many of the early CV studies which were local or regional in scope and utilized relatively small sample sizes, recent CE studies have been conducted with larger samples on a wider geographic scope (e.g. a national sample and/or strata within a national sample). This may be appealing for two primary reasons. First, WTP values derived from larger samples are often more robust and thus may be more appropriate to apply to a target population than values derived from smaller-sample studies. Second, values from well-designed national-

[9] Learning effects may occur when the repeated questioning format of CEs enables respondents to adopt a heuristic or strategy for answering choice questions. Ordering effects may occur when the sequence of information presented in a survey instrument affects choice question responses. Cameron et al. (2012) tested whether the ordering of the introduction of human uses of the Klamath River impacted responses and whether answering one or two choice questions impacted responses. No evidence was found of either effect.

level studies reflect the preferences of the U.S. population as a whole and can thus be applied in analyses at the federal level. This is important, as T&E species are federally regulated resources.

In the U.S., economic information, including non-consumptive values, is excluded from the decision to list a species under the ESA, however, economic benefits and costs may be considered in the designation of critical habitat and the development of species recovery plans (CRS 2003). To date, most applications of protected species values have been through the supporting analyses required to designate critical habitat, primarily the Regulatory Impact Review Preparatory Assessments. In several of these assessments there is a qualitative discussion of species values and the introduction of a benefits transfer value; however to date species values have not been used directly in a benefit-cost analysis (for a specific example see Final RIR/4(b)(2) Preparatory Assessment/FRFA for the Critical Habitat Designation of Cook Inlet Beluga Whale).

Outside of the ESA process, WTP values for marine protected species may be explicitly required for natural resource damage assessments conducted in response to events such as the Deepwater Horizon Oil Spill or other events that damage the marine environment. In addition, species values may be needed by analysts as U.S. ocean policy increasingly shifts toward ecosystem-based management and coastal and marine spatial planning (Interagency Ocean Policy Task Force 2009). These management frameworks require the full suite of impacts on resources, biological diversity, and ecosystems to be considered when developing policies, and non-market valuation currently provides the only option for estimating the economic value of changes to protected marine species.

## 4. NON-MARKET VALUATION IN RECREATIONAL FISHERIES

Non-market valuation related to fisheries management issues mostly arises in relation to recreational fisheries valuation. For most recreational fishing, the angler is both the producer and consumer, and thus, no market transaction is observed. Even in the case of for-hire charter or party boat recreational fishing, given the paucity of economic transaction data in that sector, most studies have relied on non-market valuation techniques for that component of the fishery[10]. Magnuson-Stevens Fishery Conservation and Management Act[11] (MSFCMA) requires the setting of a total allowable catch for different fish stocks, which may then be allocated between commercial and recreational sectors. Any major change in quota or allocation would require a benefit-cost analysis as part of a regulatory impact review, and thus, a need to value marginal or incremental changes in recreational fishing values (Executive Order 12866). Additionally, National Standard 5 under the MSFCMA requires that efficiency be considered in conservation and management, thus, requiring measures of market and non-market marginal economic value.

[10] Carter and Liese (2010) use data on charter fees in a hedonic analysis to value attributes of charterboat recreational fishing trips.
[11] http://www.nmfs.noaa.gov/sfa/magact/MSA_Amended_2007%20.pdf


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Research on recreational fishing was facilitated by the implementation in 1979 of the Marine Recreational Fisheries Statistics Survey (MRFSS, now replaced by the Marine Recreation Information Program (MRIP))[12]. Freeman's (1995) review of recreational fisheries economic studies found 21 studies providing estimates of access value per trip, access value per year, or some measure of the value of the change in angler expected catch rate. The studies were varied in the methodologies employed including travel cost models, single equation random utility models (RUM), nested random utility models, and contingent valuation approaches. Under a Cooperative Agreement with the Environmental Protection Agency and NMFS, an add-on economic survey to the MRFSS survey was piloted in the mid and south Atlantic recreational fisheries (McConnell and Strand 1994). The add-on survey captured individual angler characteristics that allowed incorporation of heterogeneity among anglers in their catch expectations that could then be employed in random utility models (McConnell et al. 1995). In 1996, NMFS began routinely implementing add-on revealed preference economic surveys to the MRFSS survey, which spurred greater application of random utility models. Table 2 provides the years that surveys were conducted by region.

*Table 2*. NMFS Add-on Socio-economic Surveys for Recreational Fishing by Year

| **Region** | **Revealed Preference Surveys** | **Stated Preference Surveys** |
|---|---|---|
| Alaska | 2002, 2004, 2006, 2011 | 2002, 2007, 2011 |
| Atlantic Highly Migratory Species | 2011 | |
| Caribbean | 2003/2004 | 2003/2004 |
| Northeast | 1996, 1997, 1999, 2000, 2006, 2009, 2011 | 2000, 2009, 2010, 2012 |
| Pacific Islands | 2006, 2011 | 2006 |
| Southeast | 1999, 2000, 2003/2004, 2006, 2009, 2011 | 2003/2004, 2009 |
| West Coast | 1998, 2001, 2006, 2009 (CA), 2011 | 2006 (WA, OR), 2009 (CA), 2013 (WA) |

Source: NMFS http://www.st.nmfs.noaa.gov/economics/fisheries/recreational/valuation-studies/index

The regular collection of economic data to coincide with MRFSS/MRIP data led to an expansion in the application of RUM models and resulted in region specific studies on the value of access to recreational fishing, as well as estimates on values related to changing catch rates (Hicks et al. 2000). Examples include New England and the mid-Atlantic (Hicks et al. 1999) and the southeast (Haab et al. 2000). The ready availability of MRFSS and the economic add-on data also stimulated broader application of random utility models to look, for instance, at how

[12] http://www.st.nmfs.noaa.gov/recreational-fisheries/index

environmental factors affect recreational fishing welfare such as via water quality (Lipton and Hicks 2003; Massey et al. 2006) and habitat (Hicks et al. 2004).

While application of random utility models to recreational fishing valuation greatly enhanced NOAA's capability to conduct benefit-cost analyses related to recreational fishing, the reliance on revealed preference limits the ability to forecast behavioral and welfare changes outside of the range of observations. Since recreational harvest quotas are typically managed by size limits and individual fishermen bag limits, there may not be fishing observations under the restrictions being considered. Stated preference choice experiments (SPCE), discussed above in the Protected Resources section, provide the flexibility to examine the impact of proposed regulations outside of what can be observed in revealed preference studies. NMFS started employing stated preference choice experiments (Table 2) to examine bag and size limits in the mid-Atlantic summer flounder fishery (Hicks 2002). Lew and Larson (2012) apply SPCE to value recreational fishing in Alaska among both residents and non-residents, and Lew and Seung (2010) applied the SPCE approach to estimate change in participation in fishing due to a variety of bag limit changes across three species and use these results in a general equilibrium model of the regional economy. Carter and Liese (2012) used a SPCE to examine preferences for catching and keeping fish versus releasing them alive in Gulf of Mexico recreational fishing. Similarly, Lew and Larson (2014) used SPCE to estimate the value of catch and keep versus catch and release recreational fishing trips in Alaska.

While the above studies on recreational fishing non-market values have informed fishery management decisions, results from a SPCE on New England recreational fishing has been incorporated directly into the stock assessment utilized by the New England Fishery Management Council while managing New England groundfish[13]. The Bio-economic Length Age Structured Tool (BLAST) utilizes angler preferences from choice experiments (Jarvis 2011) to simulate recreational fishing under varying stock conditions and size and bag limits to predict the angler impact on the fishing stock as well as resulting welfare impacts.

Both random utility models and choice experiments can take advantage of emerging technologies to sample anglers through hand held mobile electronic devices such as smartphones and tablets. For random utility modeling, location capture capabilities of these devices could help address the current limitation where intercept sampling does not capture the actual location where fishing occurs, and thus, misses a significant part of the travel cost and angler behavior during the fishing trip. Choice experiments on these devices can be used to explore the many facets that constitute the value of a recreational fishing trip beyond catch rates, bag and size limits. Care will need to be taken to recognize and adjust for biases in the angler sample and due to the self-selecting nature of these technologies.

[13] Personal communication, Min-Yang Lee and Scott Steinback, Northeast Fisheries Science Center.


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## 5. NON-MARKET VALUATION IN COASTAL MANAGEMENT

The coastal zone of the United States contains a multitude of competing social and economic forces, each based on individual or communal preferences and goals which are often incompatible or mutually exclusive. Ideally, the conflicts that arise through these dynamics would be resolved with solutions that maximize social welfare; however, due to an emphasis on market value or individual desires, the actual outcomes neither maximize social welfare nor provide for sustainable coastal and ocean resource use. One of the main reasons for this is the underutilization of non-market values in policy creation and decision-making

Historically, the use of economic values to inform coastal management has emphasized the value of activities that rely on natural resources, but the connection between the value and the health and function of those resources were not always incorporated in the analysis. These value estimates did not always account for situations in which gains in value were made at the expense of natural capital. However, the trend toward using the ecosystem services framework has placed increased emphasis on values that are reliant on healthy and functioning ecosystems.

There are numerous potential applications for non-market values in the coastal management realm. These might include those related to specific management structures, questions related to particular habitat types, and important issues that must be addressed. Here, we focus on the use of non-market valuation related to specific habitats, including beaches and coral reefs; and related broadly to Marine Protected Areas, and specifically to National Marine Sanctuaries.

### 5.1 Coastal Habitats: Coral Reefs and Beaches

Because there are coastal management issues that are unique to specific habitats, it is often helpful to examine them from a habitat perspective. NOAA's purview includes the breadth of coastal habitats, but here we highlight two: coral reefs and beaches.

Coral reefs provide many valuable services to people. Among these are food, recreation, storm protection and cultural importance. In the United States, coral reefs can be found in the Western Atlantic, the Caribbean, Hawai'i and the Pacific territories; as well as in over 100 countries around the world. The threat to this habitat is significant and due to a variety of factors, including climate change impacts, unsustainable fishing and land-based pollution[14]. In order to protect these resources, it is crucial to understand the magnitude of their value, to whom they are valuable, and the relative value of the activities that contribute to their damage.

Estimates of the economic value of coral reefs are used in many of the ways described above. Additionally, the values of these resources are important in order to demonstrate why agencies should invest in research to understand the dynamics of these resources and how they are being damaged. This information also helps incorporate coral management into the local decision-making processes. In coral reef valuation, the connection between the economic value and the ecological endpoint is a key component, making it important for economists to work with biophysical scientists to ensure this connection is appropriately incorporated into the analysis. It

[14] "Value of Coral Ecosystems." NOAA's Coral Reef Conservation Program: Values. May 13, 2011. Accessed September 24, 2014. http://coralreef.noaa.gov/aboutcorals/values/.

is vital that the contingent context used in the analysis is based on an analysis of the condition and trends of the coral reefs.[15]

In NOAA, the Coral Reef Conservation Program (CRCP) supports the conservation of coral reefs in the states and territories that contain coral reefs. The CRCP also supports valuation of coral reefs in order to answer the following questions (Edwards 2013):

- How much are coral reefs worth to society?
- How much do people care about coral ecosystems?
- Can we demonstrate the value of these unique ecosystems and account for what we stand to lose if they are irreparably damaged?

The CRCP has been conducting non-market valuation studies of domestic coral reef systems since 2001. , Table 3 summarizes these findings by study and year. The variability in context, methodology and approach in these studies do not allow for aggregation and make comparability of these resulting value estimates difficult.

*Table 3.* Economic Values of US Coral Reef Jurisdictions (source: Edwards, 2013)

| **Location** | **Study Year** | **Present Value (2012$ Million/Year)** |
|---|---|---:|
| Florida | 2001 | 324 |
| Hawaii | 2002 | 455 |
| American Samoa | 2004 | 11 |
| CNMI – Saipan | 2006 | 68 |
| Guam | 2007 | 150 |
| Puerto Rico | 2008 | 1,161 |
| US Virgin Islands | 2011 | 210 |

Regarding beaches, beach visitation is one of the most popular uses of coastal resources in the United States. Between 1999 and 2000, about 30 percent of the civilian non-institutionalized population visited a saltwater beach. This translated to more than 61 million visitors nation-wide (Leeworthy and Wiley, 2001). Like other applications of non-market valuation, there are numerous contexts for which estimates of value are needed. Because research related to beach visitation varies according to its intended use and to whether or not behavioral changes can be directly related to changes in resource quality such as water quality (Smith and Desvousges, 1986), the approaches and methodology used as well as the specific beach attributes included will also vary.

The economic value of beaches is potentially derived from a variety of sources; some natural and some manmade such as parking access, bathrooms, etc.. How much each contributes to the value varies by location, by the preferences of beach visitors, and by the other attributes associated with the beach. The attributes that can potentially influence the value of beach recreation can be numerous. One ongoing effort[16] has been compiling data on 50 beach attributes for a period of 23 years.

One of the most important attributes to consider when conducting beach valuation is water quality. Water quality supports a wide variety of activities and values, but from a beach

[15] Edwards, Peter E. T. 2014. Personal Communication.
[16] Leatherman, S., 2013. Dr. Beach: America's Foremost Beach Expert. Top 10 Beaches. Accessed September 24, 2014. http://www.drbeach.org/50criteria.html


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perspective, it is a critical attribute. Water quality can cause closure of beaches or decrease the value of the beach experience. This work is complicated by the ability to visit a different beach (substitution) or partake in an activity that doesn't rely on water quality, such as sunbathing (Hanemann et al. 2005; Larson and Lew 2005). The treatment of these issues is important to the reliability of study results.

The Hanemann et al. (2005) study looked at several scenarios, including beach closures for one day, one month, and one season; as well as water quality degradation without closure and a water quality improvement. They concluded that even minor changes in water quality can lead to significant changes in value. For example a one day closure at Huntington Beach would result in a loss of value of more than $100,000. A water quality decline at Zuma Beach (a relatively clean beach) would lead to a loss of value of over $5 million.

### 5.2 Marine Protected Areas

One of the most direct ways that natural resource management agencies conserve coastal resources is through the designation of protected areas. In the United States, Marine Protected Areas (MPA) are broadly defined as "any area of the marine environment that has been reserved by federal, state, territorial, tribal, or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein.[17]" In the United States there are over 1,700 MPAs put in place to support a wide range of objectives (NOAA and U.S. DOI 2008).

Measuring the value of establishing an MPA can be challenging for a variety of reasons. Many of the values that are important to society as a whole, such as benefits related to biodiversity or climate regulation, may not be included because of the difficulty of their estimation or because they are not foremost in the minds of local stakeholders. In many cases the range of benefits can also be a challenge. Valuing the full range of services provided by an MPA is not generally feasible, and choosing which services to value will invariably not tell the whole story. Benefits may vary with the specified size of the MPA (Wallmo and Edwards 2008). Ultimately, the choice of what benefits to include in an analysis will depend on a variety of factors, including difficulty of estimation, whether or not a benefit is rival or excludable, and whether a loss in benefits would be reversible (Dixon and Sherman 1990).

Ideally these choices are informed by the goals for establishing the MPA, or those relating to changes in the regulations associated with it. Using these as the criteria for which benefits to include, the valuation effort may not be comprehensive, but it will be effective in answering the questions related to management actions, and will ultimately lead to actions that take ecosystem health and human well-being into account to the greatest extent possible.

In order to conduct non-market valuation in a way that will account for the attributes that are most important to the potential management actions in question, the valuation method chosen is important. Some methods lend themselves better to accurately capturing changes in particular attributes. Recent efforts by the Office of National Marine Sanctuaries highlight this point.

[17] Executive Order 13158 of May 26, 2000. "Marine Protected Areas". *Federal Register. Vol. 65, No. 105 Wednesday, May 31, 2000*

### 5.3 National Marine Sanctuaries

NOAA's Office of National Marine Sanctuaries (ONMS) is the trustee for a large network of 14 marine protected areas that span the United States and its territories. These national marine sanctuaries include more than 170,000 square miles of marine and Great Lakes waters[18].

When a national marine sanctuary is established or its regulations revised, the ONMS conducts valuation studies to estimate the benefits and costs of the alternative actions. Historically, the sanctuary program has utilized a wide range of methodological approaches, including the Travel Cost Method or Contingent Valuation Method to estimate the non-market economic value of sanctuary resources, but these approaches have not been adequate in terms of their ability to capture the effects of changes in attributes. Recently, ONMS staff has been relying on stated preference choice experiments, which provides more flexibility to describe changes in attributes associated with management actions[19].

An example of the valuation work currently taking place in ONMS is a conjoint attribute approach project at the Olympic Coast National Marine Sanctuary. Part of the Marine Spatial Planning process in Washington State, this ongoing project looks at changes in value associated with changes in the quality of a series of attributes, based on the spatial distribution of use.

The National Marine Sanctuaries are also using ecosystem service valuation in their Sanctuary Condition Reports. Because valuation research across all sanctuaries is challenging from a budget and capacity perspective, the sanctuary program utilizes value indicators, such as park visitation and boat registration paired with ecological monitoring data to determine if ecological indicators are consistent with ecosystem service indicators. This also allows researchers to determine whether or not natural capital is being depleted for short term economic gains[20].

## 6. SUMMARY AND CONCLUSIONS

The demand for information on non-market values for marine and coastal resources continues to grow. More pressures are being placed on these systems due to population growth and more intensive and often competing uses of these resources. An understanding of the value of human activities - both those that impact and benefit from ecosystem health - is vital to effectively manage coastal resources. Additionally, the connection between the value of these activities and ecosystem health will require continued and expanded use of non-market valuation in such a way that it takes into account both ecosystem health and social welfare. In the absence of a consistent understanding of how healthy ecosystems contribute to societal value, the costs of activities that may degrade these resources will not be taken into account in management actions or public policy (Daily 2012). Policymakers, at all levels, need a common metric to evaluate trade-offs that accompany their decisions.

[18] National Oceanic and Atmospheric Administration. 2014. About the National Marine Sanctuaries. Accessed September 26, 2014. http://sanctuaries.noaa.gov/about/welcome.html

[19] Leeworthy, Vernon R. 2014. Personal Communication.

[20] Ibid


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The descriptions above are not intended to be a comprehensive overview of non-market valuation studies in the literature or even conducted by NOAA, but are representative of the work that has evolved over the past 10-15 years. There has been an expansion building from simple travel cost models and contingent valuation studies towards more robust nested random utility models and choice experiments, as well as a blending of the two. There is also a small, but growing effort to incorporate non-market ecosystem service values into large scale ecosystem modeling effort such as in the NOAA Integrated Ecosystem Assessment (IEA) Program[21].

While there has been an increasing trend in the use of non-market values in coastal management, many challenges remain. First and foremost among these are the high cost and time required to estimate de novo non-market values. This challenge is often one that discourages a more detailed dialogue on the benefits of having consistent non-market values when making coastal management decisions. Another challenge is related to capacity. It is not uncommon for coastal management entities to have little if any expertise in economic analysis; and although this work can be conducted by external economists, it is necessary to have sufficient internal expertise to know the right questions to ask, how to ask them, and how the valuation estimates will be applied in a policy or decision context. Finally, the requirements for non-market values in coastal management legislation are inconsistent and often only call for estimates of the negative impacts of management actions. Because estimating the economic benefits of management is often not a requirement, there are many instances for which it is not conducted. This can lead to instances in which only the costs of a management action are known, which can decrease the likelihood of those projects taking place.[22]

A continuing challenge is determining the appropriate use of non-market value in decision-making. While the Blue Ribbon Panel found that well conducted contingent valuation (and by extension stated preference methods generally) provided “useful information” on the magnitude of welfare gains or losses, the question remains, how useful is it and how should it be used? In the case of damage assessment, that may continue to be determined within the courts. In fisheries, regional Fisheries Management Councils, guided by their Scientific and Statistical Committees, will need to decide how to use this information for controversial issues such as allocation of quota among competing users. States and local authorities will be challenged to utilize information on non-market values for coastal management and planning issues. Rather than debate the merits of different applications of non-market valuation within the context of particular controversial decisions, it would be useful for an independent panel, similar in structure to the NOAA Blue Ribbon Panel, to provide guidance on the appropriate use and application of non-market valuation in coastal and marine resource decision-making.

[21] http://www.noaa.gov/iea/

[22] Leeworthy, Vernon R., 2014. Personal Communication.

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