THE ECONOMICS OF PROTECTED MARINE SPECIES: CONCEPTS IN RESEARCH AND MANAGEMENT

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THE ECONOMICS OF PROTECTED MARINE SPECIES: CONCEPTS IN RESEARCH AND MANAGEMENT

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Humpback whale - Megaptera novaeangliae - breaching Photo by NOAA Photo Library, available at: http://www.photolib.noaa.gov/bigs/sanc0601.jpg

Protected marine species have populations that are depleted, decreasing, or are at-risk of extinction or local extirpation. As of 2015 The International Union for the Conservation of Nature, a global environmental organization, lists approximately 737 marine species worldwide that are considered at risk of extinction. Many are provided legal protection through national laws requiring research and management measures aimed at recovering and maintaining the species at a sustainable population level. Integral to the policy decision process involving the management and recovery of marine species is the consideration of trade-offs between the economic and ecological costs and benefits of protection. This suggests that economics, at its core the study of trade-offs, has a significant role.

In the U.S. a somewhat traditional use of economics in protected species research and management has involved cost minimization or cost-effectiveness analyses to help select or prioritize conservation actions. Economic research has also provided estimates of public non-market benefits of recovering species, which can be used in larger management frameworks such as ecosystem based management and coastal and marine spatial planning. Inherent in much of this research, however, are complex biological and ecological relationships in which varying degrees of scientific uncertainty are present. Addressing this type of uncertainty can affect the economic outcomes related to protected species. For example, recent work suggests that increasing scientific precision in biological sampling and models can greatly affect the magnitude of economic benefits to commercial fisheries, while other research suggests that public non-market benefits of species recovery are sensitive to uncertainty about baseline population estimates.

Previous research has illustrated the importance of understanding the biological, ecological, and economic aspects of protected species management and recovery. In this research topic we synthesize current protected marine species economic research and expand the discussion on present and future challenges related to protected species economics. The series of manuscripts brings together an array of prominent researchers and advances our understanding of the ecological and economic aspects of managing and recovering protected marine species.

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EDITORIAL
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Editorial: The Economics of Protected Marine Species: Concepts in Research and Management

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The Editorial on the Research Topic

The Economics of Protected Marine Species: Concepts in Research and Management

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Protected marine species have populations that are depleted, decreasing, or are at-risk of extinction or local extirpation. As of 2015, the International Union for the Conservation of Nature, a global environmental organization, listed approximately 737 marine species worldwide that are considered endangered and vulnerable to extinction, noting that to date only a fraction of the world's marine species have been assessed. Many of these species are provided legal protection through national laws that require research and management measures aimed at recovering and maintaining the species at a sustainable population level. In the U.S. for example, the Endangered Species Act and the Marine Mammal Protection Act provide protection for 143 marine species. Integral to the policy decision process involving the management and recovery of marine species is the consideration of the economic and ecological costs and benefits of protection. This suggests that economics, at its core the study of tradeoffs, has a significant role. First, economics provides a decision theoretic framework for considering all benefits and costs to society associated with policies and management actions aimed at protecting species. In addition, economics contributes to the development of incentive-based management tools (such as property rights-based policy instruments) and can provide a framework for evaluating them against more traditional command-and-control tools (such as closures). The articles in this Research Topic identify various contributions that economics can make to protected species research and management.

In the opening article Lent outlines key reasons marine resource managers should consider economics in protected species research and management and focuses on two specific contributions: economic valuation and market-based management tools. Economic valuation tools provide a way to measure the benefits of things such as protecting marine species, their habitat, and the services they provide. This enables them to be formally included in regulatory analyses, environmental mitigation cases, and marine management approaches in which the evaluation of trade-offs is inherent. Market-based management tools such as catch share programs, permit buy-backs, conservation leasing, etc., address fundamental drivers of human behavior and provide flexibility and incentives for innovation which traditional tools (e.g., command and control) often lack. Such incentives often lead to maintaining profitability while producing desired protected species management results.

The next set of articles in the Research Topic explores economic valuation. The articles begin with Lew, who provides a comprehensive literature review of marine species valuation estimates and discusses their potential use in management and policy. This is followed by Wallmo and Lew who present an empirical application valuing the recovery of eight threatened and endangered marine species in the U.S. and examine the effect of the sampling scope and scale on benefit measures. Forbes et al. provides another example of an empirical application valuing alternative management actions for recovering a representative species of Pacific rockfish. Weber estimates the benefits of recovering salmon species in the Pacific Northwest and illustrates some of the challenges with the use of secondary data in economic valuation. Cole and Moksnes use a production function approach to quantify links between three eelgrass functions (habitat for fish, carbon, and nitrogen uptake) and economic goods in Sweden, thus relating ecosystem services such as commercial fisheries, carbon, and nitrogen sequestration (outputs) to marine habitat (inputs) to estimate the value of protecting marine habitat. Finally, Cardenas and Lew investigate issues related to funding conservation programs for endangered species. They provide insights into factors affecting tourists' willingness to contribute monetarily toward the conservation of two marine protected species, green sea turtles and hammerhead sharks.

The following set of articles begin with Innes et al., who review the literature on market-based management tools used to promote conservation while reducing negative impacts associated with commercial fishing. In Bisack and Das non-compliance issues with management and regulations are explored. They examine economic and normative (e.g., moral, social, legitimacy) factors that may affect compliance behavior by developing and estimating an empirical model to explain a fisherman's compliance decision with respect to marine mammal regulations. The final article on management tools by Bisack and Magnusson discusses the need for ex-post analyses of policy instruments used to conserve protected marine species. They assess policy instruments' expected efficacy with respect to multiple criteria, including biological, economic, socio-normative, and longevity objectives. The Research Topic concludes with an article by Speir et al. who use dynamic optimization techniques to determine the optimal timing for dam removal in a large-scale restoration project involving endangered salmon. The study compliments the focus areas of the other articles by underscoring the importance of addressing uncertainty when modeling economic and ecological costs.

The collection of articles in the Research Topic illustrates important concepts, gaps, and challenges in protected species economics, serving as a first step toward improving our knowledge base and raising critical dimensions for future research. Efforts such as this can facilitate collaboration among scientists and even across disciplines, offering a valuable tool to enhance protected species research and management.

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Conservation benefits of an interdisciplinary approach to marine mammal science

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Protected resource economists can greatly enhance the science and conservation of marine mammals, however such contributions are often hampered by a lack of understanding of the role of natural resource economics on behalf of more traditional marine mammal scientists. The three major threats to marine mammals—fishery bycatch, increasing underwater sound, and climate change could be more effectively addressed with an interdisciplinary approach that includes the full valuation of costs and benefits to society. Better management of these threats can be beneficial to humans as well as marine mammals.

Keywords: marine mammal conservation, non-market valuation, natural resource economics, externalities, protected species policy

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Marine mammal scientists tend to be rather wary of economists and may have a limited understanding of their role in marine science and conservation. This is likely due to the general belief that economists are "all about business and profits" rather than environmental conservation. Such misunderstanding is unfortunate because natural resource economics, a discipline that is somewhat more recognized in the context of terrestrial conservation, addresses the full valuation of wildlife (including marine mammals) to society (Krutilla, 1967). Economists embrace complete accountability for the costs imposed on marine mammals and ecosystems by human activities, whether or not these values derive from direct exploitation or use. Such a comprehensive approach can improve policy design, stimulate public interest, facilitate better-informed decision-making, and provide stronger incentives for compliance with regulatory measures. A more concerted effort at dialogue and collaboration between marine mammal scientists and resource economists would strengthen the case for conservation and increase policy effectiveness as well as equity.

In addition to considering all costs and benefits to society, natural resource economists look for efficiencies in regulatory policy, such as approaches that incentivize environmentally beneficial decisions rather than force them through government top-down, "command and control." For example, in addition to technological fixes to carbon emissions, economists would also assess whether market-based approaches such as cap-and-trade systems or a carbon tax would yield higher net benefits. While there is a lively debate in the economics literature over the relative merits of the two market-based alternatives (Goulder and Schein, 2013), both of these approaches provide flexibility and incentive for innovation, as the firms figure out the technological fixes needed to operate profitably given the tax or the costs of acquiring emissions credits.

Natural resource economists also add to the quality of analyses of policy alternatives by including estimates of the non-market value of wild organisms and ecosystems to society. These estimates can be based on the non-consumptive uses (e.g., whale watching) and non-use values (e.g., existence and bequest values)[1]. In some cases, economic valuation analyses focus on population-level values as opposed to individual species values. For example, in a study on the value to Americans of improving the status of North Atlantic right whales, Wallmo and Lew (2012) estimate that households are willing to pay $71.62 on average for removal of the species from the endangered species list. Estimating such values is a first step; incorporating them into the analyses of policy alternatives that inform decisions is the next, and sometimes more challenging, step in the process. Perhaps understandably, non-economists are often uncomfortable with the concept of putting a dollar value on a "charismatic megafauna" such as a whale through survey methods soliciting stated preferences. However, not all existence values are estimated through stated preferences, nor are all economic analyses predicated on existence value estimates.

Economics can also be a great asset in designing effective policy even when there is not a possibility of including values of marine mammals in the analysis. For example, if a regulation sets a limit on the number of animals affected by a given activity, a cost-effectiveness analysis would identify the least-cost approach to satisfying this objective. This approach would "release" financial resources for needed conservation measures elsewhere that would otherwise not be available.

Another pertinent point is that economists generally prefer private, negotiated solutions to adverse environmental impacts, rather than top-down, regulatory solutions. Such an approach can lower conservation costs, strengthen incentives to meet conservation objectives and compliance, and create an environment whereby innovative solutions are developed that might otherwise never occur. Negotiated solutions to externalities are particularly pertinent to cases in which the parties can be clearly identified and for which there are no public goods. Examples include the Morro Bay, California, groundfish fishery, in which an NGO purchased trawler permits and subsequently made the permits available for alternative gears having fewer adverse environmental impacts (and also achieving the NGO's objective of a smaller scale of operation) (Gleason et al., 2013). Off central California in the approaches to the ports of Long Beach and Los Angeles, a voluntary, informal, non-binding agreement was negotiated among interested parties to achieve the goals of reducing ship speed and minimizing transit time in state waters using low emission fuel and therefore minimizing carbon emissions and the probability of marine mammal vessel strike injury and mortality[2]. It is interesting to note that to some extent, existing regulatory bodies such as the regional fishery management councils and marine mammal take reduction teams seek to establish this same approach of negotiation and dialogue in addressing environmental issues, admittedly with varying success.

[1]Existence value refers to willingness of individuals to pay for the conservation of an environmental good, without being able to use or even see that good. The value can be based on altruism, intergenerational bequest value, or intrinsic worth (Blomquist and Whitehead, 1995).

[2]In the case of Santa Barbara, shippers received partial compensation for the vessel speed reduction (http://thinkprogress.org/climate/2014/08/05/3467453/ships-slow-down-to-protect-whales/) providing an interesting example of Payments for Ecosystem Services (PES).

Consider in turn the primary threats to the survival of marine mammals, notably fishery bycatch, climate change, and anthropogenic sound (shipping, energy exploration and development, military, construction) and what natural resource economists have to contribute to addressing each of these problems.

Bycatch (including entanglement in discarded or lost fishing gear) is the greatest direct threat to marine mammals, with estimates of annual mortality in excess of 650,000 marine mammals globally (Read et al., 2006). Economists approach bycatch as an unintended adverse impact of fishing for target species (i.e., a negative externality) that is not factored into the costs of fishing, and therefore not reflected in the price of seafood. Because the price of seafood is too low (it does not include the "costs" of marine mammal mortality), fishing vessel operators, seafood marketers in the supply chain, and consumers are unaware of, and do not bear, the full costs of their activities and therefore over-produce and over-consume both targeted and bycaught species. Over the longer term, imposing these higher costs on fishing operations should create "dynamic" incentives that induce technological change that reduces marine mammal bycatch. Traditional command-and-control bycatch measures could include mandatory modifications of gear or gear deployment (such as pingers on gillnets in the New England groundfish fishery), or time/area closures to reduce the overall level of bycatch in a fishery. In contrast, examples of incentivizing approaches championed by economists might include per-vessel allocations of tradable bycatch quotas or bycatch credits, resulting in trade among operators such that the vessels that are most efficient in reducing bycatch end up doing most of the fishing—thus most efficiently reducing impact on marine mammals. In evaluating the various alternative regulatory and negotiated measures for mitigating marine mammal bycatch, economists would include not just the costs to fishery operations but also estimates of the benefits, to the ecosystem and to the public, of reductions in marine mammal mortality. An interesting example of an incentivized approach negotiated by an industry group in order to meet regulatory standards is found in one of the world's largest fisheries, notably the Alaska pollock fishery. The member companies in the At Sea Processors Association implemented their own "Chinook Salmon Incentive Plan and Agreement," which includes identification of "rolling hot spots" based on vessel reporting and features stricter provisions on fishing vessels with low performance in avoiding Chinook bycatch[3].

Climate change is having profound and likely irreversible impacts on marine mammals through modification of habitat (particularly in the polar regions), such as reductions in sea ice and prey availability, altered pathogen survival and transmission, ocean acidification, and other ecosystem shifts. Mitigation of climate change requires measures that are pervasive and complex, with financial impacts on nearly all human activities and, if enacted, benefits to the entire global ecosystem and all species, including humans. As with fishing, and given the

[3]See description of the program at http://www.atsea.org/doc/Salmon%20Bycatch%20Poster%20FINAL.pdf

conclusion that climate change is driven largely by increasing carbon emissions [International Panel on Climate Change (IPCC), 2014], economists would argue that by ignoring the uncosted negative impacts (i.e., negative externalities) associated with carbon emissions, prices are too low—not just for energy products but for all goods and services that use energy for production, transportation, and consumption. Incorporating the costs of these negative externalities into business decisions (via taxes or a cap and trade system) would result in higher costs and prices, and lower levels of production and consumption. Over the longer term, these higher costs would be expected to create "dynamic" incentives that induce technological change that mitigates climate change and its impact on marine mammals—and the rest of marine and terrestrial ecosystems. In addition, while economists generally oppose subsidies (and taxes) as market-distorting interventions, when a "good" is being produced that does not have a market value (e.g., cleaner air), there are economic arguments to be made for public funding of activities such as development and adoption of technological change that reduces energy consumption.

Marine mammals use sound for virtually everything they do, which includes communicating and interacting with conspecifics, avoiding predators, locating prey, and navigating in the marine environment (Marine Mammal Commission, 2007). In addition to the overall masking effects of an increase in ambient sound levels on communication, acute anthropogenic sounds at a high enough sound pressure level can result in temporary or permanent loss of hearing, physical injury, behavioral modification, and stress impacts on the health and survivability of marine mammals. Human activities such as shipping, offshore energy development (seismic surveys, pile-driving, drilling, etc.), and military operations generate potentially harmful underwater noise—at no cost to those carrying out the activities. Current command-and-control measures for addressing noise include mitigation efforts such as slow ramp-up of sound sources, use of trained observers to monitor "safety" zones around the sound sites, and shutting down the sound source when marine mammals are sighted nearby. In contrast, if the approach were to incorporate the true costs of these externalities into private sector decisions[4] on whether and how to conduct such activities, there would be higher costs and prices for the products and services (such as shipping fees and energy prices) and therefore lower levels of production. Again, over a longer time period, these higher costs (e.g., through a tax on noise emission) would provide incentives for technological change that reduces the impacts on marine mammals and other marine organisms from various underwater sound sources.

Marine mammal scientists and managers have much to gain by collaborating with their colleagues in natural resource economics. At the same time, economists need to focus on developing relatively simple and easily understood economic parameters that can be part of the information made available to policy makers and the public. Examples include measures of the non-market value of marine mammals or willingness to pay for population recovery, and the costs of the more common externalities, such as carbon emissions or noise. Full comparative analyses of alternative measures for mitigating impacts on marine mammals, including assignment of values to all direct and indirect costs and benefits, should help inform public debate and decision-makers and lead to more rational policies and greater incentives for compliance. The synergies from cross-disciplinary collaboration can enhance the quality and quantity of information available—to decision makers who have the responsibility for marine mammal conservation and to the public who must be part of the process. With so many challenges in trying to address threats to marine mammals, an interdisciplinary effort is needed to save these animals and their marine environment, and ultimately, ourselves.

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[4]Command and control policies are likely the most effective approach for the case of military exercises.

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REVIEW
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Willingness to Pay for Threatened and Endangered Marine Species: A Review of the Literature and Prospects for Policy Use

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Non-market valuation methods have been employed to estimate willingness to pay for numerous threatened, endangered, and rare (TER) species over the past few decades. While most of these efforts have focused on terrestrial species, over 30 published studies have been conducted to measure economic values associated with the preservation, protection, and enhancement of scores of marine species. In this paper, this literature is reviewed and assessed, and an evaluation of the suitability of existing TER species values as inputs for the analysis of marine and coastal policies, and the prospects and challenges for improving them, are discussed. The published literature is found to suffer from coverage issues, both geographical and in terms of species types. It includes stated preference valuation studies focused on marine species only in developed countries (United States, Canada, Australia, United Kingdom, Spain, and Greece), with the highest concentration of studies occurring in the United States. The species valued primarily can be classified as charismatic megafauna—seals and sea lions, whales, and sea turtles—plus well-known fish species, like salmon. Only a small handful of lesser known species are included among those valued to date. Species value estimates were as much as $356 (2013 U.S. dollars), but differed in the frequency of payments (e.g., lump sum vs. annual), the entity paying (e.g., household, resident, or visitor), and the specific good being valued (e.g., species preservation or a type of enhancement). Potential sources of errors arising from the use of these values for policy analyses, and the temporal stability of them, provide reasons to be cautious in their application. Nevertheless, several trends in the literature appear to provide reasons to be optimistic about the literature, particularly the recent expansion of types of species valued and more policy-relevant values.

Keywords: threatened and endangered species, stated preference methods, non-market valuation, marine species, cetaceans, pinnipeds, sea turtles, willingness to pay

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INTRODUCTION

In recent decades, there has been a movement toward ecosystem-based management (EBM) approaches to managing marine and coastal resources. EBM is a central theme of the National Ocean Policy (Executive Order 13547) in the United States and in the European Union's Marine Strategic Framework Initiative (EU Directive 2013), as well as the newly-formed Intergovernmental

Platform on Biodiversity and Ecosystem Services (IPBES)[1]. EBM approaches take a holistic, systems-level approach to managing resources, one recognizing, and accounting for the interconnectedness of all parts of the ecosystem, including ecological and human components (Yaffee, 1996). The inclusion of social science inputs is recognized as a critical part of this approach, but it is also recognized as an area with significant challenges (U.S. General Accounting Office, 1994; Endter-Wada et al., 1998; Leslie and McLeod, 2007). From an economic perspective, one challenge to successfully implementing EBM in a marine context is to adequately account for the benefits and costs associated with the multitude of affected ecosystem services that are necessary to evaluate trade-offs associated with potential management actions (e.g., National Research Council, 2005; Farber et al., 2006).

This paper focuses on reviewing what is known about economic values associated with one particular component of many ocean and coastal ecosystems, namely, threatened, endangered, and rare (TER) marine species, which is the focus of this special issue. At present, there are approximately 125 marine species listed under the U.S. Endangered Species Act of 1973 (ESA). This represents about 6% of the approximately 2226 ESA listed species. The listed threatened and endangered marine species include 27 marine mammal species (e.g., whales, dolphins, sea lions, and seals), 16 sea turtle species, 57 fish species, and 24 marine invertebrate species (e.g., coral). In addition, there is one marine plant species, Johnson's seagrass, listed under the ESA. Among the ESA listed species are 38 species with habitats completely in marine or coastal waters of foreign countries. Globally, the International Union of Conservation of Nature (IUCN) has been conducting a worldwide marine species assessment since 2005 to determine the risk of extinction to all marine species[2]. Of the approximately 11,000 marine species assessed to date, about 15% have been determined to be threatened, a category that includes species that are "critically endangered," "endangered," and "vulnerable" with respect to extinction risk. These include the ESA listed marine species, plus numerous other species of marine mammals, sea turtles, fish, and sea birds.

Economic value information about TER marine species, particularly the non-market benefits associated with these species has been emphasized as a commonly missing, but critical, piece of information with respect to EBM (e.g., Millennium Ecosystem Assessment, 2005)[3]. In a fisheries policy context, for example, Sanchirico et al. (2013) illustrated how including economic values associated with protecting an endangered marine species can significantly affect policy recommendations from an economic efficiency perspective, which highlights the importance of efforts to better understand and incorporate economic values associated with TER marine species in analyses of EBM policies.

In the following, the literature on the economic benefits of TER marine species is reviewed. Although there are a number of studies in the gray literature that value TER marine species, such as government reports, working papers, and theses (e.g., Hageman, 1985; Medina et al., 2012), in this review the focus is on the published literature to ensure the reported studies have been peer-reviewed. Although there are likely numerous examples of high-quality unpublished work, and peer review is by no means uniform or uniformly high in standards, limiting the review to published peer-reviewed studies limits the scope sufficiently to allow for a fairly complete picture of the literature to form[4]. Additionally, even though other reviews of the TER species valuation literature exist (Loomis and White, 1996; Martín-López et al., 2008; Richardson and Loomis, 2009), the increased research activity in recent years is not captured by these studies. Given the alacrity with which efforts are being made to adopt EBM approaches in the United States and elsewhere, an understanding of the existing literature and prospects for its use in EBM and other policy applications is important.

This paper also discusses the suitability of existing TER species values as inputs for the analysis of marine and coastal policies and the prospects and challenges for improving them. To this end, the methods used to apply existing values from the literature in policy analyses, called benefits transfer or environmental value transfer methods (Navrud and Ready, 2007; Johnston and Rosenberger, 2010), are presented. Subsequently, TER species values are discussed in the context of their use as inputs to these methods, with a focus on identifying the prospects and challenges of using them in policy analyses using benefits transfer approaches.

The next section provides a detailed non-technical background on both the meaning and types of economic values for TER marine species in the literature and the methods typically used to generate estimates of them. This is followed by a description of the literature and assessment of the scope and breadth of extant literature. Then, the benefit transfer methods used to apply existing values from this literature to policy applications are discussed, and several challenges related to using existing TER marine species values for marine and coastal policy analyses using these methods, and the prospects for improving them, are highlighted.

ECONOMIC VALUES OF TER MARINE SPECIES

Economic values for TER marine species are estimated using non-market valuation methods. Non-market valuation methods were developed to measure the demand for, and value of, goods and services for which there is an absence of formal markets from which signals of value can be ascertained (i.e., prices). These methods generally aim to measure the total economic

[1] See http://www.ipbes.net/

[2] For details on the Global Marine Assessment Program and related programs, see www.iucn.org.

[3] TER marine species values are but one type of ecosystem value that may be of importance in evaluations of marine and coastal policies and programs. As noted in numerous places (e.g., National Research Council, 2005; The Economics of Ecosystems and Biodiversity, 2011), ecosystem values are important for making better environmental policy decisions, but also pose significant challenges to measure for the myriad ecosystem services and functions provided by the environment.

[4] There may be studies published in other languages that present economic values for TER species, but they are not reviewed here. This review only covers the English-language literature.

value (TEV) of the non-market good or service. Several economic models have been developed that show that TEV is the sum of use values, measurable by observing changes in the demand for market goods related to the environmental good or service, and nonuse values[5] that are not directly observable in the related good market (McConnell, 1983; Carson et al., 1999; Freeman et al., 2014). Use values, as the name implies, are those values associated with the use of the good or service and can be either consumptive (e.g., harvesting) or non-consumptive (e.g., wildlife viewing), while nonuse value is the value independent of any use of the good or service and generally attached to environmental goods and services that are unique or special and subject to irreversible loss or injury (Freeman et al., 2014).

Economic values associated with TER species are primarily the result of the non-consumptive values that people attribute to them. Non-consumptive value consists of non-consumptive use values such as viewing (as opposed to consumptive use values such as harvesting) and nonuse values apart from on-site active use, which are usually attributed to bequest and existence values[6].

Non-market Valuation Methods

Non-market valuation methods are typically classified into two types: revealed preference and stated preference methods. Revealed preference (RP) methods use data about people's behavior to infer the value of a non-market good or service (Herriges and Kling, 1999; Bockstael and McConnell, 2007), while stated preference (SP) methods use information provided directly from individuals, usually from carefully-constructed survey questions, that reveal their values (e.g., Bateman et al., 2002). Travel cost models and hedonic price models are examples of revealed preference approaches, while the contingent valuation method is the most well-known stated preference approach.

Since RP methods require data on people's behavior, they measure use values only and cannot measure nonuse values. Since nonuse values are generally believed to be a primary component of the TEV of TER species values, researchers generally rely on SP methods to estimate species values due to an absence of a behavioral link to these types of values. There are some exceptions, however. For example, RP methods have been employed in a few studies that value viewing benefits associated with endangered whales (Loomis et al., 2000; Shaikh and Larson, 2003; Larson and Shaikh, 2004). Still, since the TEV of a species is generally what researchers wish to value, SP methods are predominant in the literature, and therefore this review focuses on those studies[7].

There are two principal SP methods used to value TER marine species, contingent valuation (CV) methods and choice experiment (CE) methods. In CV, economic values for a non-market good or service are revealed through survey questions that set up hypothetical markets for a non-market good or service, and involve asking the respondent to indicate their willingness to pay (WTP) for the good or service, which is a theory-based measure of economic value[8]. In a typical contingent valuation survey, a public good is described, such as a program to protect one or more TER marine species, and respondents are asked questions to elicit their WTP for the public good through a payment vehicle, like taxes or contributions to a trust fund (Arrow et al., 1993; Bateman et al., 2002). CV methods are differentiated by the way they elicit WTP. Respondents are commonly asked to state their maximum WTP (an "open-ended" CV question), choose the amount they are willing to pay from a list of values (a "payment card" CV question), or accept or reject a specific amount (a "referendum," or "dichotomous choice," CV question). Variations of these question formats exist, but these are the most frequently used.

When asked properly, answers to CV questions yield an estimate of WTP associated with the good being valued, depending upon the format of the question posed (Freeman et al., 2014). An important point often overlooked is how sensitive these welfare estimates are to features of the good being valued. Carson et al. (2001, p. 180) note the following:

"People have distinct preferences over the exact manner in which they pay for goods and perceive different methods of providing a good to have different likelihoods of success. In this sense, the term "contingent" method is apt and one should never forget that it is only the plan to provide the good that can be valued, not the good in the abstract."

This admonition is sometimes forgotten by those interpreting the results of CV (and generally SP) studies. For instance, the CV survey used in Giraud et al. (2002) asked a referendum CV question that involved voting for a measure that would create an "Enhanced Steller Sea Lion Recovery Program" that would lead to an increase in federal taxes to the respondent's household if approved. The estimated WTP from this survey question is a measure of value of the "Enhanced Steller Sea Lion Recovery Program," which "doubled research funding and increased the restrictions of commercial fishing around the western stock of the Steller sea lion's [critical habitat] in the Gulf of Alaska, Bering Sea and North Pacific Ocean" (p. 454). The WTP is not a measure of the public's value for recovering the species, which is not the object of the valuation question (the program is), although subsequent researchers commonly treat it as such in their analyses (e.g., Richardson and Loomis, 2009). While this is not a weakness of CV *per se*, it is a

[5]Nonuse values are sometimes referred to as passive use values.

[6]See Freeman et al. (2014) for an overview of issues related to motivations for valuing non-market goods, including various use and nonuse motivations, and Cummings and Harrison (1995) for a discussion of the limitations of empirical methods to place dollar values on specific motivations. Carson et al. (1999) also provide an argument against decomposing total economic value into components based on motivations.

[7]RP-based studies valuing activities that have a TER marine species component (usually a viewing benefit) cannot separate the value associated with the TER marine species from the recreational trip value, which has implications on the interpretation of the values estimated and use in benefits transfer.

[8]The theoretically-appropriate measures of economic value are WTP and willingness to accept (WTA; see Freeman et al., 2014). Which of the two is appropriate depends upon property rights—who owns the resource. While WTA is sometimes the more relevant welfare measure, empirical and experimental evidence has pointed to the use of WTP welfare measures in stated preference studies (e.g., Adamowicz et al., 1993; Arrow et al., 1993; Mansfield, 1999). In practice, WTP and WTA need not correspond (e.g., Horowitz and McConnell, 2002; Tuncel and Hammitt, 2014). For the purposes of this article, we follow the majority of the literature and use WTP in reference to measured economic values from the studies discussed herein.

feature that those using the results should be aware of and treat carefully.

CV methods are not the only SP methods available for estimating the TEV of TER species[9]. The stated preference choice experiment (CE) approach has been increasingly used by researchers due to its flexibility (Hanley et al., 1998; Alpizar et al., 2003; Bennett and Birol, 2010; Ryan et al., 2010). In the choice experiment approach, respondents are asked to choose between two or more alternatives that differ in one or more attributes, including cost[10]. Choice experiments offer a useful alternative to CV for estimating a wider range of economic values. By decomposing environmental goods, in the form of choice alternatives (e.g., species protection programs), into measurable attributes (e.g., specific outcomes of protection such as population size, extinction risk, or improved conservation status under each protection program), economic values can be estimated from an analysis of choices between different alternatives. Since choice alternatives are described by their attributes, and the effects of these attributes on choice are estimated in the model, it is possible to estimate WTP for alternatives not originally included in the CE questions seen by respondents, something which CV generally cannot do[11]. Hanley et al. (2001) and Hanley et al. (1998) argue that CE methods have several advantages over CV, among them, built-in scope tests, the ability to estimate values of each attribute, and avoiding some biases in responses typically associated with CV questions. Bateman et al. (2002) also notes CE methods may avoid yea-saying behavior (Blamey et al., 1999; Burton et al., 2007).

The issue of validity of CV and CE results is a central focus of much SP research. Freeman et al. (2014) describes four types of validity: criterion validity, convergent validity, construct validity, and content validity.

Criterion validity involves comparing the SP value to some alternative value that can be taken as the criterion for the assessment. Ideally, the alternative value would be the "true" value. Tests for criterion validity often take the form of tests for hypothetical bias, which is the difference between actual values and those obtained from the SP study. However, the true value is generally not known for non-market goods, especially goods like TER species protection for which their values are predominantly related to nonuse. As a result, classroom or laboratory settings are often used to provide alternative values in settings that are more "market-like" and are conducive for direct comparisons of SP responses with actual behavior in a controlled setting (e.g., Ehmke et al., 2008)[12]. List and Gallet (2001) and Murphy et al. (2005) summarized this literature with respect to CV and found CV values tend to be overstated relative to actual values in these experiments, although Murphy et al. (2005), Champ et al. (2009), and others have noted that ex-ante and ex-post methods, such as cheap talk (Cummings and Taylor, 1999) and certainty scales (Champ et al., 1997), can be effective in reducing hypothetical bias.

There have also been a few studies conducted to evaluate the criterion validity of CE methods. In an experiment conducted on students from two universities in Sweden, Carlsson and Martinsson (2001) found no statistical difference between CE-based WTP estimates and actual donation behavior related to environmental projects. In contrast, Lusk and Schroeder (2004) found that CE responses led to overestimates of actual WTP in an experiment involving a private good (steaks), but the study design did not include either cheap talk scripts or certainty scales to minimize hypothetical bias. In other applications in which these mitigation schemes were used, stated CE and actual WTP were more aligned (List et al., 2006; Ready et al., 2010). Recently, Ladenburg and Olsen (2014) proposed a repeated opt-out reminder to be used in conjunction with cheap talk that was shown to reduce WTP in an empirical application involving preferences for re-establishing a stream in Copenhagen, Denmark.

Convergent validity is generally assessed by comparing SP values with measures derived from other valuation methods. Carson et al. (1996) reviewed 83 studies that compared CV estimates to RP estimates and found the mean ratio of values between the CV and RP methods to be 0.89, indicating that CV estimates yield slightly smaller WTP estimates on average than RP methods across the goods valued in these comparison studies. A small number of convergent validity studies have also been conducted to evaluate CE, most comparing CE to CV (e.g., Boxall et al., 1996; Christie and Azevedo, 2009). These studies have yielded mixed results with respect to convergent validity, though Christie and Azevedo (2009) show that a CV study with a repeated question format similar to the set up for a CE study leads to convergent validity in a study of lake water quality.

Construct validity is concerned with whether SP responses are related to variables that economic theory suggests they should be (e.g., does WTP increase with income?). This type of validity is often assessed by regressing SP values on characteristics of the good being valued and characteristics of the respondent. A specific type of test for construct validity is a scope test, which evaluates whether WTP is sensitive to how much of the good is being offered (e.g., Giraud et al., 1999). Since, CE studies involve estimating a valuation function that depends upon attributes related to the good or service being valued, scope sensitivity in CE is assessed internally by evaluating the signs and significance of parameters to ensure consistency with economic theory. Lew and Wallmo (2011) test for and confirm the presence of scope effects in the only external test of scope in CE (i.e., one using a split-sample testing approach).

[9]In addition to stated preference choice experiments and related conjoint analysis methods (contingent rating, contingent ranking, and best-worst scaling) is a recent method that employs gathering small groups of people in a participatory process that involves some group discussion and processing as a means of determining nonuse values (valuation workshops; Alvarez-Farizo et al., 2007).

[10]Variants of the choice experiment include contingent rating and contingent ranking, where the respondent rates or ranks each choice alternative, respectively, instead of choosing between them. See, for example, Siikamaki and Layton (2007) and Bateman et al. (2006).

[11]It is important to emphasize, however, that the values derived from CE studies are also dependent on the set up of the mechanisms by which the alternatives (programs) are constructed. Thus, care should still be taken in interpreting the measured values, following the Carson et al. (2001) admonition.

[12]Vossler and Kerkvliet (2003) provide one of the few examples of a criterion validity test involving stated and actual voting behavior for a public referendum.

The ability of SP questions to be used to accurately measure people's values for non-market goods depends, in large part, upon the design of the survey, the specific SP question, and the implementation of the survey. The fourth type of validity, content validity, addresses this by evaluating the quality of the survey instrument, including assessing the set-up of the good to be valued, the form and design of the SP question(s) (Kanninen, 1993; Lusk and Norwood, 2005; Johnston et al., 2012), the payment vehicle used, and other characteristics of the survey, as well as elements of the implementation of the survey (Brown, 2003).

In addition to the validity issues above, the reliability of CV estimates has been evaluated, in particular related to temporal stability of stated preferences and values over time (e.g., McConnell et al., 1998; Brouwer, 2006). In general, the weight of evidence suggests stated preferences and values from CV are fairly stable over short time periods (less than 5 years), but not over much longer periods (e.g., 20 years) (Skourtos et al., 2010). Fewer examinations of temporal stability of CE preferences and values have been undertaken, and none have examined long time periods. However, the existing studies tend to support stability of WTP values over short term periods of up to a year (Bliem et al., 2012; Liebe et al., 2012).

Much of the recent research on CE methods has focused on other issues related to improving the econometric modeling of the CE response data to better account for preference heterogeneity via latent class and random parameter discrete choice models (e.g., Colombo et al., 2009), accounting for scale (variance) heterogeneity (Fiebig et al., 2010), combining CE data with other RP or SP data (e.g., Whitehead et al., 2008; Balbontin et al., 2015)[13], and issues related to the complexity of the choice alternatives (e.g., Meyerhoff et al., 2015), such as respondents not paying attention to all attributes when deciding between choice alternatives, a behavior referred to as attribute non-attendance (e.g., Colombo et al., 2013; Glenk et al., 2015).

Although, SP methods have been subjected to criticisms related to the above validity issues (Hausman, 1993, 2012; Diamond and Hausman, 1994), the NOAA Panel on Contingent Valuation, a distinguished panel of economists led by Nobel Laureates Kenneth Arrow and Robert Solow, found that, despite its problems, these "studies can produce estimates reliable enough to be the starting point of a judicial process of damage assessment, including lost passive-use values" (Arrow et al., 1993, p.43)[14]. This conclusion was generally upheld in a recent comprehensive review of SP methods by Kling et al. (2012).

TER SPECIES VALUATION STUDIES

TER species valuation studies can be categorized into two groups—*aggregate* species valuation studies and *disaggregate* species valuation studies. Aggregate species valuation studies value one or more groups of TER species, or a group of species that include TER species, as a whole. These studies yield WTP estimates that cannot be assigned to any constituent species within the group of species valued. Disaggregate species valuation studies, on the other hand, provide estimates of value for individual TER species.

Aggregate Species Valuation Studies

An example of an aggregate species valuation study is one by Olsen et al. (1991), which involved estimating WTP to protect salmon and steelhead in the Pacific Northwest. Since the good valued was all salmon and steelhead in the Pacific Northwest, the resulting welfare values cannot be divided among the different salmon species in the region, or separated from the WTP to protect steelhead. Similarly, economic values that cannot be disaggregated to identify individual species values were estimated by Berrens et al. (2000) for protecting 11 TER fish species in New Mexico and by Lyssenko and Martinez-Espineira (2006) for protecting 17 species of whales off Newfoundland and Labrador, Canada, some of which are TER species.

Additional recent studies of this type that value marine TER species include Farr et al. (2014), Jin et al. (2010), and Ressurreicao et al. (2011, 2012). Farr et al. (2014) estimates the WTP for several broad groups of species sometimes seen by divers in the Great Barrier Reef area—whales and dolphins, sharks and rays, large fish, marine turtles, and a "wide variety of wildlife"[15]. Jin et al. (2010) estimate the WTP of marine turtle conservation using samples from four different Asian countries, but no specific species are valued. Ressurreicao et al. (2011, 2012) estimate the WTP for programs to avoid reducing marine species richness in Europe, measured in terms of the number of species. They presented the species in large marine taxa (mammals, fish, algae, birds, and invertebrates), precluding the ability to assess any individual species' contribution to the estimated WTP.

Among these studies, surveys generally contained little information about the species being valued (except Ressurreicao et al., 2011, 2012), unrepresentative (convenience) samples were sometimes used (Ressurreicao et al., 2011, 2012; Farr et al., 2014), sample response rates were low in some studies (Lyssenko and Martinez-Espineira, 2006; Farr et al., 2014), and only one of the studies (Lyssenko and Martinez-Espineira, 2006) employed either of the measures recommended to minimize hypothetical bias—certainty scales and cheap talk. These issues serve to diminish the utility of the economic value information provided in these studies. But more fundamentally, economic value information from these studies provide information about economic benefits for specific programs that affect multiple ecosystem goods and services, with TER species values embedded and inseparable from the total values estimated. Thus, in general the aggregate species valuation studies provide insufficient information for benefit transfers focused on policy applications involving individual species.

[13]This is also an active research area for CV researchers.

[14]The NOAA Panel provided a number of recommendations for designing and conducting CV surveys that would lead to "reliable" estimates of nonuse value. A number of subsequent studies have been conducted to test the reliability of CV estimates (see Boyle, 2003 for a useful summary).

[15]Note that the analysis was based on a convenience sample, which raises the question about whether the WTP estimates are representative of the intended population.

TABLE 1 | Threatened, endangered, and rare marine species values reported in meta-analyses.

Martín-López et al. (2008, CONSERVATION BIOLOGY)		
Marine species	**Source study**	**Country**
Gray seals	Bosetti and Pearce, 2003	U.K.
Hawaiian monk seal	Samples and Hollyer, 1990; Brown et al., 1994	United States
Mediterranean monk seal	Langford et al., 1998	Greece
Northern elephant seal	Hageman, 1986	U.S.
Steller sea lion	Giraud et al., 2002	U.S.
Beluga whale	Tkac, 1998	U.S.
Blue whale	Hageman, 1985, 1986; Bulte and van Kooten, 1999	U.S., Canada
Bottlenose dolphin	Hageman, 1986	U.S.
Gray whale	Hageman, 1985, 1986; Loomis and Larson, 1994	U.S.
Humpback whale	Samples et al., 1986; Samples and Hollyer, 1990; Brown et al., 1994; Wilson and Tisdell, 2003	U.S., Australia
Loggerhead sea turtle	Whitehead, 1992; Wilson and Tisdell, 2003	U.S., Australia
Atlantic salmon	Stevens et al., 1991; Bulte and van Kooten, 1999	U.S., Canada
Arctic grayling	Duffield and Patterson, 1992	U.S.
Chinook salmon	Hanemann et al., 1991; Olsen et al., 1991	U.S.
Cutthroat trout	Duffield and Patterson, 1992	U.S.
Steelhead	Olsen et al., 1991	U.S.
Shortnose sturgeon	Kotchen and Reiling, 1998	U.S.
Kelp bass	Carson et al., 1994	U.S.
White croaker	Carson et al., 1994	U.S.
Riverside fairy shrimp	Stanley, 2005	U.S.
Loomis and White (1996, ECOLOGICAL ECONOMICS) AND Richardson and Loomis (2009, ECOLOGICAL ECONOMICS)		
Salmon and steelhead	Olsen et al., 1991; Loomis, 1996	U.S.
Salmon	Bell et al., 2003	U.S.
Migratory fish in Oregon and Washington	Layton et al., 2001	U.S.
Blue whale	Hageman, 1985	U.S.
Sea otter	Hageman, 1985	U.S.
Gray whale	Loomis and Larson, 1994	U.S.
Hawaiian monk seal	Samples and Hollyer, 1990	U.S.
Humpback whale	Samples and Hollyer, 1990	U.S.
Atlantic salmon	Stevens et al., 1991	U.S.
Loggerhead sea turtle	Whitehead, 1991, 1992	U.S.
Riverside fairy shrimp	Stanley, 2005	U.S.
Steller sea lion	Giraud et al., 2002	U.S.

Disaggregate Species Valuation Studies

Disaggregate species valuation studies generate species-specific values. Among those providing values for individual TER marine species are ones that estimate the WTP associated with the protection of "charismatic megafauna" like whales (Samples and Hollyer, 1990; Loomis and Larson, 1994; Larson et al., 2004; Boxall et al., 2012; Wallmo and Lew, 2012), seals and sea lions (Samples and Hollyer, 1990; Langford et al., 1998, 2001; Giraud et al., 2002; Giraud and Valcic, 2004; Lew et al., 2010; Lew and Wallmo, 2011; Wallmo and Lew, 2011, 2012; Boxall et al., 2012; Stithou and Scarpa, 2012), and manatees (Solomon et al., 2004), to lesser known species such as the striped shiner (Boyle and Bishop, 1987), the silvery minnow (Berrens et al., 2000), and Riverside fairy shrimp (Stanley, 2005). To date, over 30 studies, representing scores of species, have been published reporting estimates of the economic value of one or more TER marine species.

Many of these TER marine species valuation studies have been summarized and incorporated in meta-analyses (Loomis and White, 1996; Martín-López et al., 2008; Richardson and Loomis, 2009)[16]. See **Table 1** for a list of the species and studies

[16]Another recent meta-analysis of species and nature conservation values in Asia and Oceania was conducted by Lindhjem and Tuan (2012) and includes a broader range of values than just those for TER marine species, including many unpublished studies. They include 16 studies in this region estimating values for one or more species, though these species include terrestrial and non-TER marine species. All the studies were conducted on or before 2009. The authors estimate a meta-regression model to assess determinants of WTP for species valued in these

contained in these meta-analyses. Loomis and White (1996) were the first to summarize the TER valuation literature by employing a meta-analysis of 20 U.S. TER species contingent valuation studies conducted between 1983 and 1994 and found that annual WTP to protect rare and threatened and endangered species (both marine and terrestrial) ranged from \$11 to \$153[17]. They estimated a meta-regression to explain variation in willingness to pay (WTP) across studies using characteristics of the study and of the good being valued as explanatory variables. Much of the variation they found in WTP values could be explained by the type of species valued (e.g., whether it is a marine mammal or bird), by the change in population being valued, and by the type of individual being asked to provide WTP (e.g., user vs. non-user). Richardson and Loomis (2009) updated the Loomis and White (1996) study, adding values from 11 additional U.S. studies conducted through 2005 (including one CE study). The values ranged from \$12 to \$406. In the meta-regression, several new variables, including one to capture effects due to the "charisma" of a species, were added. While generally confirming the results of Loomis and White (1996), they also found some structural change in values from studies conducted more recently than those examined in the earlier study. In addition, their models suggest that studies employing CE methods instead of CV have higher estimates, although this result is based on estimates from a single (unpublished) choice experiment study included in the dataset (Layton et al., 2001). Their models also suggest there is evidence that studies valuing charismatic megafauna have larger values. Loomis and White (1996) included estimates from seven studies valuing marine TER species (Hageman, 1985; Samples and Hollyer, 1990; Olsen et al., 1991; Stevens et al., 1991; Whitehead, 1991, 1992; Loomis and Larson, 1994), including three whale species (blue, humpback, and gray), salmonids (Pacific and Atlantic salmon, steelhead), sea otters, and the loggerhead sea turtle. The Richardson and Loomis (2009) study added additional estimates for salmonids (Loomis, 1996; Bell et al., 2003) and other migratory fish (Layton et al., 2001), as well as fairy shrimp (Stanley, 2005) and Steller sea lions (Giraud et al., 2002).

Another meta-analysis study by Martín-López et al. (2008) includes studies from outside the United States, but is more broadly focused on all species, not just TER species. Of the 60 studies they examined, 65% were from the United States and 15% were from Europe, highlighting the geographic concentration of TER species valuation efforts in a small number of regions. The remaining studies came from Australia (8%), Canada (6%), and Sri Lanka (6%). However, only 20 of these studies valued aquatic species, most of which are also covered by Richardson and Loomis (2009). Of the 20, four are non-U.S. studies. The first of these is a study by Bosetti and Pearce (2003), who estimate the value of several programs to preserve gray seals in Southwest England. Gray seals are not endangered, but are listed in Annex 2 of the EU Habitat Directive due to their scarcity. The second, a study by Langford et al. (1998), estimates the value of a compensation program for fishermen to incentivize them to avoid killing endangered Mediterranean monk seals in Greece. The third non-U.S. study, by Wilson and Tisdell (2003), is an aggregate species valuation study that reports the results from case studies in Australia to value the conservation of sea turtles and whales. The estimated values are for sea turtles and whales in two areas in Queensland, and specific species are not valued. The final non-U.S. study considered by Martín-López et al. (2008) was a study by Bulte and van Kooten (1999) that used benefits transfer to value minke whales in the Northeast Atlantic. Minke whales are not a TER species[18].

These meta-analyses generally do not capture how active researchers have been within the TER valuation literature in recent years. The most recent data included in the most recent meta-analysis (Richardson and Loomis, 2009) were from a study that used survey data collected in 2001 (Stanley, 2005). Since these meta-analyses have been done, over a dozen additional studies to value TER marine species have been published (see **Table 2**), with estimated values ranging from −\$120 to \$356. It should be noted that this range combines both lump sum (one-time) payments and annual payments. Across the studies, one-time payments ranged from −\$9 to \$59, while annual payments had a larger range, from −\$120 to \$356.

Taken together, these studies have greatly expanded the economic value information about TER species in large part due to the shift in valuation methods used in these studies. Specifically, researchers have begun to employ choice experiments to value TER species, which has facilitated the ability to estimate multiple individual species WTP values since protection of individual species can be treated as attributes of conservation or protection programs in this approach[19]. For example, Rudd (2009) used CE methods and a latent class logit model to estimate the value to Canadians of increasing the populations of Atlantic salmon, Atlantic whitefish, the North Atlantic right whale, the porbeagle shark, and white sturgeon off the Atlantic coast of Canada. However, since species was treated as an attribute in the choice question, all estimated WTP values are relative to an unidentifiable value of the least valuable species, which varied across latent classes. This makes comparing WTP values from this study to others difficult.

16 studies, finding good explanatory power from the set of methodological and contextual variables (e.g., population characteristics, characteristics of the good valued, geographic region, etc.). The study does not review or list the studies that form the data.

[17] All estimated values reported herein are in 2013 U.S. dollars, calculated using the Consumer Price Index and, when applicable, foreign currency conversion rates for the appropriate year.

[18] All three meta-analyses included studies from the gray literature (e.g., unpublished papers, theses, and reports), which are not peer-reviewed, instead relying on the fact that they are cited in other studies to be evidence of the quality of the study. In fact, Loomis and White (1996) indicate that half of the studies they drew WTP estimates from fall into this category. This decision may have been driven by the fact that additional data points for the purposes of estimating a meta-regression were needed when the literature had not matured. Of the U.S. studies not included in Loomis and White (1996) or Richardson and Loomis (2009) in the Martín-López et al. (2008) study, there are several unpublished works (Hageman, 1985, 1986; Duffield and Patterson, 1992; Carson et al., 1994). Two of these (Hageman, 1985, 1986) present identical data, models, and WTP estimates (one is a government report and the other a conference paper based on that report).

[19] To our knowledge, Layton and Levine (2005) was the first published study to employ choice experiments to value a TER species (northern spotted owl).

TABLE 2 | Recent disaggregate threatened, endangered, and rare marine species valuation studies[a].

Species	References	Valuation method	Mean/Median WTP range	Frequency of payment	Units[b]	Survey year	Good valued	Country
Short-nosed sturgeon	Aldrich et al., 2007	CV	−$9.38–58.89	One-time	I	1997	Recovery program	U.S.
Harbor seal	Boxall et al., 2012	Hybrid CV/CE	$78.84–201.61	Annual	H	2006	Improved status	Canada
Beluga whale	Boxall et al., 2012	Hybrid CV/CE	$113.58–355.73	Annual	H	2006	Improved status	Canada
Steller sea lion	Giraud and Valcic, 2004	CV	−$119.63–119.29	Annual	H	2000	Recovery program	U.S.
	Lew et al., 2010	CE	$39.26–229.47	Annual	H	2007	Improved status and population increase	U.S.
Mediterranean monk seal	Kontogianni et al., 2012	CV	$75.51–131.54	Unknown[c]	H	2009	Protection program	Greece
	Stithou and Scarpa, 2012	CV	$21.74–29.95	One-time	I	2003	Protection program	Greece
			$17.74–20.41	Per visit	I	2003	Protection program	Greece
Gray whales	Larson et al., 2004	CV	$37.38–56.35[d]	Annual	I	1991–1992	Population increases	U.S.
Hawaiian monk seal	Lew and Wallmo, 2011	CE	$47.47–92.68	Annual	H	2008	Improved status	U.S.
	Wallmo and Lew, 2011	CE	$47.47–73.97	Annual	H	2008	Improved status	U.S.
	Wallmo and Lew, 2012	CE	$39.37–72.00	Annual	H	2009	Improved status	U.S.
Puget Sound Chinook salmon	Wallmo and Lew, 2011	CE	$50.98	Annual	H	2008	Improved status	U.S.
	Wallmo and Lew, 2012	CE	$43.97	Annual	H	2009	Improved status	U.S.
Smalltooth sawfish	Lew and Wallmo, 2011	CE	$36.74–69.79	Annual	H	2008	Improved status	U.S
	Wallmo and Lew, 2011	CE	$36.74–57.97	Annual	H	2008	Improved status	U.S.
	Wallmo and Lew, 2012	CE	$35.24–56.35	Annual	H	2009	Improved status	U.S.
Norwegian lobster	Ojea and Loureiro, 2010	CV	$22.96	One-time	H	2006	Protection program	Spain
Hake	Ojea and Loureiro, 2010	CV	$35.63	One-time	H	2006	Protection program	Spain
Manatee	Solomon et al., 2004	CV	$13.48–28.20	Annual	H	2001	Protection program	U.S.
Loggerhead sea turtle	Stithou and Scarpa, 2012	CV	$22.46–32.12	One-time	I	2003	Protection program	Greece
			$17.22–19.51	Per visit	I	2003	Protection program	Greece
	Wallmo and Lew, 2012	CE	$47.47	Annual	H	2009	Improved status	U.S.
Hawksbill sea turtle	Wallmo and Lew, 2015	CE	$91.82–100.36	Annual	H	2010	Improved status	U.S.
Upper Willamette River Chinook salmon	Wallmo and Lew, 2012	CE	$44.14	Annual	H	2009	Improved status	U.S.
Central California coast coho salmon	Wallmo and Lew, 2015	CE	$54.55–62.13	Annual	H	2010	Improved status	U.S.
Southern California steelhead	Wallmo and Lew, 2015	CE	$75.91–82.86	Annual	H	2010	Improved status	U.S.
Southern resident killer whale	Wallmo and Lew, 2015	CE	$90.14–95.97	Annual	H	2010	Improved status	U.S.
North Pacific right whale	Wallmo and Lew, 2012	CE	$45.30–79.44	Annual	H	2009	Improved status	U.S.
North Atlantic right whale	Wallmo and Lew, 2012	CE	$42.12–77.77	Annual	H	2009	Improved status	U.S.
Humpback whale	Wallmo and Lew, 2015	CE	$65.14–67.46	Annual	H	2010	Improved status	U.S.
Johnson's seagrass	Wallmo and Lew, 2015	CE	$44.18–46.82	Annual	H	2010	Improved status	U.S.
Elkhorn coral	Wallmo and Lew, 2015	CE	$76.68–85.40	Annual	H	2010	Improved status	U.S.
Black abalone	Wallmo and Lew, 2015	CE	$75.32–85.03	Annual	H	2010	Improved status	U.S.
Leatherback sea turtle	Wallmo and Lew, 2012	CE	$41.22–73.81	Annual	H	2009	Improved status	U.S.

[a] *WTP is reported in 2013 U.S. dollars (all values converted using consumer price index and annual currency conversion rates).*
[b] *Units refer to the value's unit measurement in terms of household (H) or individual (I).*
[c] *The payment vehicle was a contribution made on the water bill, but the frequency of billing was not mentioned.*
[d] *Also presents estimated WTP in non-monetary terms (hours donated).*

In contrast, Lew et al. (2010) analyze CE questions which treat population increases and changes to Endangered Species Act (ESA) status as attributes, which allow them to estimate the value of increasing the population and improving the status of two ESA listed stocks of Steller sea lion. Using a similar framework, Wallmo and Lew (2011) and Lew and Wallmo (2011) present values associated with improving the ESA status of three TER species, the Puget Sound Chinook salmon, smalltooth sawfish, and the Hawaiian monk seal, using a small web-based national sample in the United States. Additionally, Lew and Wallmo (2011) show that non-consumptive values for these species are sensitive to scope, both in terms of the number of species protected and the amount of improvement (measured in terms of status improvement). Using data from an expanded survey effort using the same web-based survey framework, Wallmo and Lew (2012) estimated a pooled model of surveys that each asked respondents to value ESA improvements to three of eight species. The eight species included those valued in Lew and Wallmo (2011) and Wallmo and Lew (2011), as well as the North Atlantic right whale, North Pacific right whale, leatherback sea turtle, loggerhead sea turtle, and Upper Willamette River Chinook salmon[20]. The most recent CE-based study is a follow-up to the Wallmo and Lew (2012) study that presents the public's WTP for recovering each of eight additional TER marine species, including several non-charismatic species (Wallmo and Lew, 2015). Specifically, the study examines whether there are differences in recovery values between a large national sample and a geographically-embedded (i.e., a subset) sample for 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. These CE studies generally conform to recent best practices, using large national samples collected using statistical survey sampling methods and employing methods and models that minimize common biases (e.g., hypothetical bias) and account for preference heterogeneity.

Despite the increasing use of SPCE methods to value TER species protection, CV remains popular, as evidenced by several recent studies by Solomon et al. (2004), Ojea and Loureiro (2010), and Stithou and Scarpa (2012). Solomon et al. (2004) use a mail CV survey of residents of one county in Florida to ask respondents to indicate how much they would donate to a fund to protect endangered manatees under the counterfactual that government protection of manatees in Florida was removed. A modified payment card CV question was asked, and a mean household WTP of $13.48 was reported. Ojea and Loureiro (2010) analyze responses from a sample of Galician households (Spain) to referendum CV questions to estimate values for programs to preserve the minimum viable population (MVP), as well as increases in population above MVP, of two TER species, Norwegian lobster and European hake. In their final models, they pool CV responses over four different programs valued that differ in the extent to which they would increase population sizes. The pooled models resulted in WTP estimates of $22.96 and $35.63 for programs to protect the Norwegian lobster and European hake, respectively. Another recent CV study was a small pilot study conducted by Stithou and Scarpa (2012), who value the protection of two endangered species, the loggerhead sea turtle and Mediterranean monk seal, on the island of Zakynthos, Greece, by visitors. Their primary focus is exploring the difference in responses to open-ended CV questions that value protection through the use of a marine protected area where the species are found and that differ in the payment vehicle (a donation vehicle and a mandatory landing fee). Estimated WTP values ranged from $17.74 to $29.95 for the Mediterranean monk seal program and $17.22 to $32.12 for the loggerhead sea turtle program.

Several other recent CV studies provide additional values that update those from previous analyses. Giraud and Valcic (2004) re-analyze the data presented in Giraud et al. (2002) to assess whether values for Steller sea lion protection are sensitive to distance. They estimate WTP estimates for the United States, the state of Alaska, and local boroughs near Steller sea lion habitat and find significant differences and a positive relationship between geographic distance (and the extent households are negatively affected by protection measures) and WTP. Larson et al. (2004) extend the analysis of data first analyzed by Loomis and Larson (1994) to generate estimates for values held by whalewatchers for increasing the population size of gray whales in California estimated from a model that jointly estimates WTP from responses to referendum CV questions asking respondents how much they would be willing to donate in money to a dedicated protection fund or volunteer in time to the effort. Using the data of Kotchen and Reiling (1998, 2000), Aldrich et al. (2007) use cluster analysis and latent class analysis to estimate WTP for a program to protect the shortnosed sturgeon associated with different groups of respondents based on their environmental preferences. These estimates ranged from $2.54 to $58.89 for the cluster analysis based approach, and −$9.38 to $58.89 for the latent class logit modeling approach. A fourth study, by Kontogianni et al. (2012), conducts a survey of residents of Lesvos, Greece, that values a fishing compensation program aimed at reducing mortality associated with commercial fishermen targeting Mediterranean monk seals. To evaluate whether a service providing unit (SPU) approach can be used to reduce hypothetical bias (Kontogianni et al., 2010), they use a split sample approach that employs the same CV survey instrument used by Langford et al. (1998) and Langford et al. (2001) and one that is identical in all aspects except it adds a description of an ecological service provided by Mediterranean monk seals–as a species that helps to reduce jellyfish outbreaks that hamper beach activities. An open-ended CV question was used in combination with a payment principle question[21], resulting in a mean WTP of $131.54.

Another recent TER marine species valuation study combines aspects of both CV and CE. Boxall et al. (2012) value improvements in the status and population of St. Lawrence beluga whales, St. Lawrence harbor seals, and Atlantic blue whales in Canada. Their hybrid approach involved setting up the choice questions as a referendum between the status quo and a program that would lead to improvements in one or more species, lead to a change in regulations and size of marine protected areas, and cost the household in terms of higher taxes and increased prices for food. In this way, their choice question is similar to the questions in the CE studies above, except respondents were asked to choose between two options instead of three. However, their approach differed from the CE studies since they presented only six programs (i.e., alternatives)

[20]These CE studies also used mitigation schemes (cheap talk scripts and/or certainty scales) to reduce hypothetical bias.

[21]A payment principle question is sometimes used in combination with a CV or CE question to aid in the evaluation of the response to the SP question by determining whether the respondent would be willing to pay in principle for the change being discussed without discussing money amounts.

in the surveys. Due to budgetary constraints, they were unable to employ multiple surveys generated by an experimental design that would allow them to better understand the trade-offs between the attributes. As a result, the choice response data were treated as referendum CV data and analyzed accordingly, resulting in a single WTP estimate for each of the six presented programs[22].

Note that in this study, and in the recent CE studies, the sampling frames have been on a large, often national, scale. This is in contrast to most CV studies in the literature which often use smaller, local or regional populations, although there are exceptions (e.g., Giraud and Valcic, 2004; Lyssenko and Martinez-Espineira, 2006; Jin et al., 2010). In addition to sampling from sub-national populations, a few of the recent CV studies surveyed specialized sub-populations, such as tourists or other user groups (e.g., Larson et al., 2004; Stithou and Scarpa, 2012).

Although this recent literature has increased the number of TER marine species valued and the number of WTP estimates of TER marine species, the range of species appears to have remained within the existing scope of earlier studies. Except for one crustacean, the Norwegian lobster, all recent TER marine species valuation studies value either charismatic megafauna (e.g., whales, seals, sea lions, sea turtles, and manatees) or fish (e.g., salmon, smalltooth sawfish, hake, sturgeon). In terms of geographic coverage, the studies in **Table 2** also do not expand the literature much, with the only new country represented being Spain by one study (Ojea and Loureiro, 2010).

An important difference between TER valuation studies relates to what they are seeking to value. For instance, Loomis and Larson (1994) and Larson et al. (2004) ask respondents (California households and tourists) for their WTP for an "Enhanced Gray Whale Fund" that would be used to help *increase population levels* for gray whales. This valuation of an improvement to the species beyond the status quo levels is in contrast to Hageman (1985), Samples and Hollyer (1990), and Solomon et al. (2004), all of whom ask respondents to value protecting species from decreasing from current levels. That is, these latter studies elicit WTP for *preserving current levels*, which implies maintaining species at threatened or endangered levels, not changing them to some improved level. In the recent CE studies, the good being valued is generally improvements in one or more attributes describing species protection programs, such as status or population improvements. This distinction is important to the extent that WTP varies with both the size of TER species population levels and with changes to their threatened or endangered status (Fredman, 1995). Bulte and van Kooten (1999) make the important point that CV studies often are not valuing marginal values that are useful or necessary for policy analyses. They argue for studies to focus on estimating marginal values, illustrating their importance in a study valuing minke whale preservation in the Northeastern Atlantic Ocean. They use benefits transfer to illustrate how values for minke whale preservation are sensitive to the marginal value of another minke whale, as well as the total WTP of preservation (above a minimum viable population, or MVP, that is necessary for preserving the species). They argue for valuing both WTP of preservation and WTP of population increases above the MVP.

[22]Note that none of the programs allow one to identify a separate WTP for blue whales since the programs valued only include improvements to blue whales when improvements to both beluga whales and harbor seals also occur.

Several studies have also attempted to address issues related to uncertainty. Lew et al. (2010) estimate WTP for improvements in the population size and status of Steller sea lions relative to several different status quo scenarios that differ in the baseline trend of the species, which is similar to Rudd (2009), although the programs valued in that study differ in the funding mechanism and probability of success as opposed to the baseline species' trend under the status quo. In both of those studies, supply uncertainty (of the species protection programs) is treated exogenously, which contrasts with several earlier CV-based treatments that allow for both demand and supply uncertainty (e.g., Whitehead, 1991, 1992).

APPLYING TER MARINE SPECIES VALUES TO POLICY

Economic value information for TER marine species can potentially be used in several ways by policymakers and analysts. As noted earlier, these values can be used as inputs in marine-based EBM contexts to enable the fuller accounting of the scope and magnitude of the private and social benefits and costs associated with policies affecting marine biodiversity and other ocean and coastal resources[23]. The values can be used in evaluating trade-offs between multiple uses formally in a benefit-cost analytic (BCA) framework. This is the approach taken in a fisheries-based EBM setting by Sanchirico et al. (2013). They included economic value estimates associated with protecting a TER marine species (the Steller sea lion) from Lew et al. (2010) in a benefit-cost analytic framework that could inform trade-offs between the costs to the fishery sector and the benefits to the public of different levels of protection.

TER marine species values may also be important inputs in the species management process. For example, in the U.S. economic information about the non-market benefits and costs of protecting a species is precluded from the decision to list the species under the ESA, but economic values may be considered in the designation of critical habitat and the development of species recovery plans. To date, the few applications of TER species values being used have been through the regulatory analyses required in the process of designating critical habitat, such as Regulatory Impact Reviews conducted for compliance with U.S. executive orders (e.g., Executive Order 12866). These applications have been primarily qualitative in nature, but quantitative BCA is feasible in some cases, provided the estimated economic values measure changes to elements of the species' health (e.g., population size, extinction risk, conservation status,

[23]There are also efforts to value ecosystem values beyond just species values being conducted at a global scale, such as the Economics of Ecosystems and Biodiversity (TEEB) study (McVittie and Hussain, 2013). The TEEB study has produced a valuation database that includes a large number of economic values produced from 248 studies around the world related to both terrestrial and marine ecosystem services, including biodiversity.

etc.) directly impacted by policy, or the policies themselves. Another potential application for TER marine species values is in natural resource damage assessments (NOAA, 1996; Jones, 2000). When a TER marine species is harmed in an oil spill or hazardous materials leak triggering a natural resource damage assessment, economic values for the TER marine species affected may be desired (Unsworth and Petersen, 1995)[24] .

In most policy settings in which TER marine species values are desired, policy analysts will lack the time and resources to have de novo SP studies conducted to produce these values. Instead, policy analysts commonly turn to the literature to use, or transfer, economic value information from one or more previously completed studies to a new application (referred to as the "policy application"). The process of using existing value information in a new policy application is called benefits transfer, or environmental value transfer (Johnston and Rosenberger, 2010; Navrud and Ready, 2007)[25].

There are three general approaches typically used to transfer economic benefit information from an existing study to a new application[26]. The unit value transfer approach is the simplest and easiest benefits transfer method and typically involves using the mean or median economic value estimate from an existing study directly in the new policy application (Boyle and Bergstrom, 1992; Desvousges et al., 1992). Typically, no adjustments are made to the value estimate to account for differences in the population of interest that may arise due to socio-demographic, resource use, or behavioral differences.

In a second approach, the value function transfer approach, the estimated function from the existing study that was used to calculate economic values is used directly instead of the values themselves (Loomis, 1992). Adjustments to the value estimate arise by inserting information about the new policy application into the transferred value function. For example, if in the original study a WTP function was estimated as a function of demographics of the sample, a new WTP estimate could be calculated from the function by inserting the demographics of the population of interest in the new policy application.

Alternatively, the meta-regression functions estimated in some meta-analyses, such as the ones described earlier by Loomis and White (1996), Richardson and Loomis (2009), and Lindhjem and Tuan (2012) can be used similarly to the value function transfer approach to provide a customized estimate of economic value for the new policy application (Bergstrom and Taylor, 2006; Johnston et al., 2006). This third type of benefits transfer method has been employed increasingly in recent years (Johnston et al., 2006; Rosenberger and Phipps, 2007; Shrestha et al., 2007)[27].

Regardless of the method used, benefits transfer is only useful if it provides valid estimates of value for the new policy application. The decision of which benefit transfer method and the study or studies to use can greatly impact this. The validity of transferred values has been studied extensively for unit value transfers and value function transfer. The literature of evaluating the extent of transfer errors in benefits transfer appears to be mixed (Johnston and Rosenberger, 2010; Kaul et al., 2013). Rosenberger and Phipps (2007) and Rosenberger and Loomis (2003) provide useful summaries of many of these studies, which seek to evaluate the difference between the transferred values and values from *de novo* studies conducted for the policy application or site (an approximation of the "true" values); this difference is called the "transfer error". Their analysis of the tests of the validity of unit value and value function transfers indicate that the greater the similarity of the original study to the policy application, the smaller the expected transfer error will be. Moreover, there is evidence in the literature that value function transfers yield more accurate values for the policy application than unit value transfers. This makes sense, given the ability to further reduce the dissimilarity between the original study and the policy application by adjusting the value for characteristics of the policy application.[28] There is also some evidence that the use of meta-analysis to transfer benefits outperforms value function transfers (Rosenberger and Phipps, 2007; Shrestha et al., 2007). In summary, the literature seems to support the idea that the more closely the researcher can customize the value estimate to the new policy application, the more accurate the transferred value will be to the value that would be generated if a primary study had been done.

In addition to transfer errors, measurement errors, which reflect divergences between the true WTP and the primary study's estimate, are critical to a valid transfer (Johnston and Rosenberger, 2010). McConnell (1992) notes that consideration must be given to the quality of the original study, suggesting that the transferred value or function can only be as good as the original upon which it is based. This point is particularly persuasive, given that meta-analyses have shown how researcher judgments about how to define the good, the type of valuation methods used, and the manner of implementing the survey, along with other characteristics of the study, can have significant effects on economic values (Johnston and Rosenberger, 2010).

The quality of an original study depends upon the data and methods used. Best practices with respect to statistical survey sampling, SP survey design, and econometric modeling of SP

[24] An alternative approach for calculating damages (or injuries) that does not require measurement of economic values, habitat equivalency analysis (HEA), is frequently used instead of an economic valuation approach (Dunford et al., 2004; Roach and Wade, 2006).

[25] Benefits transfer has received considerable interest by researchers and policy analysts in the last two decades. Special issues of Water Resources Research (Volume 28, number 3) and Ecological Economics (Volume 60, number 2) have been dedicated to this subject. See also Brouwer (2000),Navrud and Ready (2007), and Rosenberger and Loomis (2003) for overviews and details about the methodology.

[26] An additional benefits transfer approach called preference calibration is less commonly used, likely in large part due to its complexity relative to other methods. It requires making assumptions about the specific form for a representative member of the population's underlying preferences, or utility function, then "calibrating" this preference function, using information about the economic values from one or more studies (Smith et al., 2002). The calibrated preference function is then used to generate value estimates for the new policy application, much like value function transfer.

[27] Recently, Bayesian modeling approaches have been used to extend this approach (e.g., Moeltner et al., 2007).

[28] Boyle and Bergstrom (1992) caution that in choosing a study to use for benefits transfer to maximize the likelihood of a valid transfer, the non-market good needs to be the same as the one in the new application and the population characteristics of the original study need to be similar in the new application, conditions that are rarely met in practice.

responses are not static, but evolve over time. As noted earlier, the CE studies reviewed here generally conform to recent best practices (except, perhaps, for the most recent issues related to attribute non-attendance and scale heterogeneity) and use large national samples collected using statistical survey sampling methods. In part, this is likely because they were intended to generate general population estimates that could be broadly applied in ocean or coastal management scenarios; additionally, they are more recent and therefore employ more recently developed empirical methods. Thus, these studies offer a useful, but somewhat limited in terms of overall coverage, pool of WTP values to draw upon. On the other hand, the CV studies discussed here have not all conformed to recent best practices to minimize potential biases associated with the method, in part due to many of the studies being conducted decades ago. Even among recent CV studies only Stithou and Scarpa (2012) and Boxall et al. (2012) use certainty scales and/or cheap talk in their surveys to minimize hypothetical bias. Note, however, that Stithou and Scarpa (2012) relied upon on a very small sample size to generate the estimates in their study.

In the TER marine species literature, the fact that only a small proportion of TER marine species have economic values estimated for them, and those economic values often represent different things—the value of preserving the species, the value of a protection program, or the value of a marginal improvement in population size or conservation status, for instance—poses a challenge for analysts wishing to find appropriate studies to use in benefit transfers for many TER marine species. On the positive side, with the different types of economic values being measured, it is more likely that values analysts desire can be found. For instance, many of the recent studies provide estimates of improvements in the species in terms of population size or status improvements. These lend themselves to use in evaluations of protection programs that lead to those types of species improvements, which are generally the goals of conservation actions. Moreover, given that most studies are concentrated in a small handful of developed countries, analysts may wish to transfer values across borders. However, as recent studies that have conducted international benefits transfers have shown, there remain numerous questions about the best manner in which to conduct these types of transfers to minimize transfer error (Lindhjem and Navrud, 2008; Brouwer et al., 2015).

Another complication concerns the temporal stability of WTP estimates. If people's preferences and values for protecting TER marine species change over time, then using older value information in a benefits transfer will lead to biased results (i.e., increase the transfer error). In general, the empirical literature assessing the temporal stability of WTP estimates from SP studies, generally through test-retest samples or two independent samples engaged at different time periods, suggests that time periods up to about five years yield temporally stable preferences and values (e.g., Carson et al., 1997; McConnell et al., 1998; Brouwer and Bateman, 2005; Skourtos et al., 2010; Liebe et al., 2012)[29] . If one applies this rule of thumb to the literature examined here based on publication year, only eight studies (Lew et al., 2010; Ojea and Loureiro, 2010; Wallmo and Lew, 2011, 2012, 2015; Boxall et al., 2012; Kontogianni et al., 2012; Stithou and Scarpa, 2012) comprise the set of viable studies that are recent enough to have preferences and values that are likely unchanged, but with several due to "expire" shortly. If a more strict application of this rule is used—one where the year the survey was conducted is used as the indicator of the age of the WTP estimate—then *none* of the studies are usable. Obviously, this would preclude the use of a meta-analytic benefit transfer approach. It also raises questions about using existing meta-regressions that rely on older studies in benefit transfers (e.g., Richardson and Loomis, 2009).

TER marine species values are predominantly composed of nonuse value, which are specific to the species. Transferring value information across species, therefore, assumes that nonuse values are similar across species. This was an implicit assumption in Bulte and van Kooten (1999), for instance, which used gray whale values to value minke whale populations. However, Wallmo and Lew (2012) found statistical differences in WTP between a number of species, but generally found similarity in values between similar species (e.g., between TER right whale species and distinct salmon populations). This finding reinforces the importance of using TER species values in benefit transfers that are for the same or very similar species.

And finally, we note that although in most cases related to policies and programs that affect TER marine species (or are at least focused in some way on these species), economic values representing the total economic value are appropriate for consideration, there are likely some cases where this does not hold and where only specific ecosystem goods or services related to the TER marine species may be desired. For instance, there is a literature on examining the value of recreation activities related to species—eco-tourism activities like wildlife viewing (Tisdell and Wilson, 2002) or viewing benefits associated with diving (Vianna et al., 2012). A review of that literature is beyond the scope of this paper, but on-line databases such as EVRI (https://www.evri.ca) and Envalue (http://www.environment.nsw.gov.au/envalueapp/), or the Economics of Ecosystems and Biodiversity (TEEB) (http://www.teebweb.org/) global initiative that intends to collect and make transparent economic values associated with nature, have cataloged a large number of studies from this literature, as well as the broader ecosystem goods and services valuation literature. Many of the studies reviewed here, as well as unpublished studies valuing TER marine species, are included in these repositories.

CONCLUSIONS

In this paper, the availability and use of economic value information for TER marine species that can be applied in EBM, species management, and damage assessment applications were discussed. In most cases, benefit transfer methods are needed to transfer existing economic value information from this literature to policy applications, given the resource and time costs of

[29]This assumes that no "extreme event" intervenes that would propagate a change in preferences and values (e.g., Brouwer, 2006).

conducting primary studies. Of course, the use of benefit transfer methods requires the availability of economic value estimates that are appropriate for transferring, which presumes an inventory of values exists that meet some minimum standard for use in this context.

Over 30 studies valuing TER marine species were identified from the published literature. The discussion principally focused on describing disaggregate species studies that produce WTP estimates for individual species, which is generally the desired input for policy. The review revealed that all studies published to date were conducted in developed countries (United States, Canada, Australia, U.K., Spain, and Greece), with the highest concentration of studies occurring in the United States. The majority of species valued can be classified as charismatic megafauna—seals and sea lions, whales, and sea turtles—plus well-known fish species, like salmon. Only a small handful of lesser known species are included among those valued to date. Species value estimates were as much as $356 (2013 U.S. dollars), but differed in the frequency of payments (e.g., lump sum vs. annual), the entity paying (e.g., household, resident, or visitor), and the specific good being valued (e.g., species preservation or a type of enhancement).

Attention was then turned to how to apply these values in policy applications using benefit transfer methods. In some ways, the discussion of benefit transfers of TER marine species values painted a decidedly grim picture, at least in terms of our present ability to use benefit transfer methods to transfer these values to new applications on a widespread basis. In large part, this is because of the need to closely match up the economic value being transferred to the characteristics of the desired economic value for the policy application necessary to minimize transfer errors. This is influenced by the small proportion of TER marine species for which there are economic value estimates, the limited geographic distribution of values, and concerns about the temporal stability of estimates from some studies. Moreover, methodological improvements in the stated preference methodology continue to be made and need to be adopted by researchers valuing TER marine species values to ensure the values used in benefit transfers reflect best practices and provide the most accurate estimates.

However, the message is not all bleak. Despite the holes identified in the literature, this review has highlighted that the economic value information about TER marine mammals and fish (particularly salmonids) has been improved, both in terms of species studied and the types of WTP estimates being generated that can potentially be used in policy applications. In addition, economic values for TER sea turtles have been updated. The review underscores the growth of this literature in recent years and the increased rate at which economic value information is being produced (due in part to the shift toward CE valuation methods). This is particularly true for values that are likely to be most applicable in policy, such as WTP associated with specific improvements estimated from samples of general populations. It also points to the need to continue updating these values with new studies due to concerns about temporal stability of the SP-based value information, as well as to expand the types of species valued. Moreover, benefit transfers remain a very active area of research. As these methods improve, so should our ability to integrate TER marine species values into policy.

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Public preferences for endangered species recovery: an examination of geospatial scale and non-market values

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Non-market valuation allows society to express their preferences for goods and services whose economic value is not reflected in traditional markets. One issue that arises in applying non-market values in policy settings is defining the extent of the economic jurisdiction—the area that includes all people who hold values—for a good or service. In this paper, we estimate non-market values for recovering eight threatened and endangered marine species in the US for two geographically embedded samples: households on the west coast of the US and households throughout the nation. We statistically compare species values between the two samples to help determine the extent of and variation in the economic jurisdiction for endangered species recovery. Our findings offer support to the tenet that the summation of non-market values across the country is appropriate when evaluating alternative policies for endangered species recovery.

Keywords: non-market valuation, endangered species, economic jurisdiction, stated preference, choice experiment

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Introduction

In April of 2013 there were 1438 plants and animals listed as endangered or threatened in the U.S. under the U.S. Endangered Species Act (ESA) (US Fish and Wildlife Service Species Reports, 2013a). Since the inception of the ESA in 1973, the U.S. Fish and Wildlife Service has declared <2% of species listed under the Act as "recovered" (US Fish and Wildlife Service Species Reports, 2013b). Although a handful of species have made progress towards recovery, limited public funding combined with species habitat degradation and threats from invasive species render a "recovered" designation for many ESA-listed species increasingly unrealistic (Scott et al., 2005). Concepts such as a recovery continuum (Scott et al., 2005) or the use of protected areas (Blossey, 2012) may be more feasible or effective than the current process, though, as Scott et al. (2005) note, "societal values determine how much effort or how many resources should be allocated to preventing extinctions and maintaining populations of rare or threatened species."

One method that allows society to express its value for species conservation is non-market valuation. Though putting a dollar value on nature is often debated (Ehrenfeld, 1988; Blossey, 2012; Marvier, 2012), the method can provide a systematic assessment of society's preferences for recovering species and offers a common numeraire, a dollar value, for policy analysts to evaluate tradeoffs. One issue that arises in applying non-market values in policy settings is defining the extent of the economic jurisdiction—the area that includes all people who hold values. This involves understanding whether values for a non-market good (a public good, like protecting species) extend only to those living in close proximity to the good or to a larger geographic scale. From a policy perspective, this is critical as it determines the population upon which to sum individual or household values (Bateman et al., 2006). Compounding this is the heterogeneity that may exist for a non-market good across different spatial scales. Previous research has demonstrated a distance-decay function for non-market values where the value of a good decreases as the distance from the good increases. For example Georgiou et al. (2000) found that willingness-to-pay (WTP), an economic measure of value, for a large improvement in river water quality declined to zero at a distance of about 36 miles from the river site. The distance-decay effect has been observed for use values (e.g., values for non-market goods that people use, such as parks or recreation sites) and non-use values (e.g., values for goods that people may never see or use but are nonetheless willing to pay to preserve, Bateman et al., 2002; Hanley et al., 2003).

In contrast to the above, Giraud and Valcic (2004) found that non-use values for Steller sea lion preservation were larger as the geographic extent of the market increased. The Steller sea lion is found primarily in waters of the North Pacific Ocean, Gulf of Alaska, and Bering Sea. In examining willingness-to-pay for the species protection across geographically embedded samples they found that values were highest for the U.S. sample, followed by values for the state of Alaska, and then the Alaskan Boroughs containing Steller sea lion critical habitat. This finding may be due to the fact that local populations may bear a disproportionate share of the cost of protection (in terms of resource use restrictions), uncertainty about protection measures being successful, knowledge levels in different regions (Giraud and Valcic, 2004), or other latent rationale.

Though the Giraud and Valcic (2004) results are dissimilar to the general findings on distance-decay, the increased fishing restrictions associated with protecting the species in their study—the Steller sea lion—have potentially negative impacts on employment in local communities, and this may have caused the lower WTP values at closer proximities to the resource. In this study, we use a similar approach to Giraud and Valcic (2004) by estimating values for eight different species for two geographically embedded samples (of different spatial scale): (a) the west coast region of California, Oregon, and Washington and (b) the entire U.S. For each, we estimate values for recovering taxonomically dissimilar species including the hawksbill sea turtle *Eretmochelys imbricata*, southern resident killer whale *Orcinus orca*, humpback whale *Megaptera novaeangliae*, Southern California steelhead *Oncorhynchus mykiss*, Central California coast Coho salmon *Oncorhynchus kisutch*, black abalone *Haliotis cracherodii*, Elkhorn coral *Acropora palmata*, and Johnson's seagrass *Halophila johnsonii*. The species' distributions are also disparate, ranging from localized state or regional populations to worldwide. Following Giraud and Valcic (2004) we test whether the values for species recovery are statistically different for the geographically embedded samples. In addition to adding eight species values to the non-market valuation literature, most of which have not been previously valued, our findings have important policy implications as they inform on the extent of and variation in the economic jurisdiction for endangered species recovery.

Materials and Methods

Survey Design and Implementation

Economic preferences for the eight species listed above were collected in a survey containing several stated preference choice experiment (SPCE) questions. The species, their ESA status, and geographic range are shown in **Table 1**. The SPCE approach is grounded in Lancastrian consumer theory (Lancaster, 1966), which specifies that an individual's utility for a good is a function of its attributes. In a SPCE, respondents are asked to choose between two or more alternatives that differ in several attributes. These attributes have a range of levels, and experimental design plans are used to generate different combinations of attributes and levels seen by respondents in each of several survey questions and survey versions. By including price as an attribute in an SPCE, the economic value of changes in attribute levels can be estimated. For a detailed explanation of the SPCE approach, see Adamowicz et al. (1998).

The stated preference choice experiment survey was developed over a 3-year qualitative research period that included

TABLE 1 | Species in the stated preference choice experiment survey.

Common group	Common name	ESA status	Geographic range
Marine turtles	Hawksbill sea turtle	Endangered	Atlantic, Pacific, Indian Oceans, and Caribbean Sea
Whales	Southern resident killer whale	Endangered	Off the California, Oregon, Washington, and southern British Columbia coasts
	Humpback whale	Endangered	Worldwide
Plants	Johnson's seagrass	Threatened	Small stretch of coastal lagoons in Southeastern Florida
Anadromous fish	Central California coast coho salmon	Threatened	Tributary rivers and streams of Northern and Central California
	Southern California steelhead	Endangered	Tributary rivers and streams of Central California to Northern Mexico
Coral	Elkhorn coral	Threatened	Shallow waters throughout the Caribbean Sea
Shellfish	Black abalone	Endangered	Shoreline of Northern California to Mexico

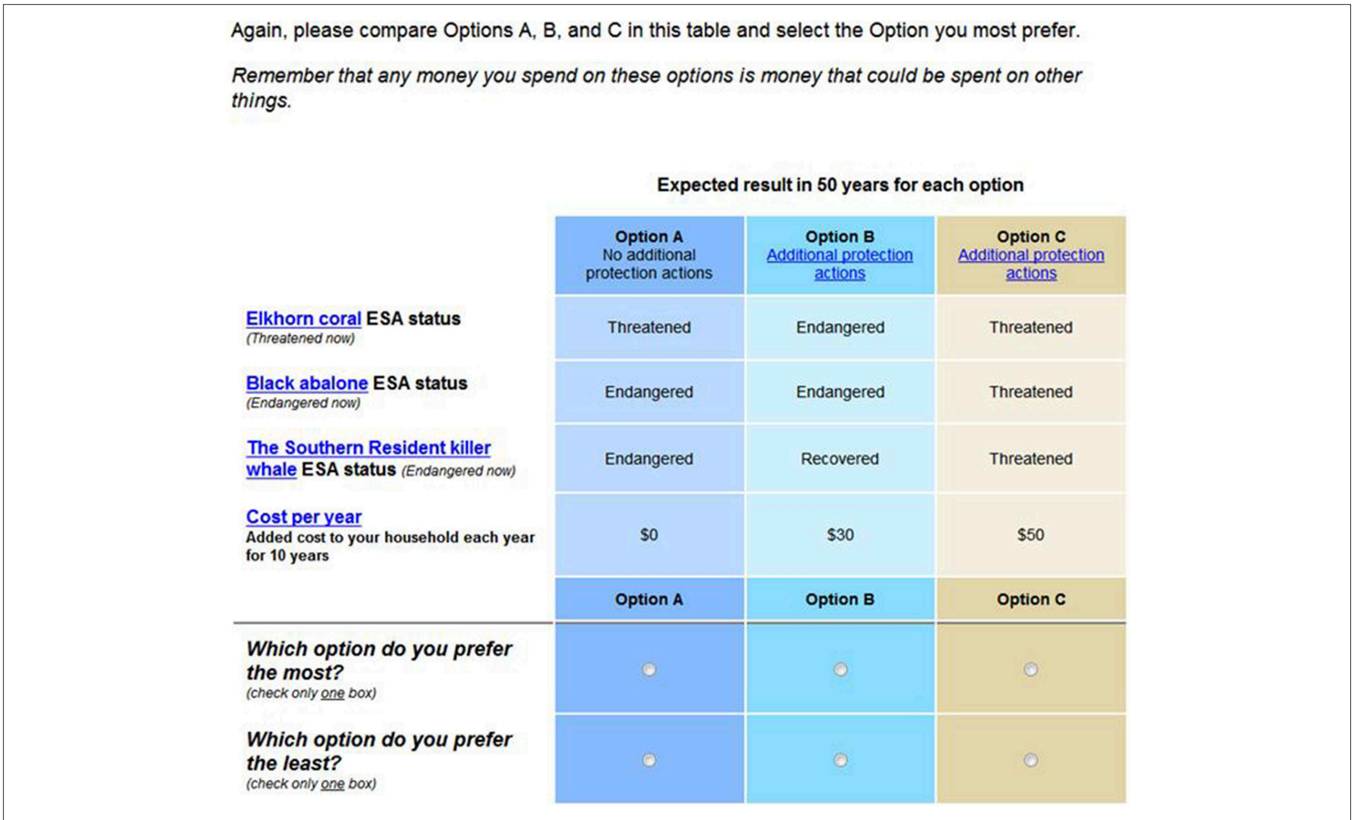
a series of focus groups, one-on-one cognitive interviews, and pretesting activities. Although the overall survey framework included eight species, qualitative research indicated that including more than three species in any one survey version was too much information for respondents to evaluate. Therefore, each survey version included information about a subset of three of the eight species. An experimental design plan was used to select the three species appearing in each survey and the improvements (if any) in each alternative in terms of the ESA-status (endangered, threatened, or recovered) for each species achieved 50 years from now. All future ESA-status levels were described as a result of additional protection measures undertaken for one or more of the species.

In the survey instrument, respondents were provided with basic information about each of the three species and additional protection measures (above and beyond current protection actions) that could be undertaken to improve the species' future ESA-status level. Respondents were then shown three separate SPCE questions (**Figure 1**), with each question containing a status-quo (a no-cost alternative that had no improvements to the ESA-status levels of any species) and two additional alternatives that improved the future ESA-status level for one or more species, at an increased cost to the household. The cost to the household is described in terms of a combination of increased taxes and costs of goods and services affected by the additional protection actions. Respondents were asked to indicate their most preferred and least preferred option, allowing for a full rank ordering of preferences.

The survey was implemented in October and November of 2010 by Knowledge Networks (KN) utilizing a random sample of the KN web-enabled panel of U.S. households (for information on Knowledge Networks web-enabled panel and panel recruitment methods see www.knowledgenetworks.com). A modified Dillman et al. (2009) approach was used to field 16,359 surveys to randomly selected panel respondents across the U.S. A total of 10,637 surveys were completed, resulting in a completion rate of 65%. Of the 16,359 surveys fielded at the national level, 2684 were fielded to households in California, Oregon, and Washington and 1742 of these were completed, resulting in the same completion rates for the geographically embedded west coast region and national samples.

Data Analysis

SPCE data are analyzed using models grounded in random utility theory, which specifies that utility for a good consists of a systematic, known component and a random component (an error term). Individuals are assumed to choose a good (from a set of goods) that maximizes their utility, with



Again, please compare Options A, B, and C in this table and select the Option you most prefer.

Remember that any money you spend on these options is money that could be spent on other things.

Expected result in 50 years for each option

	Option A No additional protection actions	**Option B** Additional protection actions	**Option C** Additional protection actions
Elkhorn coral ESA status *(Threatened now)*	Threatened	Endangered	Threatened
Black abalone ESA status *(Endangered now)*	Endangered	Endangered	Threatened
The Southern Resident killer whale ESA status *(Endangered now)*	Endangered	Recovered	Threatened
Cost per year Added cost to your household each year for 10 years	$0	$30	$50
	Option A	**Option B**	**Option C**
Which option do you prefer the most? *(check only one box)*	○	○	○
Which option do you prefer the least? *(check only one box)*	○	○	○

FIGURE 1 | Example of stated preference choice experiment question.

utility being a function of the good's attributes. In this case, the good is the alternative protection program and the good's attributes are the species ESA-status levels. The set of goods are the two increased-cost alternatives and a no-cost alternative, described above. Due to the significant literature on stated preference choice experiment theory and models (see Louviere et al., 2000) we omit a detailed accounting here, noting that the model specification for this application is a panel rank-ordered random parameters logit (see Lew et al., 2010 for further details on this model specification)[1].

WTP for species recovery was calculated over the distribution of species parameters using a simulation-based estimation procedure, following standard formulas for the measurement of economic values[2] derived from discrete choice models (Small and Rosen, 1981). Ninety-five percent confidence intervals were calculated following Krinsky and Robb (1986). WTP estimates and 95% confidence intervals were calculated for the national sample and embedded west coast region sample. To formally test whether the WTP estimates differed between samples we used a method of convolutions approach described by Poe et al. (2005) and employed by Giraud and Valcic (2004). Kolmogorov–Smirnov tests and *t*-tests were used to determine significant differences between the samples for non-choice task survey questions where responses are assumed to be categorical or linear, respectively. Statistical significance is reported at $p < 0.05$.

Results and Discussion

There were no significant differences between mean ages (national = 49.2, regional = 48.8), mean household size (national and regional = 2.7) and gender (national = 49.2% female, regional = 46.8%) for the national and regional samples. Median income range ($50,000–74,999) and median education level (some college) were the same for the national and regional samples, though the distributions for both demographic variables differed significantly between the groups (**Table 2**).

Significant differences exist between the samples in their familiarity with each species and their observation of each species in the wild (**Table 3**). Respondents were asked to indicate their familiarity with each species using a four-point likert scale ranging from "very familiar" to "not familiar at all" and whether they had personally observed the species in the wild, outside of zoos and aquariums. Response distributions between the national and regional samples differed significantly in their familiarity with the southern resident killer whale, Central California coast Coho salmon, Southern California steelhead, and black abalone. Respondents on the west coast were more familiar with these species than were respondents from the national sample. This is not surprising given the geographic proximity of these species to respondents on the west coast, which likely results in increased media exposure and opportunities to see the species in the wild as compared to respondents throughout the U.S. Similarly, more respondents from the west coast sample had observed these four species, as well as the humpback whale, in the wild.

Significant differences also exist between the national and west coast samples in the extent to which respondents felt their households would be affected by additional protection measures for the Central California coast Coho salmon and the Southern California steelhead, with respondents from the west coast sample stating they would be more affected than respondents from the national sample (**Table 4**). As Giraud and Valcic (2004) posit, those closest to the resource may bear a disproportionate share of the costs of species protection measures, though in our case the measurement scale does not differentiate between positive and negative effects on the household. Interestingly, for one protection measure for the southern resident killer whale involving increased efforts to prevent oil spills, significantly more respondents from the national sample stated they would be affected than did respondents from the west coast (For a full list of all protection measures and responses please see Supplementary Table 1).

The estimated parameters for the national and west coast choice models are presented in **Table 5**. The results meet our *a priori* expectations that improving any of the eight species to a recovered status is utility increasing (i.e., a positive sign on the

TABLE 2 | Respondent demographics.

	National (n = 10.637)	West coast[a] (n = 1742)
Mean age	49.2	48.8
Mean household size	2.7	2.7
ANNUAL HOUSEHOLD INCOME		
$5000–24,999	19.4%	20.8%
$25,000–49,999	26.2%	24.5%
$50,000–74,999	22.1%	20.1%
$75,000–99,999	14.2%	14.3%
>$99,999	18.1%	20.3%
EDUCATION		
Less than high school	7.7%	7.0%
High school diploma	20.8%	16.5%
Some college	32.2%	36.6%
Bachelors or higher	39.3%	39.9%
Female	49.2%	46.8%

[a]*Includes households in Washington, Oregon, California.*

[1]Since the focus of this article is on comparing general estimates of sample mean WTP for geographically-embedded samples, we do not attempt to explain how preferences and WTP vary across individuals in this work, beyond allowing the variation in preference parameters inherent in the random parameters logit modeling approach. However, we note that this is an important line of research, and extensions of the model specification used here (e.g., adding variables that interact individual demographics with attribute levels), as well as other modeling frameworks (e.g., latent class discrete choice models), can be employed to help explain variation in preferences and WTP across individuals.

[2]Survey respondents were asked questions to elicit their willingness-to-pay (WTP) for the recovery or down-listing of one or more threatened and endangered marine species, which represents an improvement from the status quo. As such, the SP questions measure compensating variation (CV), an economic measure of welfare change. Alternatively, equivalent variation (EV), another measure of welfare change, could have been obtained by asking for respondent's willingness-to-accept (WTA) compensation to forgo the improvement. However, we follow the majority of the literature in framing the SP questions to elicit WTP, and note that WTA estimates may not be equal to the WTP estimates reported here (Perman et al., 1996). For a more thorough discussion on the discrepancies between WTA and WTP, see Horowitz and McConnell (2002).

TABLE 3 | Percent of respondents familiar with species.

	Very familiar	Somewhat familiar	Not very familiar	Not familiar at all	Observed species in the wild (% yes)
NATIONAL/WEST COAST					
Hawksbill sea turtle	6/7	25/22	33/38	36/33	8/6
Southern resident killer whale*+	9/14	33/38	31/30	26/17	8/21
Central California coast coho salmon*+	4/12	21/32	28/24	47/33	5/14
Southern California steelhead*+	2/7	14/24	30/33	53/36	3/10
Humpback whale+	22/27	53/52	18/16	7/4	20/34
Elkhorn coral	2/2	13/16	28/29	57/52	8/9
Johnson's seagrass	<1/<1	7/8	27/29	65/62	4/4
Black abalone*+	3/7	14/20	35/39	48/33	4/12

Indicates significant difference in familiarity with species ($p < 0.05$) between national and west coast sample.
+Indicates significant difference in observation of species in the wild ($p < 0.05$) between national and west coast sample.

TABLE 4 | Species protection measures that differ* between national and west coast respondents.

		Not affected at all	A little affected	Somewhat affected	Very affected	Extremely affected	I am unsure
NATIONAL/WEST COAST							
Anadromous fish	Land use changes that increase protection of rivers where Central California coast coho salmon spawn	55/46	12/16	12/13	7/9	4/6	9/10
	Additional restrictions on agricultural pesticide and fertilizer use in areas around Central California coast spawning rivers to reduce pollution	47/40	15/18	15/15	8/9	6/8	9/9
	Better management of water released from dams to ensure sufficient water is available for Central California coast coho salmon to swim upstream	51/39	13/16	13/16	8/9	6/10	9/11
	Land use changes that increase protection of rivers where Southern California steelhead spawn	55/49	12/14	12/15	7/7	4/6	9/9
	Additional restrictions on sources of pollution in areas around Southern California steelhead spawning rivers	53/46	12/15	13/14	7/9	5/7	9/9
Whales	Increase efforts to prevent oil spills and other types of marine pollution that harm southern resident killer whales	39/45	15/15	18/18	10/7	9/7	7/6

Significant difference ($p < 0.05$) between national and west coast sample.

species parameter) for respondents in both the national and west coast samples. All parameter estimates for recovering a species are significant for both samples. Cost parameters are negative and significant for both samples, as expected.

WTP for each species' recovery and associated 95% confidence intervals were calculated as described above and reported in **Table 6** for both the national and west coast regional samples. No significant differences were found in recovery values for any of the species between the national and regional samples. For both samples, recovering the hawksbill sea turtle yielded the highest values, followed by southern resident killer whale and Elkhorn coral. Though we have not determined whether one species value is statistically higher (or lower) than another using the method of convolutions, any two species values with non-overlapping confidence intervals can be considered statistically different. It is also worth noting that the species that yielded the lowest recovery values—Johnson's seagrass, Central California coast Coho salmon, and humpback whale—all have an ESA-status of threatened, whereas the other five species are endangered. This may suggest that respondents are sensitive to the scope of the improvement, though statistical tests of scope sensitivity are beyond the focus of this paper (see Lew and Wallmo, 2011 for tests of scope sensitivity).

Conclusions

Our results demonstrate that recovering threatened and endangered marine species is economically valuable to the U.S. public. This should be of management and policy interest for several reasons. First, species value estimates can facilitate scenario analyses needed for coastal and marine spatial planning—an approach that is increasingly called for in U.S. ocean policy. For example, the Final Recommendations of the U.S. Ocean Policy Task Force (2009)[3]. require managers to

[3]Interagency Ocean Policy Task Force. (2009). *Final Recommendations of the Interagency Ocean Policy Task Force*. Available online at: http://www.whitehouse.gov/files/documents/OPTF_FinalRecs.pdf.

TABLE 5 | Parameter estimates from choice models.

Parameter	National	West coast
Johnson's seagrass_recovered*+	0.5630	0.6161
Central Ca. coast coho salmon_recovered*+	0.6563	0.8640
Humpback whale_ recovered*+	0.7831	0.9386
Elkhorn coral_improve_to_threatened	0.0357	0.0406
Elkhorn coral_recovered*+	0.9059	1.1658
Hawksbill sea turtle_improve to threatened*	0.1412	0.1913
Hawksbill sea turtle_recovered*+	1.0356	1.2987
Black abalone_improve to threatened*	0.0747	0.1607
Black abalone_recovered*+	0.8691	1.1054
Southern Ca. steelhead_improve to threatened*	0.1759	0.3439
Southern Ca. steelhead_recovered*	0.8254	0.9831
Southern resident killer whale_improve to threatened*	0.1044	−0.0041
Southern resident killer whale_recovered*+	1.034	1.3443
Cost*+	−0.0257	−0.0298

*Parameter significant ($p < 0.05$) for national sample.
+Parameter significant ($p < 0.05$) for west coast sample.

TABLE 6 | WTP* (95% CI) for species recovery for national and west coast samples.

Common name *Genus species*	National sample	West coast sample
Hawksbill sea turtle *Eretmochelys imbricata*	$85.95 (81.27–90.20)	$93.94 (79.26–108.49)
Southern resident killer whale *Orcinus orca*	$84.38 (79.15–89.69)	$89.83 (72.76–107.47)
Humpback whale *Megaptera novaeangliae*	$60.98 (57.47–64.52)	$63.15 (51.83–73.95)
Johnson's seagrass *Halophila johnsonii*	$43.83 (40.67–46.87)	$41.36 (33.08–49.44)
Central California coast coho salmon *Oncorhynchus kisutch*	$51.06 (47.59–54.67)	$58.16 (49.40–67.72)
Southern California steelhead *Oncorhynchus mykiss*	$71.06 (66.29–75.96)	$77.56 (63.58–90.54)
Elkhorn coral *Acropora palmata*	$71.78 (67.30–76.23)	$79.94 (68.12–92.19)
Black abalone *Haliotis cracherodii*	$70.50 (66.19–74.58)	$79.59 (65.45–93.52)

*Average annual household willingness-to-pay for 10 year.

consider the full suite of impacts—human and non-human—when designing policies that impact the ocean. Our value estimates provide economic benefit measures associated with actions that help recover or improve the status of a threatened or endangered species, thereby providing a more comprehensive account of the suite of benefits associated with particular policies. The estimates can also be useful inputs in standard benefit-cost models and ecological-economic models that inform ecosystem-based management (Sanchirico et al., 2013). In addition, value estimates for threatened and endangered species can be used in natural resource damage assessment cases and in recovery planning and critical habitat designation efforts.

In our examination of geographically embedded values for recovering threatened and endangered species, our results were unlike those of Giraud and Valcic (2004), as we found no differences between a national and west coast regional sample. It is not possible to determine whether our findings demonstrate distance decay, as we did not estimate a spatially explicit model. However, our results do show that on average recovery values for three localized U.S. west coast species and one species found from Northern California to Mexico were no different for national and west coast respondents. Likewise, recovery values from the west coast sample were no different than values from the national sample for Elkhorn coral and Johnson's seagrass, species found on the east coast and Caribbean Sea. These results may provide insight for the field of benefit transfer—the process of transferring a value from a study site to a policy site (Johnston and Rosenberger, 2010). In times of limited funding and financial constraints, the notion that values from small, localized samples are statistically similar to values from large scale national samples may help agencies in allocating their funds.

Our results do not support the concept that familiarity with a resource may induce higher values for the resource, nor do our results support the notion that those affected by measures undertaken to protect species may hold different values than those who are affected to a lesser extent or not at all. As Bateman et al. (2006) point out, spatial patterns observed in non-market values such as distance decay may depend on the type of good being valued. Perhaps in this case people believe that national wildlife should be managed as a public trust, intended not only for those in close geographic proximity to the resource but as a benefit for the entire country.

Although our research found no significant differences in WTP between geographically embedded samples, we did not test explicitly for different sources and types of preference heterogeneity (Boxall and Adamowicz, 2002; Wallmo and Edwards, 2008). Further research examining the effects of socio-economic variables (e.g., age, gender, income, and education) or other individual-specific characteristics on WTP could help identify opportunities to target specific policies and enhance the non-market valuation literature. In addition, while our research compares a national sample to only one region, our results support the concept that the economic jurisdiction for endangered species recovery includes the entire U.S. Future research comparing a larger array of geographically embedded samples, as well as explicitly testing for distance decay effects in species recovery values, would further inform this concept. This type of research is important for policy makers as it elucidates the extent of the summation of individual values when developing economically efficient policies for endangered species recovery.

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Supplementary Material

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fmars.2015.00055

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Recovering Pacific rockfish at risk: the economic valuation of management actions

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Over 35 species of rockfish are found along Canada's Pacific coast, some of which have been considered for listing under Canada's Species at Risk Act. We estimate Canadians' welfare for recovery of a representative Pacific rockfish species using referendum-style stated preference methods administered to a sample of the Canadian public via an internet panel. Hypothetical recovery programs were presented as options to a baseline of current management measures. The programs resulted in varying long term outcomes distinguished by species' future population projections. An increase in household taxes for a fixed 10 year period was employed as the proposed payment mechanism. The econometric analysis found positive and significant welfare measures for all management programs, as well as sensitivity to scope. Willingness to pay ranged from $48 to $180 per year per household depending on the recovery program valued. Welfare measures were found to differ significantly between those who believed their responses to be consequential and those who did not. The former provided measures that were significantly higher than the latter. We conclude with a discussion of the findings in relation to recent literature on consequentiality and incentive compatibility of stated preference questions.

Keywords: rockfish, welfare measures, consequentiality, species at risk recovery

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Introduction

The introduction of Canada's *Species at Risk Act* in 2002, and the regulatory requirements put forward by the Canadian Federal Government's *Cabinet Directive on Regulatory Management*, highlight the importance of accurate and complete benefit cost analyses (BCA) of species listing decisions. While cost information and data are often readily available, corresponding benefit information is not.

Over 35 species of rockfish are found along Canada's Pacific coast, of which nine have been considered for listing under the *Species at Risk Act*[1]. While numerous stated preference studies estimating non market values associated with environmental and species protection have been performed, to the best of our knowledge there are no studies addressing Canadian rockfish populations or similar species and associated recovery plans. The closest work we are aware of is that of Anderson et al, which examined the impacts of management programs involving partial and full recreational fishing area closures to benefit rockfish in Puget Sound on recreational fishers (Anderson et al., 2013). This study estimates Canadian passive use benefits

[1] Government of Canada, Species at Risk Public Registry. (Available online at: https://www.registrelep-sararegistry.gc.ca/).

associated with the protection and recovery of a representative rockfish species using a stated preference approach.

This paper also discusses the impact of perceived consequentiality on estimated welfare measures. The questionnaire presented hypothetical but realistic and possible management programs for recovering the rockfish species. Respondents who indicated they believed the results would influence marine policy programs were identified; their welfare measures were assessed both jointly and separately with the full sample. An issue of on-going interest in stated preference valuation is the assessment of the incentive compatibility of the survey instrument. Carson and Groves argued that if a respondent views survey results as potentially influencing an agency's actions, and the respondent is invested in the outcomes of those actions, standard economic theory applies. In this analysis we examine the impact of perceived consequentiality on welfare measures. Emerging literature suggests that this approach provides estimates of welfare measures that are incentive compatible (Vossler et al., 2012). Vossler et al. found a modest positive bias on WTP estimates was removed when respondents believed that the survey results would have more than a weak impact on policy. This is in seeming contrast to what Vossler and Watson found when comparing survey responses and real world referendum results on support of a conservation program to be funded by a property tax increase. Their results showed that an under-prediction of support for the program, and a negative bias on WTP, disappeared when respondents who did not believe their survey vote would be consequential were removed from the estimation (Vossler and Watson, 2013).

Thus, the contributions of the paper are empirical (presenting welfare measures associated with the recovery of a little-studied Canadian species), and methodological, (identifying a key sample segment to focus on as well as survey design insights). We also employ a novel empirical approach by jointly estimating the willingness to pay for conservation programs and the probability of a respondent believing that the result will influence policy.

Materials and Methods

Stated Preferences and Non Market Value Estimation

Without observable behavior related to the general Canadian populations' quantitative values regarding rockfish conservation, a stated preference approach was the sole estimation option. Economic values associated with rockfish conservation stem from a shift in an individual's utility due to the knowledge that a management program that benefits the continued existence of the species is in place. An individual's utility may change with the implementation of a management program that benefits the species due to a desire to use it in the future or bequeath it to future generations (Grafton et al., 2004).

Stated preference approaches ascertain values through questioning a respondent. Two established stated preference methods are contingent valuation (CV) and choice experiments (CE). CV is widely recognized as an established technique for valuing wildlife enhancements (Randall, 1997). Respondents choose between differing states of the world. This may include payment for an improved state of the world such as an increase in wildlife populations, indicating willingness to pay (WTP).

A variant of the traditional CV method was selected as the best option for this study due to a limited number of alternatives in practice, and the fixed nature of the attributes apart from cost within each of the alternatives. Only three management programs were valued. Attributes such as the survival outcome for the species and the increased catch restrictions stayed constant within each management program presented. In addition, the economic value of introducing programs, rather than the attributes of the species or programs, is the relevant policy benefit component. A referendum approach was selected, as strategic behavior has been shown to be less likely with the referendum approach (Jakobson and Dragun, 2001).

In a referendum-style valuation study, respondents indicate a preference by voting for one of two options presented. It allows for believable presentation, and data that can by analyzed through well-developed techniques. Since, obtaining empirical values suitable for socio-economic analysis was the primary objective of the research, employing an established measurement technique to a previously unmeasured good was thought to be the best approach.

Survey Instrument Design

Central to the study was administration of a questionnaire containing qualitative, quantitative and program attribute-based stated preference questions. The questionnaire was developed over the years 2009–2010 with the aid of focus groups and pilot studies.

Background, Baseline and Scenario Projections

Development of the survey instrument involved collaboration with species experts to provide an accurate picture of the attributes of a representative Pacific rockfish population, and the characteristics including impacts of management programs. Information provision in stated preference surveys is a challenge, as respondents must be given sufficient information to make a meaningful decision while staying within a manageable survey length and without biasing their choices. The survey instrument specified that there were over 35 species of rockfish widely dispersed geographically along the Pacific coast, and stated that the management programs being valued were to benefit a single representative rockfish species. Descriptions of other species listed under the *Species at Risk Act* were included, in an effort to ensure respondents considered that other species may also require management programs when indicating whether they would vote for management programs benefitting the rockfish species.

Central to the welfare estimates were the baseline and scenario projections. The respondents were asked to choose between what was described as the "current management scenario" in which the species continued on its present population trajectory, with no new management measures introduced and no additional costs to the respondent, and a "proposed management scenario" with a reduced total allowable catch (TAC) levels, and costs.

Each management scenario had an associated distinct species population trajectory.

Respondents were asked how they would vote on two options. The first, the current management option (baseline), had no new management measures introduced, no additional costs, and the species would be endangered in 40 years. The alternative option was one of three new management programs. Each new program had an additional cost, which was selected at random from \$1, \$10, \$50, \$150, \$300, \$600, annually for 10 years. See **Table 1** for a management program summary. Bids were selected to have a relatively large proportion choosing the program at the lowest price, and a relatively small proportion choosing the program at the highest price. However, we also recognize that bid levels must be credible to respondents and probed on this issue in the focus groups and in debriefing questions. Nonetheless, bid design is a concern in all stated preference studies.

The three programs possessed varying activity restrictions with corresponding socio-economic impacts, of which respondents were informed. The restrictions and impacts corresponded with species improvement 40 years into the future. This included a description of likely job losses under certain management programs. While the description was included for transparency, respondents were told that compensation would occur to ensure that respondents' passive use values for Pacific Rockfish were being measured as opposed to their values related to jobs. Species improvement was described using the classifications under Canada's *Species at Risk Act* (Extinct/Extirpated, Endangered, Threatened, Special Concern, Not at Risk). Respondents saw a definition of each classification as well as an example of a species corresponding to each. This served a dual purpose. It ensured respondents understood the definitions while reminding them that other species also face difficulties. Each respondent compared the three programs to the status quo and voted (as in **Figure 1**). The order in which each respondent saw the programs was randomized in the administration of the instrument.

Focus Groups

Four focus groups were conducted using the questionnaire to ascertain the suitability of the instrument and the comprehension of the provided information by potential respondents. The groups were held across Canada to reduce the possibility of regionally specific issues, and included between 9 and 12 randomly recruited participants using random digit dialing telephone recruitment. A challenge with the topic of the study is potentially the low knowledge level respondents may have on the survey topic. Focus group discussions and responses to a number of the questions indicated that many Canadians were unfamiliar with the *Species at Risk Act*, as well as rockfish species themselves. To present the complex issues while staying within a manageable questionnaire length required a delicate balance. The focus groups provided direct feedback on areas they felt required more detailed information to allow them to make a decision, and which areas could be abbreviated. Starting points for WTP value estimates, necessary for contingent valuation questions, were also sought. Following each focus group the survey instrument was updated and refined.

Socio-demographic questions addressing age, gender, marital status, and location were also collected. Respondents were asked if they or their family members were involved in the fishing industry and whether they belonged to an environmental organization. This individual specific information was needed for understanding heterogeneity in the responses to the referendum questions. From a policy perspective assessing distributional impacts of the proposed management plans may be necessary. The results from these questions were assessed through statistical summaries of the various variables as well as their inclusion in econometric models of voting behavior.

Pilot Tests

Once the focus groups and adjustments were completed, the questionnaire went through two pilot tests using a combined total of 469 respondents. These pilot tests allowed for further calibration of the WTP values included in the contingent valuation questions. Ultimately, the survey was finalized with some minor changes, including an increased bid range and a simplified experimental design.

Addressing Hypothetical Bias

Steps were taken to minimize potential hypothetical bias in stated preference responses. Four different strategies were employed: (1) a cheap talk script was included in the survey before the choice

TABLE 1 | Management programs presented to respondents.

Program 1	Program 2	Program 3
This Rockfish is still allowed to be caught through incidental catch	This Rockfish is still allowed to be caught through incidental catch	This Rockfish is still allowed to be caught through incidental catch
Catch level stays the same	Catch level would be reduced by 33%	Catch level would be reduced by 66%
Catch levels of other species in the trawl and hook and line fleets will be reduced by 5%	Catch levels of other species in the trawl and hook and line fleets will be reduced by 20%	Catch levels of other species in the trawl and hook and line fleets will be reduced by 45%
A small amount jobs and income will be affected. Those affected will be compensated through a separate process that includes a variety of programs	A moderate amount jobs and income will be affected. Those affected will be compensated through a separate process that includes a variety of programs	A large amount jobs and income will be affected. Those affected will be compensated through a separate process that includes a variety of programs
Species would be threatened in 40 years	The species would be special concern in 40 years	The species would be not at risk in 40 years

Please indicate which option you would vote for if there was a national vote (referendum) on managing this species:

	Current Management Option	*Proposed Management Option*
Strategy for protection	No new regulations	Program 3
Listing status (in 40 years)	In 40 years the listing status for this species will be:	In 40 years the listing status for this species will be:
	Endangered	Not at Risk
Probability of extinction (in 40 years)	Very High	None
Increased cost to your household in extra taxes every year for 10 years	$ 0	$

Please remember this is a cost estimate for the program. The range of costs may vary depending on economic conditions and other factors.

The increased taxes would be used to monitor and enforce fishermen's and fishery compliance with the recovery programs

FIGURE 1 | Survey question example comparing Program 3 to the status quo.

questions that asked respondents to make choices as if these were real transactions; (2) multiple voting scenarios in randomized order were given to each respondent; (3) follow up questions on respondents' level of certainty regarding their votes were included and uncertain responses were identified; and finally (4) additional follow up questions designed to identify strategic voters were included following elicitation of vote choices.

Cleaning the data involved identification of speeders, protest votes and yea-sayers. Speeders are respondents who race through a survey without considering the questions, their main objective being to complete the survey as swiftly as possible. As panel members receive reward points from the research company for each survey they participate in, there is some incentive for such behavior. Given the survey length, and the time focus group members took to complete the survey, respondents who finished the survey in less than 5 min were deemed speeders and removed from the data set.

Protest voters are respondents who vote "no" as a way to make a point. In effect they give a $0 WTP value, despite possessing a positive WTP. In the case of an issue such as rockfish conservation, they may be protesting government interference or tax increases. The referendum-style stated preference survey is thought to be potentially incentive compatible as respondents are limited in their opportunities to over- or under-estimate their WTP. The realistic and familiar voting format was designed to reinforce the need for realistic votes to the respondents, while the wording of the survey encouraged truthful responses and attempted to avoid inflammatory terms. However, the possibility of protest bids is a concern in stated preference studies. As such, follow up questions designed to identify such responses were included.

The follow up questions came immediately after the CV questions. Respondents were asked to "indicate the most important reason for voting the way [they] did." Multiple reasons were presented; respondents could also select "other." Respondents who voted yes for all three programs and indicated their reason to be they felt "species at risk should be protected at any cost" were classified as yea-sayers and removed from the data set prior to modeling (Blumenschein et al., 2008). Respondents who voted for the management program but indicated uncertainty regarding their choice were classified as no votes, as this has been shown to reduce hypothetical bias (Carson and Groves, 2007).

Respondents were also asked "to what degree [they] thought [their] votes would influence management programs chosen for the species," to help identify in part if the criteria of consequential survey questions was met. Those indicating a strong or very strong degree of impact were classified as believing the survey to be consequential. Vossler et al. (2012) identified the importance of perceived consequentiality on the part of the respondents, and the possible merit of including survey questions allowing researchers to control for it (Vossler et al., 2012). Following development of a game theoretic framework to analyze the incentive properties of discrete choice experiments they conducted a field experiment which showed a modest positive bias of WTP estimates was no longer present when respondents had a more than weak belief in the consequentiality of their responses. Note that it is still possible that respondents who considered the survey influential may feel that it may influence the policy (policy consequentiality) but not their tax payments (tax consequentiality).

Survey Administration

The questionnaire was administered online as this allowed a significant volume of information to be compressed into a more digestible format for respondents. Many color rich diagrams were included, as well as pop up definitions where necessary. The survey was completed by 1242 individuals out of 2215 sent invitations, for a response rate of 56%. The survey sample was drawn from an internet panel of over 100,000 individuals maintained by Ipsos-Reid, and designed to be representative of the Canadian population based on a range of demographic characteristics. The panel required respondents to previously opt-in, and as such there may be inherent challenges associated with the representativeness of such panels (Government of Canada, 2006).

Data Modeling and Value Estimation

Stated preference data modeling techniques assume individuals make utility maximizing choices, and that their choices reflect their personal constraints such as time or income. For the CV data from this study, an individual *j*'s utility for program *i* can be written as:

$$u_i = \propto + \beta P_i + \gamma H_j + \delta \left(y_j - C_i\right) + \varepsilon,$$

where u represents the respondent's indirect utility for program i, P is a vector of program attributes, H is a vector of individual and household characteristics of respondent j, y represents respondent j's household income, and C is program cost. The error term ε represents factors that affect an individual's utility but are unknown to the researcher. The utility of the status quo of no management program is represented by $\propto$, the coefficient γ represents the effect of household characteristics on utility of program selection, and β represents the coefficients for the marginal utility of each program vs. the status quo. Finally, the marginal utility of money is represented by δ.

When the individual chooses between a new management program and the status quo, it is akin to a vote for or against the new program. The respondents indicated whether, if faced with a referendum, they would choose the new management program complete with increased cost in the form of higher per household income tax (a "yes" choice), or the current management program with no increased cost (a "no" choice).

To estimate WTP for each management program, u_{ij} represents respondent j's utility from management program *i* and u_{0j} represents respondent *j*'s utility from the status quo of no new management program. Assuming the first model from **Table 3** with utility dependent solely on income and a management program M_i, WTP is equivalent to the amount of household income that would need to be taken away from respondent j if management program i is implemented to keep respondent j's utility at the same level. Then $u_i\left(\gamma - WTP_j, M_i\right) = u_0(\gamma, M_0)$. Substituting the indirect utility function yields: $\alpha_i + \delta\left(y_j - WTP_j\right) + \varepsilon_{ij} = \alpha_0 + \delta y_j + \varepsilon_{0j}$
which gives:

$$WTP_j = \frac{\alpha_i - \alpha_0}{\delta} + (\varepsilon_{ij} - \varepsilon_{0j}).$$

Normalizing the utility of the status quo to 0, and assuming the difference in error means is equal to 0, gives $\mathrm{E(WTP)} = \alpha_i/\delta$.

To examine the possibility of a correlation between respondents' choices and their perceptions of survey consequentiality, a bivariate probit model was employed. This approach allowed for two equations to be estimated with correlated error disturbances. The first equation was an individual's choice between a new management program and the status quo dependent on program attributes, $y_1 = \beta P_i + \cdots + \varepsilon_1$; and the second involved whether the respondent perceived the survey as consequential dependent on individual and household characteristics of respondent j, $y_2 = \gamma H_j + \cdots + \varepsilon_2$.

Results and Discussion

Population Representation

Key demographic characteristics were examined including age, gender, marital status, and income, to assess how the survey sample corresponded to the Canadian population as reflected in the 2006 Canada Census results (see **Table 2**). This is an important gauge of how representative the survey results were of the population of Canada.

The gender distribution for the survey sample was 50.89% male and 49.11% female, compared to distribution found through the census of 48.95% male and 51.05% female. No respondents selected other or prefer not to answer. The age distribution of the survey sample differed somewhat from the most recent census distribution, with a slight over-representation of the 45–69 category, and a slight under-representation of the youngest and oldest populations. There are a number of probable reasons for this. The census groups 15–19 year olds together, while the survey was not administered to anyone under 18. As such the census would be expected to have a significantly higher percentage of people in that category than is found here. The under representation of the highest age groups (those 70 years and older), may be due to lower levels of computer use in that age group (Statistics Canada Report, 2007). Marital status of the survey respondents (30.0% single, 56.0% married, 14.0% domestic partnership) was closely reflected the Canadian population (27.6% single, 58.4% married, 14.0% domestic partnership). Household income was divided into seven categories. The category with the largest discrepancy between survey respondents and the Canadian population was that of household income <\$20,000, with 9.90% compared to 6.45% respectively.

TABLE 2 | Comparison of survey demographics with the 2006 Canadian Census.

Characteristic	Survey	Census
Male (%)	50.89	48.95
AGE BY RANGE		
18–19*	1.82%	8.22%
20–24	6.90%	7.99%
25–29	6.45%	7.63%
30–34	7.84%	7.76%
35–39	5.27%	8.48%
40–44	7.81%	10.03%
45–49	12.17%	10.07%
50–54	16.62%	9.06%
55–59	13.47%	8.01%
60–64	10.84%	6.11%
65–69	6.45%	4.74%
70–74	2.45%	4.05%
75–79	1.18%	3.38%
80–84	0.64%	2.48%
>85	0.09%	2.00%
MARITAL STATUS		
Single	27.60%	30.00%
Couple	58.40%	56.00%
Other	14.00%	14.00%
INCOME BY RANGE		
<\$20 000	9.90%	6.45%
\$20,000–\$39,999	18.36%	17.56%
\$40,000–\$59,999	18.92%	19.23%
\$60,000–\$79,999	15.62%	17.25%
\$80,000–\$99,999	12.24%	13.41%
\$100,000–\$124,999	11.67%	11.00%
\$125,000–\$149,999	5.88%	6.14%
>=\$150,000	7.41%	8.96%

**Survey age range 18–19 years, census age range 15–19 years.*

Background Questions

The survey contained background questions on the *Species at Risk Act* and fishing industry involvement. These questions were included to allow researchers to evaluate impacts on WTP, and potentially identify heterogeneous effects of management programs on individuals or groups.

Familiarity with the Species at Risk Act

This survey asked respondents their level of familiarity with the *Species at Risk Act*, to allow researchers to gauge the knowledge levels in the sample population. Respondents chose from one of three responses: very familiar, somewhat familiar, and not familiar. 1.449% and 27.29% of them said it was very familiar and somewhat familiar respectively, while 71.26% answered that they were not familiar with it at all. The low proportion of respondents identifying themselves as very familiar indicates that for most of the respondents, the majority of their *Species at Risk Act* knowledge will have come from the background information included in the survey.

Fishing Industry Involvement

The survey asked whether respondents or any members of the respondents' households presently or had previously worked in fishing-related industries, including processing plants, recreational fishing charters/tours, or commercial fishing or harvesting. Respondents could also answer none of the above or prefer not to answer. The majority of respondents (96.7%) chose none of the above; 0.5% indicated they or family members currently work or had previously worked in commercial fishing or harvesting; 1% indicated the same for recreational charters/tours; and 1% for aquatic species processing plants. The remainder of respondents preferred not to answer. This level of industry and direct involvement with the species amongst respondents suggests that the values found will be largely passive use in nature.

Modeling

Binary Probit Model

Probit models were used in the analysis of the choice questions to develop estimates of WTP values for programs by specific outcome. Each of the respondents saw three choice questions representing three possible program outcomes. These were contrasted with a status quo outcome representing the current management actions and forecast outcome. The respondents indicated whether, if faced with a referendum, they would choose the new management program complete with increased cost (a "yes" choice), or the current management program with no increased cost (a "no" choice). The order of questions was randomized between respondents. As well, each of the three programs had six possible costs, one of which was randomly assigned to each question.

We developed five probit models to estimate Canadians' willingness to pay for management programs benefitting a Pacific Rockfish based on the utility function described before. The tax increases associated with the management programs was coded as a continuous variable. Household income and age were converted from an ordinal scale to a continuous variable using the midpoints of categories. Whether or not the respondent works in the fishing industry, is a member of an environmental organization, has children, and their gender, were modeled as indicator variables. The models assessed the likelihood of respondents voting for a management program with one of three improved outcomes for the species over 40 years vs. the status quo of no new management measures. Outcomes of "Threatened" and "Special Concern" were included as indicator variables, with the outcome of "Not at Risk" reflected in the constant.

Econometric results of the five models are found in **Table 3**. The first four models contained cost and program outcomes as independent variables. Model 1 was estimated with the first vote of each respondent. Model 2 was estimated with the first vote results of respondents who believed the study would have an impact on policy. The first and second models used a binary probit model. Model 3 was estimated with results from all votes from respondents who indicated they believed the study would have an impact on policy. Model 4 was estimated with results from all three votes. Model 5 was estimated with all vote results, and includes additional socio-demographic characteristics.

Given that the standard binary probit model treats each vote as an independent observation, biased standard errors of the coefficients may result when each respondent provides several votes (Guilkey and Murphy, 1993). To alleviate this, the three models which included multiple votes used a random effects structure for the error components. Using this approach, two independent components make up the error term. One represents an unobservable characteristic for each individual, while the other varies both for individuals and votes.

Parameters from all five models were used to estimate positive and significant WTP values for management programs benefiting a Pacific rockfish species (see **Table 4**). WTP was estimated as an

TABLE 3 | Estimates of probit regression parameters.

	Probit first choice		Random effects probit all choices		
	Model 1	**Model 2 Yes to influence**	**Model 3 Yes to influence**	**Model 4**	**Model 5**
Constant	0.6037***	0.9357***	1.2565***	0.9471***	0.3871**
	(0.0815)	(0.2510)	(0.2091)	(0.0697)	(0.1828)
Cost	−0.0033***	−0.0041***	−0.0055***	−0.0058***	−0.0057***
	(0.0002)	(0.0007)	(0.0006)	(0.0002)	(0.0002)
Threatened	−0.1978**	−0.3058	−0.5967**	−0.4487***	−0.4482***
	(0.0993)	(0.2960)	(0.2441)	(0.0726)	(0.0727)
Special concern	−0.0748	−0.0387	−0.0453	−0.1941***	−0.1958***
	(0.1002)	(0.2941)	(0.2182)	(0.0725)	(0.0727)
Children	–	–	–	–	0.0610
					(0.1012)
Male	–	–	–	–	0.0226
					(0.0920)
Household Income	–	–	–	–	0.0047***
					(0.0010)
Age	–	–	–	–	0.0028
					(0.0033)
Fish industry	–	–	–	–	0.3716
					(0.2649)
Enviro Org	–	–	–	–	0.5556***
					(0.1917)
Rho	–	–	0.6225***	0.5560***	0.5418***
			(0.0737)	(0.0264)	(0.0271)
Log Likelihood	−624.55	−70.396	−205.88	−1678.09	−1783.58
Number of votes	1097	134	402	3291	3291
Number of individuals	1097	134	134	1097	1097

****Significant at 1% level.*

***Significant at 5% level.*

TABLE 4 | Summary of WTP estimates per household per year for 10 years found through random effect probit models with standard errors.

	Probit first choice		Random effects probit all choices		
	Model 1	Model 2 Yes to influence	Model 3 Yes to influence	Model 4	Model 5
Species outcome: Not at Risk	\$180.32	\$228.23	\$229.41	\$164.01	\$126.35
	(21.59)	(54.79)	(33.91)	(10.46)	(31.33)
Species outcome: Special Concern	-	-	-	\$130.49	\$92.26
				(11.39)	(30.74)
Species outcome: Threatened	\$121.23	-	\$120.47	\$86.47	\$48.32
	(20.77)		(39.04)	(10.85)	(31.59)

annual per household payment for 10 years. Welfare estimates for Model 1 were significant for population improvements from endangered to threatened and endangered to not at risk. A recovery level of special concern was not significantly different from not at risk. Model 2 welfare estimates did not show sensitivity to scope; WTP values were positive to attain a species outcome better than endangered, but did not significantly differ across outcome levels. Model 3 welfare estimates showed some sensitivity to scope with willingness to pay differing between species outcomes of threatened and not at risk, but not differing between special concern and not at risk. Models 4 and 5 (with the largest samples) showed the greatest sensitivity to scope, with statistically significant differences in welfare estimates across all recovery levels.

Model 5 also introduced respondent socio-demographic characteristics, recognizing that the associated coefficients may not be consistently estimated because of potential endogeneity. We include this model to assess correlations between demographic characteristics and program choices. Respondents' household incomes, and if the respondent belonged to an environmental organization, were found to have statistically significant impacts on respondents' management program choices. WTP was found to increase by \$0.82 (with a standard error of 0.17) for each \$1000 increase in household income. As such, WTP by program was estimated using the mean household income of \$72,000. A value of \$126 (with a standard error of 31) to avoid the species status being endangered in 40 years and instead achieve a species status of not at risk was found. WTP values of \$48 (with a standard error of 32) and \$92 (with a standard error of 31) were found for threatened and special concern outcomes as opposed to an endangered outcome respectively. In addition to household income, if the respondent belonged to an environmental or conservation organization their WTP was estimated at an additional \$97 for any of the three program outcomes (see **Table 4**).

Bivariate Probit Model

Comparing the modeling of a subsample of respondents who indicated they viewed the surveys as consequential, as well as the full sample of respondents, the binary probit models and random effects probit models identified differences in WTP estimates between the two groups. WTP estimates were measurably higher for the consequential subsample. This is in contrast with the findings of Vossler et al., whose field experiment found that when respondents both believe their decisions are consequential and the information gathered will be used in such a way that maintains choice set independence and one-to-one matching between management projects, a modest positive bias in WTP estimates is removed. Rather these findings are in line with Vossler and Watson's later paper which compared survey and referendum results, and found an under-prediction of yes votes by the survey which disappeared when only respondents who believed the survey results to be consequential were examined.

To further examine the question of whether perceived consequentiality may be correlated with WTP, a bivariate probit model was employed. The first equation in the model matched that of models 1–4, with vote as the dependent variable and cost and program outcomes as the independent variables. For the second equation a binary variable equal to one if the respondent indicated they believed the survey would have an influence on policy making (was consequential) was the dependent variable, and gender, age and if the respondent belonged to an environmental organization were independent variables. All parameter estimates were significant at the 1 or 5% levels. The disturbance correlation was significant at the 1% level, indicating the likelihood of a respondent voting for a proposed management program is related to the respondents' perception of consequentiality (**Table 5**).

Model 6, Equation (1) parameter estimates found positive and significant WTP values for management programs benefitting a Pacific Rockfish at risk. Scope effects are reflected in the significant differences between WTP values for management programs resulting in varying species recovery level (**Table 6**). Model 6, Equation (2) parameter estimates indicate that the likelihood of a respondent believing that the survey responses would influence management programs for the species increases if the individual is male, or if the individual belongs to an environmental organization. In contrast, the likelihood is negatively correlated with age. Employing the bivariate probit approach does not result in statistically significant differences in WTP estimates, however it modestly improves the efficiency of the parameter estimates.

Conclusion

The results presented in this paper provide benefit estimates for the implementation of a range of management programs benefiting a rockfish species on Canada's Pacific coast. These

TABLE 5 | Bivariate probit model.

	Model 6
EQUATION 1: VOTE	
Constant	0.6047***
	(0.0464)
Cost	−0.0037***
	(0.0001)
Threatened	−0.2970***
	(0.0589)
Special concern	−0.1159**
	(0.0577)
EQUATION 2: YES INFLUENCE	
Constant	−0.9953***
	(0.0985)
Male	0.2663***
	(0.0579)
Environmental Organization	0.5296***
	(0.1044)
Age	−0.0074***
	(0.0020)
Log likelihood	−2999.6130
RHO (Disturbance Correlation)	0.1291***
	(0.0383)
Number of votes	3291
Number of individuals	1097

****Significant at 1% level.*
***Significant at 5% level.*

TABLE 6 | WTP estimates per household per year for 10 years found through bivariate probit model.

	Model 6
Species outcome: Not at Risk	$162.91
	(10.58)
Species outcome: Special Concern	$131.70
	(11.25)
Species outcome: Threatened	$82.90
	(11.93)

types of values are necessary for cost-benefit analyses undertaken for regulatory and legislative decisions on the implementation of such programs in Canada. The management programs involved actions and restrictions resulting in improved species status as defined by Canada's *Species at Risk Act*. The *Species at Risk Act* definitions were used for established species status reference points, however the study examined management actions with associated species outcomes rather than the listing or not listing of species under the Act. The economic values Canadians place on three management programs, each resulting in an improved status of the species from "endangered" to either "threatened," "special concern" or "not at risk," were estimated. Care was taken to specify in the survey instrument that the management program was directed at one of over 35 rockfish species. Despite this, it is still possible that a number of respondents interpreted the management programs as benefitting more than one species.

When designing the survey we sought to ensure the programs valued were possible and realistic, including their projected outcomes for the species, albeit hypothetical. The commercial fishing industry has demonstrated success reducing the total catch of some groundfish species in the Pacific groundfish fishery, including rockfish species, while maintaining a viable multi-species fishery of other species (Fisheries and Oceans Canada, 2009). Continual monitoring of what species are being caught where, and close communication between fishers about the locations of species to be avoided, is central to this effort. This indicates that management programs such as those presented here will not necessarily benefit other species. Future, valuations should be program-specific, and take into account all anticipated spillover benefits.

The range of household WTP values found indicates that respondents were sensitive to scope. The respondents were more willing to pay more for greater degrees of species improvement. This suggests that despite the majority of respondents indicating they were not familiar with the *Species at Risk Act*, they were able to grasp the concepts and understand what they were voting on. The finding that household income was significant in Model 4 further supports credibility, as it points to respondents considering their ability to pay when voting. Additional research on the relationship between WTP values and responses to consequentiality questions, including the assessment of endogeneity and the impact of various forms of consequentiality questions, should be considered given the finding of higher WTP values for respondents who believed the study would influence policy, and the significant correlation between consequentiality and the likelihood of voting.

Study Approval

Ethics approvals for the survey design and data analysis were obtained from the University of Alberta Research Ethics Board PER-ALES-NS, project numbers Pro00010526 and Pro00025927.

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Navigating benefit transfer for salmon improvements in the Western US

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A perennial problem in environmental resource management is targeting an efficient level of resource provision that maximizes societal well-being. Such management requires knowledge of both costs and benefits associated with varying management options. This paper illustrates the challenge of estimating the benefits of an improvement in a marine resource when secondary data must be used, and when total economic benefits include non-use values. An example of non-use values is existence value, which is not contingent on resource extraction nor recreational activities. State of the art techniques for adapting secondary data, or "benefit transfer," are reviewed in the context of increasing anadromous salmon for an example Western US policy scenario. An extensive summary of applicable primary studies is provided, compiling observations from several studies surveying several thousand Western US households. The studies consistently indicate a high willingness to pay for increased salmon abundance. Analytical techniques for transferring data are described, with calculation examples using published tools, focusing on meta-regression and structural benefit transfer. While these advanced benefit transfer tools offer perspective on benefits beyond what can be learned by relying on a single study, they also represent a variety of challenges limiting their usefulness. While transparently navigating these issues, a monetized estimate of increased salmon for the policy case is provided, along with discussion on interpreting benefit transfer techniques and their results more generally. From this synthesis, several suggestions are also made for future original salmon valuation studies.

Keywords: salmon, meta-analysis, preference calibration, structural benefit transfer, non-use value

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Introduction

In the Western US, migratory salmon are iconic symbols of nature's strength and bounty. However, wild salmon stocks have precipitously declined in the last century (Nehlsen et al., 1991). Some populations may be on the brink of extinction, already the fate of hundreds of West Coast evolutionarily significant units (Gustafson et al., 2007). Numerous anthropogenic stressors have played a role, such as dams, overfishing, hatchery practices, and multiple forms of habitat degradation (Stouder et al., 1997; Lackey et al., 2006). More recently, negative impacts from climate change have also been recognized (Doney et al., 2012).

As salmon losses continue, policymakers are increasingly called upon to consider ways of mitigating impacts and/or promoting salmon recovery. Indeed, numerous state and federal policies are oriented toward this goal, such as Total Maximum Daily Loads for pollutants such as temperature, and critical habitat designations under the Endangered Species Act. However, regulations do not guarantee a given salmon stock will persist, nor is simple survival necessarily the sole objective. Often the debate over different options involves economic questions—i.e., benefits and costs. For example, an entity responsible for dam operations may be making tradeoffs between flood control, hydropower generation, water availability for agriculture, reservoir recreation, and migratory fish. Similarly, planning agencies confront tradeoffs between development proposals and environmental mitigation strategies of various costs. While not every decision context requires monetized environmental impacts, it does allow resource management options to be compared in a common unit. This can be helpful at multiple levels of governance, with established use of such information at the US federal level (Weber, 2010; Lipton et al., 2014). The direct interpretability of benefit and cost information also facilitates public discussion on issues such as salmon recovery, rather than simply "leaving it to the experts," or special-interest lobbyists. Typically, costs of environmental protection are better characterized than benefits. If benefits are not represented, there is cause for concern that protection efforts will be sub-optimal.

How can the benefits of salmon be estimated? This difficult problem is the focus of the paper. First, the case study literature on the "total economic value" of changes in abundance of Western US salmon will be synthesized, including particularities of each study. Second, challenges in utilizing these data (in conjunction with other available literature) will be illustrated in the course of conducting "benefit transfer" for a new Western US example context. Several benefit transfer methods are applied, insights and pitfalls that arise are documented, and the range of results is discussed. While the paper revolves around a single policy case, the discussion is designed to make benefit transfer techniques more accessible for those seeking to apply or develop the tools more generally. Finally, based on lessons learned, suggestions are made for future salmon valuation studies, both for more robust case studies and improved benefit transfer capacity from them.

Valuing societal impacts from changes in salmon proceeds from recognizing various pathways of human benefit. Some benefits are relatively obvious, such as resource use and extraction in the market economy, e.g., commercial fish harvest, and revenue from fishing-related expenditures. A less recognized but important dimension are nonmarket benefits, such as the recreational enjoyment of a fishing experience. An angler may contribute only minimally to a local economy through the act of fishing—yet the opportunity to engage in this pastime may be of extraordinarily high value to that individual. By studying recreation behavior analysts can construct a demand curve for recreational fishing for a given site or a site network, and estimate the monetary value per day of the enjoyment associated with an angler-day, as well as monetary impacts from site closures or fish abundance changes. Such nonmarket environmental amenities are an important dimension of natural resources management, and have been referred to as a "second-paycheck." For example, a person may be willing to accept less income in order to live near particular environmental amenities (Power and Barrett, 2001).

Yet human appreciation of natural resources such as salmon goes deeper still. For decades environmental economists have recognized an important category of benefits known as non-use values (e.g., Krutilla, 1967; Johnston et al., 2003). Essentially, resources may be valued without the necessity of direct experience. Notions of value predicated on resource extraction, harvest, and even nonconsumptive recreational use are overly limiting. Categorically neglecting non-use values can lead to significant underestimates of public welfare (Freeman, 2003: p. 138). The evidence for non-use values comes from survey research, in which respondents have consistently demonstrated a "willingness-to-pay" (WTP) to protect or increase environmental amenities even when there are essentially no resource use opportunities. Non-use values have enormous potential importance for managing environmental resources for the best benefit of society. The total economic value (TEV) conceptual framework helps maintain attention on the diverse components of value potentially associated with changes in a natural resource: market as well as nonmarket values are included, with nonmarket values including both use and non-use (US National Research Council of the National Academy of Sciences, 2004).

The survey-based methods that allow insight into both the use and non-use components of TEV are known as contingent valuation and choice experiments (Mitchell and Carson, 1989; Louviere et al., 2000). Numerous such "stated preference" valuation survey studies exist (e.g., tallied by Carson, 2000). However, only a few feature Western US salmon, despite their high-profile role in historical and contemporary culture. To address this problem, methodologies of benefit transfer can be employed to apply valuation results from prior relevant studies to a new context. Benefit transfer has received substantial academic attention. Notable milestones are edited compendiums: Brookshire and Neill (1992); Florax et al. (2002); Wilson and Hoehn (2006); and Navrud and Ready (2007). For a summary of the recent literature, see Johnston and Rosenberger (2010). Compared with an original study, benefit transfer is usually viewed as second best or even a last resort. This is tempered by acknowledgment that environmental decisions need guidance more often than valuation studies can be marshaled. Furthermore, methodological idiosyncrasies and biases associated with any single study are dampened when placed in context of additional observations. Benefit transfer will continue; more awareness of the techniques including their weaknesses will aid both analysts and those interpreting the resulting monetized estimates.

Methods

While there is no single way to conduct benefit transfer, counsel is found in multiple sources, e.g., Brouwer and Spaninks (1999), Nelson and Kennedy (2009), Johnston and Rosenberger (2010), and US Environmental Protection Agency (2010). A general three-step outline for benefit transfer follows: describe the policy case; select study cases; and transfer values (US Environmental Protection Agency, 2010). To satisfy the first step, in the next section we describe an illustrative policy case of the Willamette

Valley, Oregon, although certainly many more policy cases are possible. Regarding step two, we review the context and background for various available study cases to gauge similarities with the policy case, and to address any potential study quality or bias issues. For step three, several benefit transfer methods will be explored in turn:

- Transfer a point value from a single study,
- Transfer with the aid of a study's valuation function,
- Apply an existing meta-regression,
- New meta-regression, and
- New structural benefit transfer.

An Illustrative Policy Case

To illustrate benefit transfer methods, as well as provide management insight in a specific case, this paper will estimate the TEV of an increase in Spring Chinook for the Willamette Valley, Oregon. The valley is an 11,704 sqmi watershed in northwest Oregon, draining a north-trending valley between the Coast Range to the west, and the Cascades to the east. The basin has a rapidly growing population, currently home for nearly three million people. This encompasses most of Oregon's population, despite the valley representing only about 10% of the state's total area. Significant tourism occurs in the region, attracted by recreational, scenic, and cultural amenities. Urban and exurban areas in the river valley share space with agricultural lands, timberland, and natural areas. With the watershed size, human population, and diverse land use, environmental policymaking processes are complex, similar to multi-use contexts found in many other watersheds.

The wild salmon run of the entire Columbia River of the late 1990s was estimated to be less than 2% of runs in the late 1800s (Gresh et al., 2000), allowing some inference as to the decline for the Willamette as a Columbia subwatershed. Sheer and Steel (2006) estimate over 40% of salmon habitat in the Willamette and lower Columbia watersheds has been lost due to dams alone, the majority of which was higher-quality upland habitat. Spring Chinook comprise by far the largest salmon run in the Willamette, although there are Coho and Winter Steelhead runs as well. All three are threatened under the US Endangered Species Act [US National Oceanic and Atmospheric Administration (NOAA), 2010]. Recovery plans are in process by the Oregon Department of Fish and Wildlife (ODFW) for Spring Chinook and Winter Steelhead, and critical habitat for Coho has been proposed by NOAA as of 2013. These listings do not guarantee recovery, and proposed goals go beyond simply preventing extinction. For example, ODFW describe both minimum viability and "broad sense" recovery options with different associated salmon abundance levels (ODFW, 2011).

Spring Chinook are the state fish of Oregon and salmon are a conspicuous symbol in the US Pacific Northwest in general. Given their cultural importance, a significant TEV for increasing salmon abundance in the Willamette seems likely[1]. As of this writing, no empirical studies are known estimating the TEV of changes in abundance specifically for Willamette salmon. However, Olsen et al. (1990) and Layton et al. (1999) implicitly include the Willamette watershed as a portion of the change they consider for the entire Columbia River watershed. In addition, Wallmo and Lew (2012) query a national sample to estimate the value of a change in status for Willamette Spring Chinook from threatened to recovered (not explicitly tied to a change in abundance) as one of eight listed marine species included their study.

To constrain the benefit transfer, a specific salmon population change for the Willamette must be cited. At Willamette Falls on the Willamette River, the ODFW has counted returning Spring Chinook since 1946: the most recent ten-year average (up through 2014) is 35,115 fish. Not all spawning habitat is upstream of the Falls, and there is significant attrition before they reach that point, e.g., anglers in the popular Lower Willamette fishery. ODFW also reports an estimated entire Willamette Run, which was 64% higher than counts at the Falls in 2014. Thus, for the purposes of this paper, the status quo Willamette Spring Chinook run is estimated to be 164% of the 10-year average, rounded to the nearest 100 fish, or 57,600 fish. The majority of returns are hatchery fish. The specific commodity valued in this paper is a doubling in the average annual Willamette Spring Chinook run, from the estimated status quo average of 57,600 fish per year, to 115,200 fish per year. Note that this is not the only possible fish-related commodity. Also note that the change does not specify whether increases pertain to hatchery or wild fish. This decision was forced mainly by most available studies neglecting to specify for survey respondents whether hatchery or wild fish were impacted. It seems likely that preserving wild fish in particular would matter mainly for the non-use component of TEV, but perhaps also for angling use value.

The market extent—which households will be considered in the analysis—must also be specified. This judgment determines how values will be aggregated, and the outcome can be especially sensitive when non-use values are involved. Defining market extent also aids selection of study cases (and their market extents) most relevant for the policy case. Salmon recovery within a relatively small watershed has been found to be valuable to households across the nation (Loomis, 1996). Thus Pate and Loomis (1997) caution against artificially limiting market extent specifically for changes in salmon, due to the underestimates of public welfare that could result. However this paper takes a relatively conservative stance, defining the market extent as just the households within the Willamette watershed. While values for increases in Willamette salmon abundance may well extend more broadly, the author believes it preferable to err on the conservative side when relying on secondary estimates. Furthermore, substitute salmon resources do not appear to have been well characterized within available studies. This leads to somewhat more caution than usual when applying such values

[1]Describing the policy case helps define the appropriate scope of the benefit analysis, such as whether relevant values are exclusively ecological in character or if human health concerns are an issue; unequal distributive impacts of an environmental change may also be important to consider (US Environmental Protection Agency, 2010). This application focuses on ecological values. Adverse health impacts seem unlikely since contamination advisories in the region focus on resident (non-migratory) fish. Distributional analysis would require additional social and biological data, including a model of fish distribution at baseline and improved levels.

at market extents beyond which households could be reasonably expected to have familiarity with the range of substitute salmon resources available in the Western US. However, the value estimates derived in this paper will be on a per household basis to facilitate comparison with other valuation work, and will not actually be presented in aggregate form.

To provide an example of how monetizing the societal benefits of salmon could assist regional decision-making, consider the long-running debate in Oregon over the appropriate distance buffer for logging on private timberlands near streams. The issue was recently reignited with a study finding that the current buffer set by the Oregon Board of Forestry does not do enough for small and middle-sized streams to maintain shading and temperature requirements set by the Oregon Environmental Quality Commission (Barnard, 2015a,b). Costs to private timberland owners of larger buffers are relatively easy to estimate, and are readily available. Missing from the debate is an estimate of the TEV of increased salmon associated with cooler streamwater (although this value would certainly not be the only consideration).

The above forest practices example raises the interconnectedness of estimating TEV with biophysical predictions of salmon populations. Such predictions are extraordinarily challenging—models quantifying salmon response to changing conditions and restoration strategies contain significant uncertainty (e.g., ODFW, 2011). Interconnected freshwater factors must be further combined with ocean conditions, one of which is the Pacific Decadal Oscillation (Mantua et al., 1997). Limiting factors for salmonids in the freshwater environment that have been identified by regional research, in addition to elevated water temperature (see also McCullough et al., 2009), are lack of large wood in the channel (Oregon Department of Environmental Quality, 2009), and increased silt (Honea et al., 2009).

Data

In assembling case study data, criteria were that the study must supply at least one estimate in WTP format to preserve, increase, or avoid the loss of a given number of Western US salmon; sample the general population in one or more regions of the US; and be intended to capture TEV. A broad internet-based search was conducted including the "Environmental Valuation Resource Inventory" of Environment Canada, and inquiries were made with other valuation researchers. Ultimately, only six relatively well-known studies were located: Jones and Stokes Associates (1990); Olsen et al. (1990); Loomis (1996); Layton et al. (1999); Bell et al. (2003); and Mansfield et al. (2012). A summary of these studies is given in the Appendix.

Qualitative and quantitative metadata from the six studies and corresponding 29 observations are summarized in **Table 1**. Publication dates range from 1990 to 2012. Since the studies occur in different years, have varying payment plans, and reference different salmon changes, TEV results are not directly comparable. Raw values were adjusted to 2015 dollars using an inflation calculator based on the national Consumer Price Index (CPI) (US Bureau of Labor Statistics, 2015). To account for different payment plans ranging from 10 years to perpetuity, net present value (NPV) was calculated with a 7% yearly discount rate, a common base-case rate (US Office of Management and Budget, 2003). However, the reader should be aware that the discount rate is a sensitive and controversial variable (Weitzman, 1998). Furthermore, it is not clear how survey respondents themselves discount a stream of payments into the future when responding to a WTP questionnaire. Since each observation values a different change in fish population from a different baseline scarcity, **Table 1** provides additional context for value interpretation. **Figure 1** plots the NPV of salmon abundance against the scope of the salmon population increase. Both axes are on a logarithmic scale to make it easier to differentiate clustered observations. Data labels in **Figure 1** match the observation numbers in **Table 1**. Note that observation 28 is dropped from the figure (and from later analyses) since it is negative, an anomaly in the dataset.

Although all the studies in **Table 1** meet selection criteria, they have numerous differences, such as examining different salmon populations (although they occasionally overlap), and surveying different market extents. The early studies by Jones and Stokes Associates (1990) and Olsen et al. (1990) are fully or partially conducted by telephone, in contrast to later studies which rely almost exclusively on mail surveys, with the most recent study combining mail and internet modes. The elicitation format is contingent valuation for all but Layton et al. (1999) and Mansfield et al. (2012). Explanation within the survey regarding how improvements would occur varies, with Jones and Stokes Associates (1990) and Loomis (1996) providing the most detail. In the author's opinion, all of the surveys could have included more information on substitute migratory fish resources in the Western US. None of the studies included information on salmon resources outside of the watersheds that were the topic of the survey. That said, the Olsen et al. and Layton et al. studies considered large systems, e.g., the entire Columbia River watershed, which implicitly captures regional substitutes.

What then can be gleaned from **Table 1** regarding the TEV of changes in Western US salmon abundance for the policy case and other applications? The studies consistently indicate that households in the Pacific Northwest and beyond have a high WTP for increased salmon, yet they do not cover all of the areas in the Western US that currently provide salmon habitat. If the parameters of a given study in **Table 1** happen to match a context of interest, perhaps no further analysis is required. More often, there is interest in estimating a value for an "out of sample" context. Furthermore, insights drawn from a group of observations are arguably stronger even if the parameters of a single study are well matched to a given policy scenario. Benefit transfer tools developed for these situations will be reviewed in the next several sections.

Before proceeding, it should be noted that studies in **Table 1** do not include all of the available insight on salmon values in the Western US. Notably, some survey studies have elicited a WTP for recovery of salmon listed under the Endangered Species Act, rather than specifying population changes. This focuses specifically on wild fish, and the values would seem to be more associated with non-users than an equivalent study citing abundance changes. For examples see Bell et al. (2003),

TABLE 1 | Studies estimating the total economic value of migratory pacific salmon abundance changes in the US.

Observation	Reference	Salmon baseline	Salmon increase	Change context	Watershed area (sqmi)	Sample frame	Payment plan	Response rate %	Willingness to pay per household, net present value: $r = 0.07$ (2015$)	Willingness to pay per household, as if a yearly payment: $r = 0.07$ (2015$)
1	Jones and Stokes Associates, 1990	100	14,900	San Joaquin River, CA	41357	227 San Joaquin Valley households	Perpetuity	60	$5454.00	$381.78
2	Jones and Stokes Associates, 1990	100	14,900	San Joaquin River, CA	41,357	576 CA households outside of SJV	Perpetuity	60	$4887.00	$342.09
3	Jones and Stokes Associates, 1990	100	14,900	San Joaquin River, CA	41,357	201 OR, WA, NV households	Perpetuity	60	$2781.00	$194.67
4	Olsen et al., 1990	2,500,000	2,500,000	Columbia River	240,354	2097 WA, OR, ID, W.MT households	Perpetuity	72	$1359.51	$95.17
5	Loomis, 1996	5000	30,000	Elwha River, WA	321	284 Clallam County, WA households	10 yr	77	$700.57	$49.04
6	Loomis, 1996	5000	30,000	Elwha River, WA	321	467 "other" WA households	10 yr	68	$866.81	$60.68
7	Loomis, 1996	5000	30,000	Elwha River, WA	321	423 U.S. households	10 yr	68	$807.44	$56.52
8	Layton et al., 1999	500,000	25,000	Columbia River	240,354	810 WA households	20 yr	68	$1527.54	$106.93
9	Layton et al., 1999	2,000,000	100,000	Columbia River	240,354	801 WA households	20 yr	68	$797.62	$55.83
10	Layton et al., 1999	500,000	375,000	Columbia River	240,354	810 WA households	20 yr	68	$4097.79	$286.85
11	Layton et al., 1999	2,000,000	1,500,000	Columbia River	240,354	801 WA households	20 yr	68	$2139.70	$149.78
12	Layton et al., 1999	500,000	750,000	Columbia River	240,354	810 WA households	20 yr	68	$4755.67	$332.90
13	Layton et al., 1999	2,000,000	3,000,000	Columbia River	240,354	801 WA households	20 yr	68	$2483.21	$173.83
14	Layton et al., 1999	2,500,000	125,000	Puget Sound, WA	20,177	810 WA households	20 yr	68	$2305.78	$161.40
15	Layton et al., 1999	5,000,000	250,000	Puget Sound, WA	20,177	801 WA households	20 yr	68	$1678.20	$117.47
16	Layton et al., 1999	2,500,000	1,250,000	Puget Sound, WA	20,177	810 WA households	20 yr	68	$6185.50	$432.98
17	Layton et al., 1999	5,000,000	2,500,000	Puget Sound, WA	20,177	801 WA households	20 yr	68	$4501.95	$315.14
18	Layton et al., 1999	2,500,000	2,500,000	Puget Sound, WA	20,177	810 WA households	20 yr	68	$6597.65	$461.84
19	Layton et al., 1999	5,000,000	5,000,000	Puget Sound, WA	20,177	801 WA households	20 yr	68	$4801.92	$336.13
20	Bell et al., 2003	64,000	64,000	Willapa Bay, WA	1,268	299† high income WA households, in 30 mi radius	5 yr	62	$726.79	$50.88
21	Bell et al., 2003	64,000	192,000	Willapa Bay, WA	1268	299† high income WA households, in 30 mi radius	5 yr	62	$706.68	$49.47
22	Bell et al., 2003	64,000	64,000	Willapa Bay, WA	1268	144† low income WA households, in 30 mi radius	5 yr	62	$466.29	$32.64
23	Bell et al., 2003	64,000	192,000	Willapa Bay, WA	1268	144† low income WA households, in 30 mi radius	5 yr	62	$453.40	$31.74
24	Bell et al., 2003	128,900	128,900	Grays Harbor, WA	2704	267† high income WA households, in 30 mi radius	5 yr	49	$713.25	$49.93
25	Bell et al., 2003	128,900	386,700	Grays Harbor, WA	2704	267† high income WA households, in 30 mi radius	5 yr	49	$695.65	$48.70

(Continued)

TABLE 1 | Continued

Observation	Reference	Salmon baseline	Salmon increase	Change context	Watershed area (sqmi)	Sample frame	Payment plan	Response rate %	Willingness to pay per household, net present value: r = 0.07 (2015$)	Willingness to pay per household, as if a yearly payment: r = 0.07 (2015$)
26	Bell et al., 2003	128,900	128,900	Grays Harbor, WA	2704	150† low income WA households, in 30 mi radius	5 yr	49	$471.00	$32.97
27	Bell et al., 2003	128,900	386,700	Grays Harbor, WA	2704	150† low income WA households, in 30 mi radius	5 yr	49	$459.43	$32.16
28	Mansfield et al., 2012	130,000	100,000	Klamath River, OR and CA	13,426	1164 households outside of OR and CA	20 yr	30	$ −210.67	$ −14.57
29	Mansfield et al., 2012	130,000	150,000	Klamath River, OR and CA	13,426	1164 households outside of OR and CA	20 yr	30	$122.44	$8.57

† *Approximate sample size based on high-income and low-income sample proportions given in Bell et al. (2003).*

Lew and Wallmo (2011), Wallmo and Lew (2011, 2012), and Mansfield et al. (2012). Other survey studies have treated habitat improvements for salmon, without specifying the impact on salmon either in terms of listing status or in terms of abundance (e.g., Garber-Yonts et al., 2004). At least one study has elicited values for changes in salmon abundance as a commodity lumped with other environmental attributes, Douglas and Taylor (1999). Still more studies have focused on just recreational or commercial benefits of salmon.

Point Estimate Transfer

The simplest form of benefit transfer is transferring a point estimate. The value for the most similar context should serve as a reference, but this is challenging to identify since there are multiple dimensions of applicability. There are no obvious reasons to suspect quality issues with estimates in **Table 1** except observations 28 and 29 from Mansfield et al. (2012), which as described in the Appendix have broad confidence intervals; the response rate for this study is also the lowest in **Table 1**. Survey response rates are quite high for the other five studies, from 49% (Bell et al., 2003: Grays Harbor) to 77% (Loomis, 1996, Clallam Co.). No two studies were conducted by exactly the same protocol, and there remains active debate about best practices in valuation. One of the debates concerns contingent valuation vs. alternative stated preference formats. Layton et al. (1999) and Mansfield et al. (2012) are the only studies that do not use contingent valuation. On one hand, alternatives to contingent valuation are gaining in popularity and choice experiments seem to have become the new standard. On the other hand, value estimates can be higher (Stevens et al., 2000; Bateman et al., 2006), and consensus protocols are still emerging (e.g., Hanley et al., 2001; Louviere et al., 2010; Boyd and Krupnick, 2013). However, to be sure, a variety of application styles is possible within either contingent valuation or choice experiments that may eclipse the effect of methodology alone.

In addition to study methodology, other obvious differences between studies in **Table 1** are the size of fish change, regional scarcity, and market extent. It is tempting to utilize the survey results of Olsen et al. (1990) since some of the respondents would have been Willamette valley households, but the scale of change is the entire Columbia, an area about 20 times as large as the Willamette, and features a salmon abundance change measured in millions rather than tens of thousands of fish. Jones and Stokes Associates (1990) offer an estimate for San Joaquin Valley households, a watershed similar in size to the Willamette, but the baseline fish population is extremely low. The change would essentially create a Chinook fishery rather than supplement an existing one. The Bell et al. (2003) estimate for a 100% improvement in Coho salmon in Willapa Bay, WA is the most similar in terms of baseline and final fish population, even though the extent of market surveyed and the watershed itself are relatively small. Since no prior studies match the Willamette context in all dimensions, a judgment was made to match closest available baseline and final fish population. Comparing US Census household income for the Willapa Bay and Willamette valley regions indicates that of the two possible Bell et al. models, the high income (observation 20) more closely

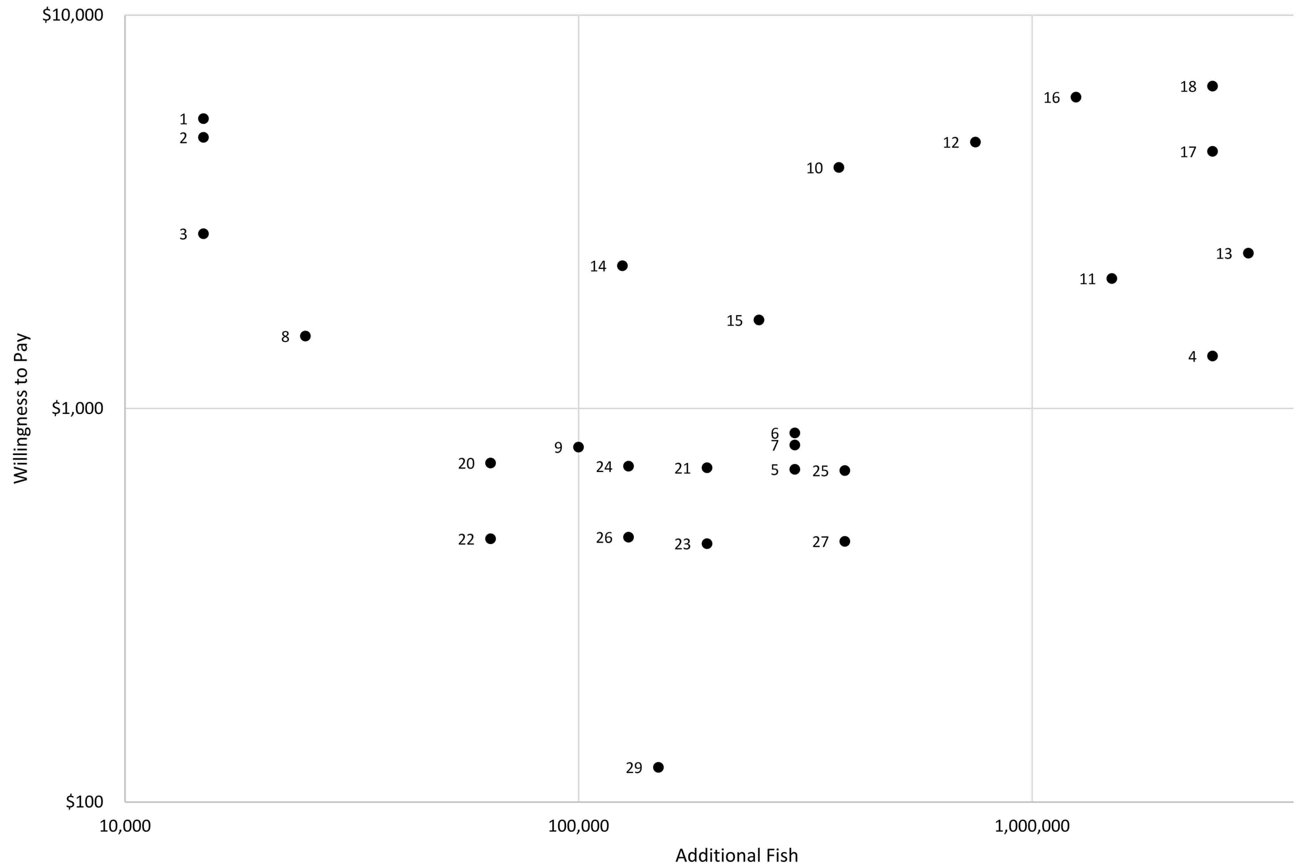

FIGURE 1 | Total economic value of Western US salmon increases vs. size of salmon increase (Net Present Value per household 2015\$, $r = 7\%$).

matches Willamette valley demographics. The corresponding estimate is \$726.97 NPV, or \$50.89 per household as if a yearly payment (2015\$).

Functional Transfer

The next step in complexity is functional transfer, which uses a model offered by the study case, and adjusts context variables to estimate the policy case value (Loomis, 1992). By elimination, Layton et al. (1999) is best suited for functional transfer; other studies either did not include a model or used highly specialized information available only through their survey instrument. The Layton et al. models are scaled by the context variable of percentage change in fish population. There are different models for different fish category, and either high or low baseline fish population. The Layton et al. functions were developed for a much larger area and much larger absolute numbers of fish, however there are a few to choose from. Of the two migratory fish models, Columbia River and Puget Sound, the former was chosen since the Willamette is at least a tributary of this system. The Columbia models also have lower status quo fish populations than the Puget Sound models, although even the low baseline option is still higher than the Willamette Spring Chinook status quo by an order of magnitude. An alternative possibility is choosing the high baseline model in an attempt to harness diminishing returns to offset the larger scope of the Columbia system, but here the low baseline option is employed as a more logical usage of the functional transfer method. The Layton et al. (1999) formula for monthly WTP when the increase in fish is greater than 5% is:

$$\mathrm{WTP} = \beta_{\mathrm{fish}}(0 - \ln(\mathrm{fish}\ \%\ \mathrm{change}))/\beta_{\mathrm{cost}}$$

Solving the function for a 100% change using their regression parameters, converting the monthly value to a yearly value, and then converting this to a NPV using a 7% discount rate yields \$4,370.83 per household; if expressed as a yearly payment the value is \$305.96. Both values are converted to 2015\$ using the CPI.

Existing Meta-regressions

With both point transfer and functional transfer, the analyst must choose a single study. In contrast, meta-analysis uses observations from multiple studies to gain insight, and may use statistical techniques such as meta-regression to isolate sources of influence on a dependent variable (Stanley, 2001). Three published meta-regressions were found that could be applied to the policy case of doubling Willamette Spring Chinook. Johnston et al. (2005) estimated a meta-regression of TEV for changes in US water bodies that provide fishing or other recreational improvements, using 81 observations. Richardson and Loomis (2009) supply a meta-regression of TEV for US Threatened and Endangered species using 67 observations, as an update to Loomis and White (1996). Last, Loomis and Richardson (2007) provide a meta-analysis specifically for Western US Salmon, using 20 observations from five of the six studies earlier described.

While meta-regressions supply predictive relationships, applying them is not necessarily straightforward. Context variables can reflect both household preferences and study methodology. For methodological variables, it is usually recommended that the analyst employ sample means, yet one may also wish to select particular "favorite" methodologies for a given transfer, e.g., a method known to be relatively conservative. Preference variables may be straightforward, but can also be specialized and difficult to generate for the policy case. Meta-regressions may also present multiple functional forms, leaving the analyst uncertain regarding which to utilize.

To illustrate how these factors can affect the benefit transfer for the selected policy case, limited sensitivity analyses are presented based on the three existing meta-regressions. **Table 2** shows choices used in applying Johnston et al. (2005). Of their three models, the weighted semi-log specification is used since it had the best fit as published [the functional form is not reproduced here, but parameters and estimates can be viewed in (Johnston et al., 2005): Table 3]. Only eight of the 34 regressors are adjusted here, chosen for relevance to the application and for their influence. Variables not shown were constant throughout[2]. The first two adjusted variables are based on the well-known water quality index, mathematically determined from numeric values of fecal coliform, dissolved oxygen, biological oxygen demand, turbidity, and pH (Vaughan, 1986). The index has been used to define rungs on a water quality "ladder": boatable; fishable; swimmable; and drinkable, that has had a legacy influence on water quality valuation studies (Carson and Mitchell, 1993). A primary limiting water quality factor for salmon in the Willamette is thought to be elevated stream temperature, with migration barriers posed by dams being another major factor. Yet there is no defined "rung" for increased fish abundance *per se*. Thus, different changes in the water index are shown as part of the sensitivity analysis.

In applying the Johnston et al. (2005) meta-regression, the first column of **Table 2** shows a potential study, i.e., one that might be designed if pursuing an original survey for the case study. For the water quality index, a baseline corresponding to "game fishing" is entered with a change halfway to the next rung of "drinking without treatment" (Vaughan, 1986). A choice experiment would allow direct inclusion of potential substitutes, an attractive feature in valuation, thus discrete choice methodology is selected. Mail survey mode is selected due to its low cost, and high response rate is not selected since achieving rates above the 75% threshold is uncommon. The second and third columns adjust eight selected variables within reasonable bounds to explore lower and upper bounds, respectively. The fourth column utilizes Johnston et al. (2005) sample means. For year, the most recent study year is used as typically recommended for benefit transfer. The impact of study year is shown by providing an additional upper bound value using the earliest meta-data date instead. The function returns estimates in 2002$, which were adjusted to 2015$ using the CPI. Using sample means, the estimate is $46.51 per household NPV, or $3.26 per household as if a yearly payment in perpetuity.

The meta-regression by Richardson and Loomis (2009) for endangered species values is applied in **Table 3**, again comparing four columns. The authors recommend their reduced double log model 3 for benefit transfer, which includes 10 regressors (the functional form is not reproduced here but a calculation example showing parameters and estimates is provided by Richardson and Loomis, 2009: Section 3.5). The key preference variable is

[2]Methodological variables not in **Table 2** were set to sample means. Sample means were not used for preference variables; settings constant throughout were that the value represents a lump-sum payment, the geographic context is the Pacific Mountain USDA region, the population includes the non-user community, water quality improvements only benefit fin fish, the improvement is more than a 50% increase, and mean household income is $71,690 (weighted average based on Census data for the 10 counties most closely corresponding with the watershed boundary, adjusted to 2015$ with the Consumer Price Index).

TABLE 2 | Meta-regression results applying Johnston et al. (2005).

Variable	Correlation with willingness to pay	Potential study	Lower bound	Upper bound	Sample means
Baseline water quality index, 1−10	−	5	7	4	4.6
Water quality index change	+	2.25	2	5	2.42
High response rate dummy (over 75%)	−	0	1	0	0.31
Choice experiment dummy	+	1	0	1	0.35
Mail survey mode	+	1	0	0	0.56
Telephone survey mode	Reference Mode	0	1	0	N/A
In-person interview survey mode	+	0	0	1	0.19
Study year	−	2001†	2001†	2001† (1973)	2001†
Willingness to pay per household, net present value (2015$)		$76.73	$8.64	$247.74 ($7,152.34)	$46.51
Willingness to pay per household, as if a yearly payment (2015$)		$5.37	$0.60	$17.34 ($500.66)	$3.26

† *Metadata span 1973–2001.*

TABLE 3 | Meta-regression results applying Richardson and Loomis (2009).

Variable	Correlation with willingness to pay	Potential study	Lower bound	Upper bound	Sample means
Response rate	−	49%	75%	25%	49%
Choice experiment	+	1	0	1	0.075
Mail survey mode	−	1	1	0	0.851
StudyYear	+	2007†	2007† (1983)	2007†	2007†
Willingness to pay per household‡ (2015$)		$2,608.95	$127.07 ($8.50)	$9,667.31	$185.84

† *Metadata span 1983–2007.*
‡ *The payment schedule is uncertain, underlying observations did not appear to have been scaled to net present value.*

the percentage change in population of the endangered species; this is set to 100%. There is no adjustment for the baseline level of fish, although one would expect that a lower baseline would mean a higher marginal value for each additional fish as compared with a location where fish are already plentiful. As with using Layton et al. (1999) for functional transfer above, percentage increase must be used rather than absolute increase in fish numbers. Selections in the function were made to indicate that the endangered species was a fish. The "visitor" variable was not selected to indicate that value beyond recreation was desired. For a potential study scenario in column 1, selections were made to parallel those in column 1 of **Table 2**. Study year again has a large impact, but in the opposite direction. Thus, an additional lower bound value is shown using the earliest meta-data date. The function returns values in 2006$, which were adjusted to 2015$ using the CPI. The estimate based on sample means is $185.84 per household. The payment schedule is unspecified since the observations had various payment plans that did not appear to be adjusted to a NPV.

The third meta-regression, utilizing Loomis and Richardson (2007), only requires input on percentage change in fish run, with the meta-regression utilizing percentage change and percentage change squared (with no constant term). This function has a strong advantage in simplicity of application, particularly given the online calculator provided by the authors. However again there is no adjustment possible for baseline fish populations, and percentage change must be used instead of absolute fish numbers. The calculator provides estimates in 2006$. Using the CPI to adjust to 2015$, the resulting estimate for a 100% increase in salmon population is $89.23 per household. The payment schedule is unspecified since the observations had various payment plans that did not appear to be adjusted to a NPV.

New Meta-regression

Taken together, the three meta-regressions represent five of the six studies in **Table 1**, but not all of the observations that can be extracted from those five studies. For example, Loomis and Richardson (2007) represent the various Layton et al. (1999) models at only the 50% fish increase levels. In addition, there are two new observations from Mansfield et al. (2012). As noted earlier, observation 28 is the only negative value in **Table 1** and thus appears to be an outlier. This leaves 28 observations with which to explore a new meta-regression. The dependent variable was defined as NPV of household TEV elicited as WTP in 2015$.

As seen earlier, meta-regressions typically include methodological regressors to control for study differences. Here, this has not been explored due to there being few studies, which often vary in more than one methodological respect from each other, leading to confounding effects. This meta-regression is limited to preference variables. Different hypotheses were considered regarding conversion of before and after fish abundance into one or more resource quality variables. As discussed above it was desired to incorporate not only the scope of change, but also the baseline level of fish. However, change in fish and baseline fish have a correlation coefficient of 0.7, signaling multicollinearity issues in a regression model using both variables. The problem is that when researchers have elicited WTP for large increases in salmon, this has tended to occur when the baseline levels of salmon were also large. Thus, there is limited independent variation in added fish and baseline fish. Transforming the variables by centering data (subtracting the mean from each observation) was ineffectual in reducing this high correlation. A different transformation is possible by dividing baseline fish by watershed area, under the rationale that when gauging scarcity a household might consider overall watershed size. Although numeric watershed areas were not specified in the study cases, a map was typically provided, except when telephone sampling. Transforming the scarcity variable in this manner substantially reduced correlation (correlation coefficient reduced to 0.4). Note that both variables cannot be transformed or the high correlation reappears. Watershed areas were found using publicly available national hydrography data (Horizon Systems Corporation, 2015; US Geological Survey, 2015) and are shown in **Table 1**. Constructing the independent variables in this way hypothesizes that survey respondents viewed scarcity in context of the watershed, but instead reacted directly to the added number of fish. This is questionable, thus the transformation is only employed in one of the regression models presented below.

All models use a log form for both the baseline fish and added fish variables. For baseline fish, this means that as baseline level decreases, the (expected) influence on increasing WTP would accelerate per unit decrease. For added fish, a log form is one way of instituting the common assumption of diminishing returns: as added fish goes up, added WTP also goes up but at a decreasing rate. All models also use a cluster adjustment for standard errors, to control for non-independence of observations from the same study. Results from four models are shown in **Table 4**. Model 1 has the expected positive sign for added fish

TABLE 4 | New meta-regression of the total economic value of migratory pacific salmon abundance changes in the US.

	Model 1	Model 2	Model 3	Model 4
Observations	Drop 28	Drop 28	Drop 16, 18, 28	Drop 8 to 19, 28
n	28	28	25	16
r-squared	0.14	0.40	0.12	0.78
Intercept (std error)	−3100.27 (3626.968)	−7125.27** (2405.383)	−636.724 (3275.488)	2417.42 (3730.635)
ln of added fish (std error)	699.152** (206.191)	844.183** (207.139)	473.052* (181.7151)	493.498 (432.0829)
ln of baseline fish (std error)	−278.066 (263.4649)	N/A	−273.68 (227.6074)	−672.078** (137.0615)
ln of baseline fish/unit area (std error)	N/A	−439.32** (128.2208)	N/A	N/A
Policy Case: willingness to pay per household, net present value (2015$)	$1515.37	$1427.96	$1548.65	$459.96
Policy Case: willingness to pay per household as if a yearly payment (2015$)	$106.08	$99.96	$108.41	$32.20

*Significant at the 0.05 level.
**Significant at the 0.01 level.

and expected negative sign for baseline fish. However, Model 1 has several weaknesses. Only the added fish variable is significant, the model has a low r-squared, the two regressors are highly correlated, and examination of standardized residuals shows two outliers (exceeding positive or negative two). Model 2 utilizes the transformed baseline fish variable as described above to reduce correlation between variables. This improves *r*-squared, both variables retain expected sign, and both variables are now significant. Model 3 repeats Model 1, dropping the two outliers (observations 16 and 18). This only minimally affects the model. However, the model is sensitive to dropping certain observations and/or studies, since there are only 28 observations total from six studies. To illustrate this, the study contributing the most observations (Layton et al., 1999) is dropped for Model 4. Now only the baseline fish variable is significant, and interestingly the r-squared improves dramatically. Model 4 also shows a dramatic reduction in predicted TEV for the policy site.

Further development of the meta-regression is certainly possible. Despite potentially confounding effects with so few studies, methodological variables could be attempted. A treatment for heteroskedasticity would be desirable, however variance was not uniformly available from the study cases. Observations could be weighted by sample-size, a second-best solution for benefit-transfer (Nelson and Kennedy, 2009). However, given the known efficiency of some valuation methodologies relative to others, and development of valuation techniques across the span of years in the sample, weighting on sample size would seem a dubious approach. There are also few observations overall, limiting the ability to include additional context variables due to the danger of overfitting. Overall the investigation of a new meta-regression is less than satisfying. Parameters do show the theoretically expected sign, but parameter significance and overall model performance are low or unstable.

Structural Benefit Transfer

Meta-regression reanalyzes valuation estimates along with other meta-data from original studies in search of statistical predictors for WTP. However the resulting equation cannot be viewed as a utility function when there are variables (i.e., methodological) not theoretically linked to preferences. A separate form of meta-analysis known as structural benefit transfer (aka preference calibration) uses a different approach. A utility function for a representative agent is defined, and outcomes of valuation studies are used to calibrate preference coefficients. From that point, application is similar to functional benefit transfer. Advantages of this technique are its explicit connections to economic theory (for example WTP can be bounded by household income), consistency, and its ability to integrate value estimates from different techniques and welfare measures, such as recreational value from travel-cost techniques, and TEV from a survey. A weakness is subjective identification of the underlying utility function, and as typically employed, use of relatively few observations to calibrate parameters. Studies used for calibration should reasonably apply to representative households for the policy case. In contrast to the relatively large number of environmental valuation meta-analysis studies, there are relatively few for structural benefit transfer.

For references developing structural benefit transfer, see Smith and Pattanayak (2002), Smith et al. (2002), Van Houtven et al. (2011), and Van Houtven and Poulos (2009). Only Van Houtven et al. include treatment of non-use values, a crucial concern for the policy case. Here we include two of the five functional forms they considered, a modified constant elasticity of substitution (CES) functional form (Equation group 1), and a linear functional form (Equation group 2), both of which performed reasonably well for their case study. It should be plainly stated that multiple functional forms are possible other than the two tested here. Both functional forms predict indirect utility "V," with changes in the resource quality from "Q." Also included are the price of visiting the resource "P" (i.e., travel cost), and income "Y." Subscripts "0" and "1" correspond to initial and final resource quality levels. Each equation has five parameters to calibrate, alpha α, beta β, gamma γ, delta δ, phi φ, and psi ψ. Income less WTP for improved quality (or WTP to avoid degraded quality) that equilibrates initial and final indirect utility corresponds to a Hicksian welfare change expressed in dollars. Via Roy's Identity the functions can be re-expressed in terms of number of trips demanded per household per year (X) which can in turn be related to a Marshallian welfare change (ΔMCS) in dollars typical in travel cost studies. Based on these algebraic manipulations the following formulas are taken from Van Houtven et al. (2011):

(1) Modified Constant Elasticity of Substitution (CES) Functional Form:

$$\begin{aligned} V &= \varphi Q^{\psi} + ((P - Q^{\gamma})^{-\alpha} Y^{\delta})^{\beta} \\ WTP &= Y - (((\varphi Q_0^{\psi}) - (\varphi Q_1^{\psi}) \\ &\quad + ((P - Q_0^{\gamma})^{-\alpha} Y^{\delta})^{\beta})^{1/\beta\delta}) \,/\, (P - Q_1^{\gamma})^{-\alpha/\delta} \\ X &= (\alpha/\delta) * (Y \,/\, (P - Q^{\gamma}) \\ \Delta MCS &= (\alpha/\delta) * Y * (\ln(P - Q_0^{\gamma}) - \ln(P - Q_1^{\gamma})) \end{aligned}$$

(2) Linear Functional Form:

$$\begin{aligned} V &= \varphi Q^{\psi} + (Y + 1/\delta(\alpha - \beta P + \gamma Q - \beta/\delta))^{*} \\ &\quad \exp(\delta/\beta(\gamma Q - \beta P)) \\ WTP &= 1/\delta((X_1 - \beta/\delta) \\ &\quad - (X_0 - \beta/\delta)^{*}\exp((\delta_{\gamma}/\beta) * (Q_0 - Q_1)) \\ &\quad + \varphi(Q_1^{\psi} - Q_0^{\psi})^{*} \exp((\delta/\beta) *(\beta P - \gamma Q_1)) \\ X &= \alpha - \beta P + \delta Y + \gamma Q \\ \Delta MCS &= (X_1^2 - X_0^2)/2\beta \end{aligned}$$

In both cases the first term on the right-hand-side of "V" is a simple means of expressing non-use value, while the second more complicated term represents use value. Only preference variables are included: income (Y); trip price (P); the quality variable (Q); and the six parameters (greater than or equal to 0) to be calibrated. Changes in utility are effected by changing from the initial to a final quality state, Q_0 to Q_1. Increases in quality increase the non-use component directly, and reduce the effective price of the trip within the use value term. For example, all else equal, people are less likely to incur a high travel cost P to visit a site with low Q. Hicksian and Marshallian welfare estimates, an average income estimate, an average trip price, and recreational demand statistics can all be used to calibrate parameters.

In addition to testing two functional forms, we also test two ways of calculating quality, resulting in four models total in **Table 5**. The first method of calculating quality is based on amount of fish per watershed area. To rescale the variable to have an upper limit of 10, observed fish density was multiplied by 10 and divided by the in-sample maximum density post-increase (observations 5, 6, and 7 all have this maximum density). The second method of calculating quality is based on the number of fish without regard to watershed area. In this latter case, scaling from 0 to 10 was done based on the upper limit being the maximum in-sample total number of fish post-increase (observation 19).

For recreational statistics and recreational angling use-value, we utilize angler survey results for Spring Chinook in the Lower Willamette River. The Research Group (1989) conducted the survey and Lin et al. (1996) provide additional analysis by leveraging data on site characteristics during 1988 from ODFW. Four observations, A, B, C, and D of angler trips per household per season are possible from the data, corresponding to different Spring Chinook run sizes (these receive letter labels in **Table 5** to avoid confusing these observation with the numbered observations in **Table 1**). First, overall angler-day effort for the fishery was estimated at 222,457 days in 1988 while the average run size for 1986–1993 was 86,000 fish (as cited in Lin et al., 1996). The interception-mode survey conducted in the 1988 season asked anglers about their current average trip frequency (11.6 trips/season), their expected trip frequency with a 10% increase in the run (2.3 additional trips/season), and with a 20% increase in the run (3.0 additional trips/season). A fourth estimate of how fishing trips change with fishing quality is possible using an estimated average angler-day effort in 1974–1979 being 147,000 and average run size during a similar timeframe 1976–1985 being 63,500 fish (as cited in Lin et al., 1996). All four of these observations are scaled to represent trip frequency for a representative household by dividing by the approximate number of households in the Willamette valley based on Census data. Note that observations A, B, C, and D assume the Willamette valley is a feasible market extent for the Willamette Spring Chinook fishery. Observations B and C based directly on the angler survey assume no influx of new anglers with run size: the stated change number of trips by those who are already anglers is used to estimate total change in angler effort. All four are "X" observations, i.e., average trips per household per season.

Loss in Marshallian consumer's surplus based on change in Willamette Spring Chinook run size is also available from Lin et al. (1996). Based on a random utility model they calculate welfare loss at $0.4657 per trip with a reduction in run size of 5000 fish, or $-0.92 in 2015$ using the CPI. This calculation relies on the more conservative formulation of the travel cost variable presented by Lin et al. with opportunity cost of time valued at 1/3 of the average wage rate rather than at 100%. Adjusting the welfare loss by the average number of trips per household from the first observation results in observation E; loss in use value for the given quality decrease.

Remaining observations in **Table 5** used for the four calibrations are Hicksian WTP estimates from **Table 1**. In an effort to have the calibrated utility function more reasonably match relatively limited market extents, given that the policy case is limited to the Willamette valley, observations were dropped in which the WTP was associated with out-of-state respondents (observations 3, 7, 28, and 29).

Estimates for average household income and average travel cost are also needed to calibrate the utility functions. For average household income, Jones and Stokes Associates (1990) was the only study that provided sample information. For other observations average income was calculated from Census estimates corresponding to the sampled geography. When more than one geography was sampled (e.g., Olsen et al. included more than one state), a weighted average based on population was calculated. For the Bell et al. (2003) models distinguished by above and below median income respondents, an estimate of average household income given the condition of being above or below the median was interpolated based on Census county level income quintiles (Census table b19081). In all cases average income estimates were adjusted to 2015$ using the CPI.

TABLE 5 | Structural benefit transfer of the total economic value of migratory pacific salmon abundance changes in the US.

Observation label†	Observation type	Observation value	*Y*	*P*	Models 1 and 3 CES functional form		Models 2 and 4 linear functional form	
					Q_o	Q_1	Q_o	Q_1
A	X	0.287	71,690	52.68	0.067	N/A	0.086	N/A
B	X	0.190	71,690	52.68	0.050	N/A	0.064	N/A
C	X	0.355	71,690	52.68	0.074	N/A	0.095	N/A
D	X	0.373	71,690	52.68	0.081	N/A	0.103	N/A
E	ΔMCS	−0.26	71,690	52.68	0.067	0.063	0.086	0.081
1	WTP	381.78	94,617	248.47	0.000	0.003	0.0001	0.015
2	WTP	342.09	94,617	457.03	0.000	0.003	0.0001	0.015
4	WTP	194.67	68,051	259.62	0.095	0.191	2.5000	5.000
5	WTP	95.17	85,494	13.34	1.429	10.00	0.0500	0.350
6	WTP	49.04	73,307	267.80	1.429	10.00	0.0500	0.350
8	WTP	60.68	84,055	256.02	0.019	0.020	0.5000	0.525
9	WTP	56.52	84,055	256.02	0.076	0.080	2.0000	2.100
10	WTP	106.93	84,055	256.02	0.019	0.033	0.5000	0.875
11	WTP	55.83	84,055	256.02	0.076	0.134	2.0000	3.500
12	WTP	286.85	84,055	256.02	0.019	0.048	0.5000	1.250
13	WTP	149.78	84,055	256.02	0.076	0.191	2.0000	5.000
14	WTP	332.90	84,055	149.44	1.136	1.193	2.5000	2.625
15	WTP	173.83	84,055	149.44	2.273	2.386	5.0000	5.250
16	WTP	161.40	84,055	149.44	1.136	1.705	2.5000	3.750
17	WTP	117.47	84,055	149.44	2.273	3.409	5.0000	7.500
18	WTP	432.98	84,055	149.44	1.136	2.273	2.5000	5.000
19	WTP	315.14	84,055	149.44	2.273	4.545	5.0000	10.000
20	WTP	461.84	76,676	8.90	0.463	0.926	0.0640	0.128
21	WTP	336.13	76,676	8.90	0.463	1.852	0.0640	0.256
22	WTP	50.89	20,259	5.77	0.463	0.926	0.0640	0.128
23	WTP	49.47	20,259	5.77	0.463	1.852	0.0640	0.256
24	WTP	32.64	86,022	6.83	0.437	0.874	0.1289	0.258
25	WTP	31.74	86,022	6.83	0.437	1.749	0.1289	0.516
26	WTP	49.93	23,118	4.38	0.437	0.874	0.1289	0.258
27	WTP	48.70	23,118	4.38	0.437	1.749	0.1289	0.516
Policy Case	N/A	N/A	71,690	52.68	0.045	0.090	0.0576	0.115

† *Note that observations 3, 7, 28, and 29 are dropped from the structural benefit transfer models as described in the text.*

The average cost of salmon angling "P" relevant for each observation in **Table 1** is the travel cost facing a representative household within the market extent, not just the price paid by households that regularly fish. An average travel cost depends on the opportunity cost of time, vehicular depreciation and fuel cost per mile, the distribution of population for a given market extent, and the distribution of places to fish. The last two factors are particularly difficult to assess precisely, and this study relies on simplifying assumptions as follows. Salmon fishing is often considered preferable closer to a river's mouth, due to fish attrition and lower meat quality upstream. Thus a single angling location was researched for each fishery, as a point close to the river mouth with public boat ramp facilities[3]. For each observation's market extent, the top five population centers were used as starting point "hubs," with roundtrip travel times and distances calculated with Google Maps. For small county-level market extents, the single top population center in that county was used. Travel cost was then calculated from the hubs to the angling site and weighted based on hub population. Round trip travel cost for a given hub was:

$$1/3 * (\mathrm{Y}/2,000) * (\text{round} - \text{trip driving time}) + 0.58 * (\text{round} - \text{trip driving distance})$$

The calculation assumes 2000 h worked per year, counts the opportunity cost of time at 1/3 of the average wage rate

[3]The angling sites used for the observations are as follows: 1 and 2 = Buckley Cove Park, CA; 4 and 8 to 13 = Rainier City Park, OR; 5 and 6 = Lake Aldwell, WA; 14 to 19 = Port of Edmonds, WA; 20 to 23 = Town of Willapa, WA; 24 to 27 = Morrisson Riverfront Park, WA. Last, for the Willamette policy case, Cathedral Peak Park, OR.

(Parsons, 2003; p. 285), and uses a national average of $0.58/mile for driving cost (American Automobile Association, 2015).

For the four models tested, preference parameters were calibrated by minimizing the sum of errors between observed and predicted values (with error calculated as percent difference between observed and predicted). Observed and predicted values correspond to WTP (25 observations per model), X (four observations per model), and ΔMCS (one observation per model) associated with the quality changes from Q_0 to Q_1. Calibrations were achieved by crosschecking an evolutionary algorithm with a generalized reduced gradient algorithm, using a multi-start option to avoid local optima. The calibrated parameters, associated minimized errors, WTP, X, and ΔMCS data for the four models are shown in **Table 6**.

Discussion

The TEV estimates of doubling Willamette Spring Chinook using each benefit transfer method are summarized in **Table 7**. There is a remarkable order of magnitude range in values, from $46.41 to $4,370.83 per household. The lowest estimates are from prior meta-regressions. For comparison, **Table 1** values (excluding observation 28) range from $122.44 from Mansfield et al. (2012) to $6,597.65 from Layton et al. (1999).

Point transfer is the most straightforward benefit transfer technique, with a strong underlying study having similar baseline and final salmon populations to the policy case. However, as compared with the Willamette policy case the Willapa study case is about one-tenth the area, with about one-hundredth the human population. Furthermore, it is possible that TEV would reflect more angling use value in the Willapa watershed than in the Willamette watershed, due to a close proximity of the population surveyed to the fishery. Functional transfer, a technique slightly more complex than point transfer, returns an extreme high value prediction. This is attributable to a 100% change for Layton et al. (1999) representing an order of magnitude more fish than the policy case, even using the low baseline model option. In other words, there remains a context mismatch.

Prior meta-regressions revealed both insights and difficulties. Estimates based on both Johnston et al. (2005) and Richardson and Loomis (2009) were sensitive to selections in key methodological and preference variables. Study year in particular stands out for having a large impact in each study but in opposite directions, and thus may be proxying for other unobserved variables. All else equal, the two studies find that higher response rates and use of choice experiments boost values. Estimates utilizing sample means and most recent study year in the meta-data provide the least extreme estimates from **Tables 2** and **3**, and represent best-practices estimates utilizing the meta-regressions. If meta-data for both studies could be recovered, discrepancies between observed and predicted values for the salmon-related observations could be gauged as a further test of how the regressions performed in those instances.

Comparing all three existing meta-regressions, values for increasing abundance for threatened and endangered species appear to be highest, followed by values for increasing

TABLE 7 | Comparing estimates of the total economic value of doubling willamette salmon.

Method	Predicted total economic value per household
Point Transfer	$726.97 Net Present Value (2015$)
Functional Transfer	$4,370.83 Net Present Value (2015$)
Existing Meta-Regression 1 (Johnston et al., 2005)	$46.51 Net Present Value (2015$)
Existing Meta-Regression 2 (Richardson and Loomis, 2009)	$185.84 Undefined payment schedule
Existing Meta-Regression 3 (Loomis and Richardson, 2007)	$89.23 Undefined payment schedule
New Meta-Regression (Model 1)	$1,515.37 Net Present Value (2015$)
Preference Calibration (Model 4)	$305.34 Net Present Value (2015$)

TABLE 6 | Structural benefit transfer calibrated parameters and predicted values.

Run	Model 1: CES functional form; Quality as fish density	Model 2: Linear functional form; Quality as fish density	Model 3: CES functional form; Quality as number of fish	Model 4: Linear functional form; Quality as number of fish
α	4.616E-05	1.217E-04	2.215E-01	0.000E+00
β	2.763E+00	1.012E-01	7.057E-04	2.898E-05
γ	6.689E-02	2.174E+00	3.591E-07	5.673E-03
δ	2.465E-01	6.373E-01	1.040E-06	3.635E-06
Φ	2.327E+04	2.128E+00	2.384E+04	2.645E+05
Ψ	3.318E-05	5.751E-01	1.469E-03	1.006E-04
Sum of Squared % Errors	13.828	11.289	14.361	8.726
Policy case: use value	$0.01	$0.00	$0.00	$2.93
Policy case: trip rate	0.259	0.260	0.259	0.260
Policy case: willingness to pay per household, net present value (2015$)	$397.99	$236.86	$345.44	$305.34
Policy case: willingness to pay per household as if a yearly payment (2015$)	$27.86	$16.58	$24.18	$21.37

salmon abundance, followed by values for aquatic resource improvements in general. However existing meta-regression results are difficult to interpret and compare since the underlying observations mix payment plans, without calculating a NPV using a defined discount rate. Modeled outputs thus also reflect a mix of payment schedules, not strictly yearly payments in perpetuity nor NPVs. Johnston et al. (2005) provide a separate "lump-sum" dummy variable regressor, set to 1 for this paper for clarity in calculating NPV, but this does not actually adjust underlying observations to NPV using a defined discount rate.

A simplistic new meta-regression using two variables has only limited success, but does include more salmon-oriented TEV observations than any other available regression. Of the four models, Model 1 employs the fewest adjustments regarding dropping observations or transforming underlying variables, and has the theoretically expected signs on both variables. The salmon fish change is expressed in absolute numbers, retaining raw information rather than rescaling to percentage. The log format of the salmon change variable conforms with diminishing returns; WTP per additional fish decreases the larger the increase in fish. Modeled WTP is explicitly NPV; underlying observations were discounted to NPV using a discount rate of 7%. If a rapid, rough estimate is needed, the model could be applied to other Western US salmon contexts, with awareness of model sensitivity to relatively few observations and even fewer studies. In particular there is the disclaimer that the model does not accommodate important methodological differences between studies. For example, extent of market is not a regressor and thus the function implicitly assumes no distance-decay effect. It would be possible to re-estimate the function by dropping observations with especially large market extents, if desired. For the policy case, the function returns a relatively large NPV similar to Layton et al. observations associated with much larger fish increases. Dropping Layton et al. observations returns a value about 1/3 as large. It seems that even when controlling for the relatively high baseline and high fish changes considered in Layton et al. within a meta-regression, the Layton et al. observations still represent high TEV estimates.

Structural benefit transfer was the most time intensive method employed for this paper, even testing just two functional forms. An obvious weakness of functional forms as they were tested is that quality must be summarized in a single variable. Furthermore, the quality scale was tied to a maximum observed number rather than a theoretic number. Testing two representations of quality showed the scale based on absolute number of fish to be superior to fish per unit area for both the CES and linear functional forms, based on lower error as reported in **Table 6**. Exploration of combining variables into a quality index, or functional forms with different quality variables, would be possible with future research. Certain observations could also be weighted as more important for the optimization, e.g., use value observations. The linear functional form with quality scaled as absolute number of fish has the lowest error overall, and predicts a use-value that appears to be the most realistic. As compared with the new meta-regression, structural benefit transfer predicts lower estimates. This may be due to the meta-regression explicitly accounting for a separate impact of scarcity on WTP. There may also be some effect of the meta-regression and structural benefit transfer minimizing different calculations of error: sum of squared errors and sum of percentage errors respectively. However, re-running the calibration to minimize sum of squared errors yielded only a slighter higher value, still far less than the new meta-regression models. The structural benefit transfer also has more stable estimates than the meta-regression, despite much variability in the six optimized parameters.

Any of the estimates aggregated over time and over the approximately 1 million households in the basin (as a possible market extent) supports substantial recovery efforts for Willamette Spring Chinook[4]. Since it is already listed under the Endangered Species Act, such quantifications may seem moot. Yet the decision space for recovery is broad and estimates of TEV can inform policy decisions. Economic criteria figure directly into critical habitat designation by 16 U.S.C. § 1533(b)(2). This often reduces designated habitat since costs are easier to estimate than benefits (Duane et al., 2007).

Conclusions

The hypothetical context of doubling a salmon run in the Western US is used to guide passage through the analytic procedures of benefit transfer techniques. Compiling observations from several studies surveying several thousand Western US households, there is a consistently high WTP for increased salmon abundance. Applying benefit transfer tools to these results requires that the analyst consider numerous factors that influence value, which offers perspective beyond what can be learned from a single estimate. With the illustrative policy case as a vehicle, challenges of applying each benefit transfer technique were discussed.

Simplistic benefit transfer methods and previous meta-regressions displayed a variety of weaknesses for the application. The new meta-regression and structural benefit transfer also required a number of judgments. These are detailed in the paper for transparency, and to facilitate further tailoring best suiting future applications. Each method returns vastly different TEV estimates for the policy case as seen in **Table 7**. The new meta-regression is simpler to estimate and includes two separate gauges of resource quality, yet is highly sensitive to modeling

[4]Ideally, survey studies in **Table 1** that provided the foundation for benefit transfer have captured both use and non-use values, in the correct proportions that they exist across households. However, there are relatively few commercial salmon anglers, thus their values may not be well represented by a sample, or observations could have been removed as outliers. Thus, for additional context on this issue, salmon constituted 8% of onshore landed value across all coastal Oregon commercial fisheries in 2013, amounting to about $12.4 million in gross revenue (The Research Group, 2014). The year 2013 was abnormally high compared to the several prior years. There are no known estimates of how Willamette Spring Chinook contribute to the coastal Oregon commercial salmon fishery overall, since the fishery has many other sources, such as the rest of the Columbia system and Oregon's Coast Range. For these reasons welfare impacts on commercial anglers from doubling Willamette Spring Chinook would appear to be relatively small, especially given that costs of fishing would need to be subtracted from gross revenue. Hanna et al. (2006) citing Huppert et al. (2004) discuss the possibility that net profit may even approach zero for Columbia Spring Chinook (one factor noted is the rise of farmed salmon, potentially offset by the popularity of wild-caught brands).

decisions. The structural benefit transfer allows inclusion of more data, produces separate use value and trip rate estimates, and yields more stable estimates over the sensitivity analysis. However structural benefit transfer required more assumptions to execute and was more time consuming. In this paper both methods are utilized solely with preference variables, but meta-regression has the ability to include methodological factors if desired.

Ultimately both forms of meta-analysis are useful for crosschecking values, and the wide variety in estimates of the "true" value should be kept in mind. For the selected policy case the author would recommend use of structural benefit transfer for two reasons. First, the meta-regression was unstable, at least partially due to relatively few studies and observations. Second, the structural benefit transfer allows portraying TEV in the context of use value. According to available data, use value appears to be much less than non-use value. This has potentially important policy implications for fishery management, since commercial and recreational anglers are typically thought of as the main stakeholders. For different policy scenarios, it is possible that a different benefit transfer approach would be preferred—for example if the fishery had dwindled to extremely low numbers as in Jones and Stokes Associates (1990), a point transfer from that study could be justifiable since it is the only one dealing with a highly scarce situation.

For conducting TEV benefit transfer for other marine resources, it would not be obvious which technique would be preferred until a literature review of available studies, as compared with the intended application, is complete. Meta-regression and structural benefit transfer are both sophisticated techniques, and are likely to receive continued development. Their intention is to make benefit transfer more reliable; however both techniques of meta-analysis require a series of judgments which this paper attempts to make clear to facilitate use of the tools and interpretation of their results. Of the two techniques, structural benefit transfer is relatively less developed in terms of potential functional forms and other best practice details. There would seem to be opportunities to combine meta-regression with structural benefit transfer, to leverage the benefits of each and offset respective weaknesses. For example, the empirical behavior of preference variables from numerous studies (e.g., linear or exponential relationship with WTP) could be integrated into development of theoretically consistent utility functions, to provide more reliable guidance on functional form for both techniques.

One of the surprising outcomes of this study is the advantage of proactively applying benefit transfer to glean contextual factors explicit or implicit in prior research. These help plan an original valuation effort to address holes in meta-data. During this journey through benefit transfer, several suggestions emerge for future salmon TEV studies:

- Distinguish between wild and hatchery fish. It is currently unclear whether values are sensitive to the distinction. This is important due to controversies regarding hatcheries. Complicating the issue is that angler harvests are sometimes limited to hatchery fish.
- Differentiate between abundance changes and species loss/recovery in the survey. ODFW recommend population increases in wild Willamette Spring Chinook simply to guard against extinction. Surveys do not appear to have represented the concept of a biologically viable population, thus it is unclear whether value estimates apply solely to increased abundance, with a separate, yet undiscovered value applying to avoid a species loss. None of the studies in **Table 1** includes discussion of threatened or endangered species status for the salmon values in **Table 1**, although Mansfield et al. (2012) do mention such status for other species treated in their choice experiment.
- Include information on substitutes, perhaps in a split sample approach, to test Pate and Loomis' (1997) result (which used external data) that substitutes were of low importance. Optimally, relevant information on substitutes would be discovered through focus group and/or interview research with case study residents, but could include presence of or numbers of other salmon species in the watershed or neighboring watersheds. Salmon density per unit watershed area is another possibility.
- Further test the impact of scarcity and salmon baseline. There is evidence from the new meta-regression that low baseline populations have a positive influence on value, but isolating this was hampered by the correlation of fish baseline with fish change.
- Conduct complementary new use value studies. Despite established econometric methods and widespread interest in the recreational and commercial salmon fishery, relatively few recent studies have quantified these for the Willamette or other areas.

Disclaimer

This manuscript has been subjected to Agency review and has been approved for publication. The views expressed in this paper are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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Supplementary Material

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fmars.2015.00074

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Valuing Multiple Eelgrass Ecosystem Services in Sweden: Fish Production and Uptake of Carbon and Nitrogen

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Valuing nature's benefits in monetary terms is necessary for policy-makers facing trade-offs in how to spend limited financial resources on environmental protection. We provide information to assess trade-offs associated with the management of seagrass beds, which provide a number of ecosystem services, but are presently impacted by many stressors. We develop an interdisciplinary framework for valuing multiple ecosystem services and apply it to the case of eelgrass (*Zostera marina*), a dominant seagrass species in the northern hemisphere. We identify and quantify links between three eelgrass functions (habitat for fish, carbon, and nitrogen uptake) and economic goods in Sweden, quantify these using ecological endpoints, estimate the marginal average value of the impact of losing one hectare of eelgrass along the Swedish northwest coast on welfare in monetary terms, and aggregate these values while considering double-counting. Over a 20–50 year period we find that compared to unvegetated habitats, a hectare of eelgrass, including the organic material accumulated in the sediment, produces an additional 626 kg cod fishes and 7535 wrasse individuals and sequesters 98.6 ton carbon and 466 kg nitrogen. We value the flow of future benefits associated with commercial fishing, avoided climate change damages, and reduced eutrophication at 170,000 SEK in 2014 (20,700 US$) or 11,000 SEK (1300 US$) annualized at 4%. Fish production, which is the most commonly valued ecosystem service in the seagrass literature, only represented 25% of the total value whereas a conservative estimate of nitrogen regulation constituted 46%, suggesting that most seagrass beds are undervalued. Comparing these values with historic losses of eelgrass we show that the Swedish northwest coast has suffered a substantial reduction in fish production and mineral regulation. Future work should improve the understanding of the geographic scale of eelgrass functions, how local variables affect the value of these functions, and how to defensibly aggregate a multitude of economic values.

Keywords: Swedish northwest coast, double-counting, non-market valuation, fish production, nutrient regulation, social cost of carbon, ecological endpoints, *Zostera marina*

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INTRODUCTION

Valuing nature's benefits—either explicitly in monetary or non-monetary forms, or implicitly through laws and cultural norms—is necessary for policy-makers facing trade-offs in how to spend limited resources on environmental protection. Because many of the economic benefits of human development are measured in monetary terms, the estimation of non-market environmental costs, and benefits is becoming increasingly relevant, particularly for the marine environment. The net benefits of coastal development require more information about the economic values associated with *marginal changes* in the benefits provided by the sea, i.e., the types of gradual but persistent—rather than massive and non-marginal—changes we are seeing today in ecosystem function (Arkema et al., 2015). These types of marginal economic values can help society allocate scarce resources for e.g., the establishment of marine protection areas, the development of equitable compensation payments for ecosystem injuries (Cole, 2011), or stimulating environmental markets (Palmer and Filoso, 2009).

Although economists recognize the existence of many types of value [SAB, (Science Advisory Board), 2009; Mace and Bateman, 2011], we focus on human-centric economic values for nature that measure the contribution of certain objects (e.g., ecosystem functions and services) to human well-being. These so-called instrumental values are sometimes contrasted with intrinsic values, which suggest that nature may have value "for its own sake" independent of its contribution to human welfare (Davidson, 2013). Instrumental values are based on what individuals are willing to give up to obtain something else of value, and may be measured in monetary or non-monetary terms. Economic values for nature may capture *use values* directly (e.g., being able to fish) or indirectly (carbon sequestration leads to mitigation of damages from climate change) or even *non-use values,* which is the value an individual may assign to economic goods even if they never have, or never will, use it (non-use values are sometimes further divided into option, bequest, and existence values). This framework is often referred to as Total Economic Value (TEV) (Freeman et al., 2014).

The Millenium Ecosystem Assessment (Alcamo and Bennett, 2003) recognized and categorized several types of benefits provided by ecosystems (Ecosystem Services, or ES), which has been followed-up by additional work by economists to assign value to these services (Kumar, 2010). The ES concept provides a strong theoretical basis for valuing nature's contribution to our well-being and has received increased attention in Europe and Sweden (TEEB, 2010; European Parliament, 2012; Regeringskansliet, 2013; Delgado and Marín, 2015). For example, the European Marine Strategy Framework Directive requires information about the benefits provided by the sea and has led to increased use of monetary estimates for these values (Beaudoin and Pendelton, 2012). A number of frameworks designed to value nature's benefits suggest a focus on three critical links between (1) underlying *ecological functions*, (2) resulting (or intermediate) benefits to society provided by *ecosystem services*, and, finally, (3) the *final economic goods* that provide well-being and that can, moreover, be valued in monetary terms (see **Figure 1**; Mace and Bateman, 2011; Keeler et al., 2012).

However, the challenges in this area, including the obstacles in mapping and classifying sometimes remote services, has resulted in a limited valuation literature (Maes et al., 2012). Delgado and Marín (2015) note that despite the massive increase in ES literature since 1991, the majority focuses on terrestrial landscapes, with only 13% covering the marine environment. Liquete et al. (2013) recommends several indicators for assessing "the capacity, flow or benefit derived" from marine and coastal ecosystem services, while Börger et al. (2014) emphasize the importance of interdisciplinary coordination between marine ecologists, economists, and planers. Delgado and Marín (2015) note that the ES concept is most useful to decision-makers when studies assess and value specific ecosystems or geographic areas rather than generic and large-scale ecosystem service assessments. The authors found a shortage of such site-specific studies for the marine environment and also note ineffective information systems for disseminating research results (e.g., literature databases).

Seagrass beds provide several benefits to society, but are impacted by multiple stressors including nutrient pollution, sediment runoff, dredging, and coastal development (docks, marinas, etc.). The global loss of seagrass ecosystems has led to a decline in key ecological functions such as habitat provision for fish and other organisms, uptake of carbon and nutrients, sediment stabilization, storm protection, etc. (Orth et al., 2006; Waycott et al., 2009). As seagrass functions decline, so too do valuable ES and the resulting economic goods that depend on them such as *food* (e.g., fish and other seafood); *protection of real estate* from coastal erosion; *recreation* (e.g., sports fishing and improved amenity values for swimming including clearer water and stable sandy beaches; Short et al., 2000; Rönnbäck et al., 2007; Barbier et al., 2010; Tanner et al., 2014).

A number of valuation studies have examined the multiple economic goods provided by seagrass and their impact on welfare (see e.g., Cullen-Unsworth et al., 2014), but most limit their focus to a subset of goods, such as *enhanced commercial fishing* using market-based approaches (Watson et al., 1993; McArthur and Boland, 2006; Stål et al., 2008; Bertelli and Unsworth, 2014; Blandon and Zu Ermgassen, 2014; Tuya et al., 2014; Jackson et al., 2015); *improved recreational fishing* (e.g., increased catch rate for species that depend on seagrass; Johnston, 2002; Francis, 2012), or *avoided economic damages* from climate change due to seagrass's ability to sequester carbon (Mangi et al., 2011; Pendleton et al., 2012; Luisetti et al., 2013). A number of studies have used cost as a proxy for value when estimating the benefits of seagrass habitat. An oft-cited study estimates the global value of *nutrient cycling benefits* per hectare provided by seagrass/algae beds, based on the cost of providing equivalent nitrogen-reducing measures such as wastewater treatment (Costanza et al., 1997, see also Costanza et al., 2014). Tanner et al. (2014) valued beach amenity values provided by seagrass based on the potential cost savings to a sand management program, while Thorhaug (1990) highlight the cost of seagrass restoration projects as a proxy for value. Some seagrass studies consider non-economic values by relying on biological proxies (areal coverage, biomass

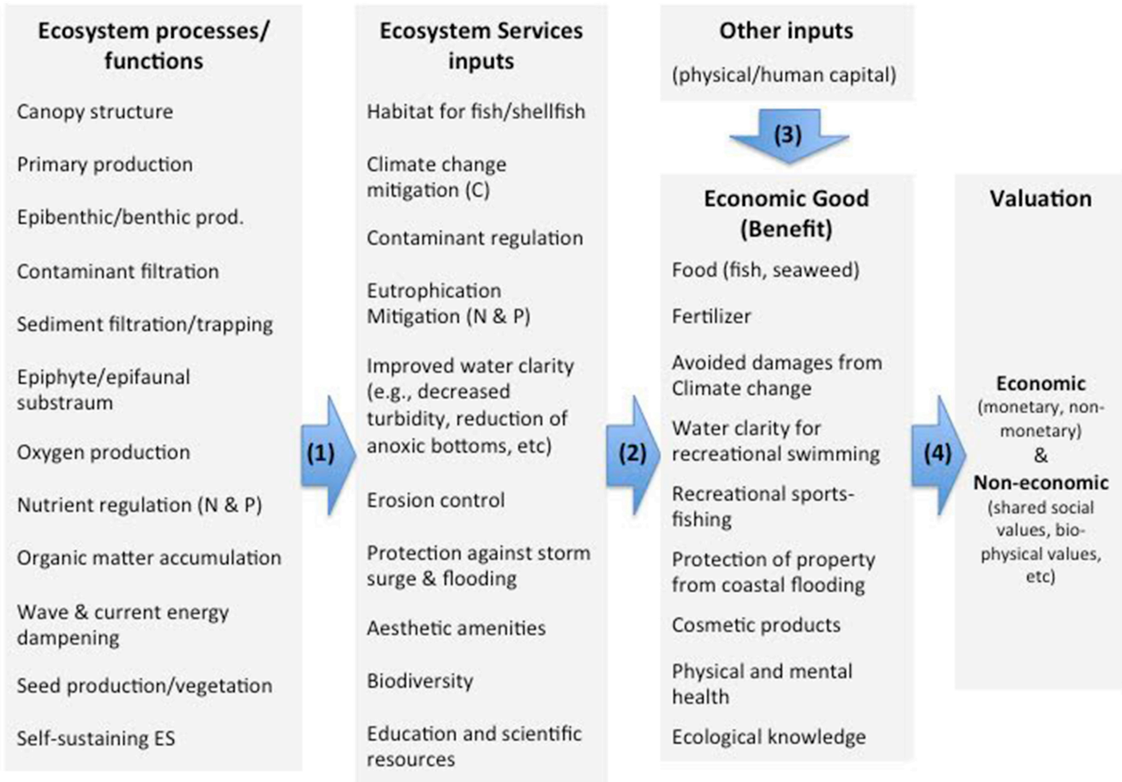

FIGURE 1 | Conceptual approach to identifying and valuing seagrass benefits to society. We follow a four step process: Step #1 maps relevant ecosystem functions and links them to economic goods [arrows (1) and (2)]. Step #2 identifies biophysical changes in ecosystem functions [first column] that are to be valued and Step #3 identifies how these changes affect the flow of multiple ecosystem services and economic goods [arrows (1) and (2)]. Finally, step #4 estimates the value of these multiple changes in economic goods [arrow (4)], which may also be affected by other non-ecosystem inputs [arrow (3)].

of bird and mammal groups that eelgrass supports) or the energy resources invested by nature to satisfy human needs (Plummer et al., 2012; Vassallo et al., 2013). Some studies use the quality or extent of seagrass as a variable when valuing ecosystem services in general from coastal ecosystems (Kragt and Bennett, 2009; Brenner et al., 2010). While we are not aware of studies examining non-use values associated with seagrass, several Swedish studies have estimated positive non-use values for an increased cod population, a species that depends on seagrass (Eggert and Olsson, 2009; Eggert, 2015).

While the economic valuation literature cited above is important, its usefulness for policy assessment is limited (Naber, 2008; Barbier et al., 2010; Bertelli and Unsworth, 2014) in part because they tend to focus primarily on single economic goods. Those that attempt to capture multiple goods provide little guidance on how to aggregate values, a key concern identified in Keeler et al. (2012). Further, landmark studies like Costanza et al. (1997) and Costanza et al. (2014) play a key roll in raising awareness of society's dependence on ES in general, but they do not support improved decision-making, which requires information on the economic value associated with relatively small marginal changes in ecosystems. Moreover, most valuation estimates do not account for the fact that the ecological functions underlying these goods and services vary spatially and temporally, which greatly affects the benefits they provide (Barbier, 2008).

This paper improves upon the "single economic good" approach found in the existing literature for valuing ecosystem services. Our contribution is first to identify links between seagrass ecological functions, ecosystem services, and the multiple economic goods in Sweden to which they contribute; second, to quantify these links using ecological endpoints where possible and an assumed marginal environmental change; and third, to provide an estimate of the monetary values at stake. Including the contribution of ecosystem services to our well-being, even if they are captured with imperfect monetary estimates, will improve the existing decision-making processes, which typically assumes these values are zero. Our approach aggregates multiple values while avoiding double-counting of ecosystem benefits. The double-counting trap occurs when valuing functions instead of final goods or, when summing the value of economic goods that benefit from the same function, and is the result of our weak understanding of the complex interactions of ecosystems (Turner et al., 2010).

We focus on the case of eelgrass (*Zostera marina* L.) on Sweden's northwest coast and estimate an *average marginal value per hectare* that captures the benefits associated with avoiding economic damages from climate change, increasing economic value to commercial fishing, and reducing nitrogen levels. Finally, we discuss the effect of spatial variables in identifying beds that provide relatively greater or lesser value, thus helping decision-makers better assess trade-offs associated with the management of eelgrass beds.

MATERIALS AND METHODS

Study System

Eelgrass is the most abundant seagrass species in the northern hemisphere and plays a critical structural and functional role in many coastal ecosystems. It is an important ecosystem engineer that provides substrate, shelter, feeding, and nursery environments for a large variety of species, some of which are commercially important (e.g., Short et al., 2000; Lilley and Unsworth, 2014). It protects against coastal erosion and increases water clarity through the reduction of wave energy, trapping of particles, and stabilizing of sediments (Orth et al., 2012). It is also important for nutrient trapping and cycling (McGlathery et al., 2012) and contributes to reduced climate impact through sequestration of carbon from the atmosphere (Duarte et al., 2005; Fourqurean et al., 2012).

Eelgrass is the dominant seagrass on the Swedish west coast where it forms dense meadows from 1 to 5 m depth (Boström et al., 2014) that support diverse communities in which 41 fish species (Pihl et al., 2006), 72 algal epiphytes, and 125 species of epifauna have been identified (Fredriksen et al., 2005). The focus area of this study, the Swedish northwest coast, stretches from Gothenburg to the Norwegian border (∼170 km) and includes a complex coastline with fjords and archipelagos where eelgrass is present more or less continuously in smaller meadows in sheltered, soft sediment habitats. Since the 1980s, approximately 60% of the eelgrass has been lost from the Swedish northwest coast (Baden et al., 2003; Nyqvist et al., 2009) due to eutrophication and overfishing (Moksnes et al., 2008; Baden et al., 2012), leading to a decline in valuable ES. In recent decades, water quality measures have reduced the nutrient load and improved water quality along the Swedish Skagerrak coast (SwAM, 2014), slowly improving the conditions for eelgrass growth.

Valuation Approach

We rely on the Keeler et al. (2012) framework as a structure for mapping, modeling, quantifying and monetizing nature's benefits. This framework, together with other approaches in the literature [see e.g., SAB, 2009; Mace and Bateman, 2011; Guerry et al., 2015; Olander et al., 2015], underscores the importance of a stepwise approach. Our conceptual approach follows a four-step process summarized in **Figure 1**.

To map eelgrass ES on Sweden's west coast and link them to economic goods (Step #1) we rely on previous literature that has assessed the types of functions provided by eelgrass worldwide and adjusted them to reflect the conditions on Sweden's west coast. We summarize links between ecological function (e.g., biophysical processes) and the resulting ecosystem services (i.e., indirect benefits to society). For the purpose of valuation without double counting (see "Mapping ecosystem functions…" below), we assume ecosystem functions and process are captured in the value of the final economic good that provides benefits to society, as shown in **Figure 1** (Step #4). Just as GDP measures car production, rather than (intermediate) inputs like steel and rubber, we measure the final economic goods from eelgrass meadows, which is assumed to capture the value of intermediate inputs, such as ecosystem services and other physical/human capital. We acknowledge that *economics goods* may also include "services" such as child care, financial services etc, just as *ecosystem services* may include "goods" such as fish. To avoid confusion we rely on the terminology of the UK NEA such that any output that provides benefits to society is considered an "economic good," see Bateman et al. (2013).

Step #2 defines an anticipated marginal change to the ecosystem services provided by eelgrass for our **valuation scenario**. To capture how a hectare of eelgrass contributes to our welfare we make an assumption about an expected biophysical change that will occur in the future under a business as usual approach. Specifically, we assume permanent conversion (loss) of a one hectare eelgrass bed to bare sediment, where the lost bed is assumed to be mature and delivering a full suit of ecosystem services, e.g., the absorption of a significant amount of carbon and nutrients in the sediment. We assume a (marginal) loss of one hectare from a coastal region with several hectares of eelgrass meadows. The economic benefit provided by that hectare is the avoided loss of multiple economic goods. Economic theory suggests that the selection of a hypothetical valuation scenario should not affect the estimated value, i.e., the willingness to pay (WTP) for a marginal gain (eelgrass restoration) should be equivalent to the willingness to accept (WTA) for the same size loss (eelgrass damage). However, in practice economic studies have found differences when valuing the same change using WTA vs. WTP (see e.g., empirical divergence in Kim and Kling, 2015). This discrepancy is often explained empirically from an ecological or economic perspective by examining how a specific biophysical change affects the provision of an economic good(s) or how an individual may *experience* a given valuation scenario. Because we focus on a relatively small marginal change in eelgrass provision, we assume our estimate is equally applicable for valuing gains or losses.

Under Step #3 we link changes in ecosystem function to changes in value by relying on **ecological endpoints** (Boyd, 2007), which represent meeting points between ecological (biophysical changes) and economic modeling (interpreting how biophysical changes affect welfare). We estimate ecological endpoints to assess the value of *marginal* (i.e., relatively small) changes in economic goods rather than the total value of "having a resource versus not having a resource." In practice this may involve losing a hectare due to coastal development or gaining a hectare from compensatory restoration. The values are less applicable for valuing large non-marginal changes (see Discussion).

To assign monetary values under Step #4 we consider a variety of economic methods for estimating values for the types of market and non-market economic goods in **Figure 1** (see e.g., Freeman et al., 2014). We aim to capture the value of all economic goods arising from the ecological functions provided by eelgrass (see **Table 1**), but in practice we exclude some goods,

TABLE 1 | Summary of underlying ecosystem functions provided by eelgrass on Sweden's West coast and how we value them.

Ecosystem function	Economic good	Beneficiaries	Geographic scale	Explicitly valued in our framework?
1. Structural habitat	Recreation, aesthetic, education	All citizens	Local/regional/global	No. Data not available
	Fish production	Fishers/consumers/ sportsfishers	Local/regional	Yes → increased value to the commercial fishing industry No → Data not available for valuing sportsfishing (recreational) benefits
2. Carbon uptake	Reduced impacts of climate change	Global citizens	Global	Yes → Avoided Economic Damages from floods drought, sea level rise, etc.
3. Nutrient uptake	Recreation (swimming)	Rec. users	Local/regional	Yes → All goods are assumed to be captured through cost of replacing nitrogen-reducing services of eelgras
	Fish production	Fishers/consumers/ sportsfishers	Local/regional	
	Real estate values	Landowners	Local	
4. Reduces wave energy and stabilizes sediment	Recreation (swimming)	Rec. users	Local	No, the potential incremental improvement in secchi depth that benefits recreation cannot be captured due to a lack of data
	Real estate values	Landowners	Local	No. Data not available
5. Provides unspecified functions	Existence or bequest values	Non users	Local/regional/global	No. Data not available

and capture only a portion of others, due to a lack of ecological and/or economic data and robust valuation methods (see section "Mapping Ecosystem Functions …"). The valuation of multiple economics goods is an iterative process that requires careful consideration of the appropriateness of a given valuation method (i.e., what it aims to value, what it is unable to value, the data it requires, etc.). Further, it requires consideration of how to aggregate valuation results from a variety of different methods in a rigorous and defensible manner. Our study ultimately relies on three valuation methods that capture different aspects of monetary value associated with eelgrass ES: avoidance of economic damages, increase in value to commercial fishing, and replacement costs.

The non-market values associated with carbon and nitrogen are based on a transfer of existing values in the literature, rather than primary valuation studies. Such transfers are common when (1) a policy site (e.g., Sweden's northwest coast) exhibits similar characteristics to the study site from which the value is derived and (2) when resources for carrying out a primary study are limited (see Richardson et al., 2015 guidance in the case of ecosystem service valuation). The price of carbon used in our study is based on a transfer of the global value associated with economic damages arising from carbon emissions. Our price of nitrogen is based on a cost transfer, i.e., we examine the costs of nitrogen-reducing measures near the study area (Sweden's northwest coast), and use this cost as a proxy for value.

Economic benefits that accrue far in the future are generally valued less than those that occur today. We account for this so-called positive rate of time preference based on the observation that humans are inherently impatient and prefer to have access to goods and services "today" rather than "tomorrow." The observation is based on the fact that waiting to consume a good/service affects our welfare negatively, i.e., we may die in the future and not have a chance to consume the good or, future generations may have greater wealth at their disposal based on economic growth and therefore their welfare is *relatively* less important than ours, etc. (see e.g., Dasgupta, 2008). Thus, we discount the value of benefits provided by eelgrass that accrue in the future based on an assumed discount rate of 4% based on Swedish economic guidance (SEPA, Swedish Environmental Protection Agency, 2003; SIKA, (Statens institut f;r kommunikationsanalys), 2009). Discounts rates in these types of environmental analyses typically vary between 1 and 7% [see e.g., NOAA, 1999; Moilanen et al., 2009; Mangi et al., 2011]. This means that if the economic estimates for fish production, nitrogen storage or carbon storage are valued at 100 SEK in nominal terms in 20 years (or alternatively in 50 years), we value it at 46 SEK (or 15 SEK), respectively, in present value (2014) terms. Discounting is even used to adjust non-monetary ecological measures of value (see e.g., Cole, 2011 or Sperduto et al., 2003). The value estimates for nitrogen regulation in this study capture future benefits over a 20-year period rather than the 50 years for carbon uptake. Most economic analyses limit the flow of future benefits to those within 20 years because of the uncertainty associated with projecting ecological and economic assumptions too far into the future. In contrast to the local/regional benefits of nitrogen, however, the carbon valuation literature tends to focus on the long-lived nature of global carbon sequestration benefits, which explains our differing time periods.

Mapping Ecosystem Functions to Economic Goods for Swedish Eelgrass

Eelgrass meadows along the Swedish west coast provide a number of important ecosystem functions that link to one or more economic goods that can be valued monetarily. Although we aim to capture the value of all economic goods arising from eelgrass, we can only value goods from three of the ecosystem functions: structural habitat along with both carbon and nitrogen uptake and storage. Our approach is limited due to a lack of ecological and/or economic data and robust valuation methods (**Table 1**).

Eelgrass is an ecosystem engineer that provides structural habitat to a large number of species, which enhances local biodiversity and increases the production of fish and invertebrates. Many of these benefits identified in our conceptual model (**Figure 1**) related to habitat provision are excluded from **Table 1** because they are inherently difficult to value due to a lack of data, e.g., production of medicine and cosmetic products (Farber et al., 2006), improvements in physical health, recreational, aesthetic, and educational benefits. Similarly, a number of unspecified functions may give rise to *existence and bequest values* (which may include biodiversity benefits), but we are unable to value these. We capture instead the benefits to the commercial fishing sector related to the production of gadoid fish (codfish family) and Labridae fish (wrasses). Although benefits could also accrue to recreational sports fishermen, either concurrently or in-place of the commercial sector, we focus on the latter due to lack of data for allocating enhanced fish production across the two sectors. Further, we lack data on how to assign increased fish catch per hectare of eelgrass to individual sports fisherman along the northwest coast, who would likely benefit from increased sea trout (*Salmo trutta*) populations. Although many economically important species rely on eelgrass beds for their life cycle, we are forced to exclude many due to lack of biological or economic data. For example, eel (*Anguilla anguilla*) has been an economically valuable species and highly dependent on eelgrass beds, but its stocks are dwindling and the market has closed in Sweden.

Eelgrass beds provide an important global ecosystem function related to carbon uptake and long-term storage in the sediment. Carbon accumulation leads to a *reduction in climate change impacts* that are captured in our analysis through the social cost of carbon (SCC).

A regional and local function of eelgrass is the uptake and storage of nutrients, which reduces the negative effects of eutrophication in Swedish coastal waters. For example, excessive nitrogen leads to increased production of phytoplankton and decreased water clarity (which affects recreation and property values), increased growth of filamentous algal mats (which may reduce fish recruitment for e.g., plaice), and increased deposits of algal mats on beaches (which affects recreation). Nutrient pollution also decreases oxygen levels in bottom waters which leads to negative impacts on the bottom fauna and commercial fish and crustaceans such as e.g., Norwegian lobster (Rosenberg, 1990; Troell et al., 2005; Stål et al., 2008). On Sweden's west coast, the only positive effects of moderate levels of nutrient pollution are for species with no commercial or recreational value such as the small fish stickle back, shore crabs, and species of ephemeral macroalgae (Pihl et al., 1995, 1999).

The ideal valuation approach for nutrient reduction services would be based on individuals' WTP for explicit and marginal improvements in an economic good such as recreation. For example, sight depth is a useful ecological endpoint that has been used in several studies that demonstrate a WTP by Swedish beachgoers for improved recreational experiences (see Sandström, 1996 and Soutukorva, 2005 for a study of travel expenditures and Söderqvist and Scharin, 2000 for a stated preference study). The recreational value stated by survey respondents in e.g., the Söderqvist and Scharin (2000) study could be linked to the site depth improvement provided by an eelgrass bed to provide a value-based monetary estimate. However, if respondents also *internally* considered benefits to fish populations or carbon sequestration when stating their WTP for the hypothetical water clarity improvement, then we may be double counting benefits and thus over-estimating the contribution of eelgrasses' water clarity-generating functions. Farber et al. (2006) note the difficulty of valuing multiple economic goods that depend, to some extent, on nutrient uptake, such as recreational swimming (benefits from clearer water), recreational fishing (benefits from improved catch rate/size), and food (benefits from increased commercial fish production). Because the nitrogen uptake function contributes to all of these economic goods, a valuation approach should capture as many of them as possible without over-estimating the total contribution of this underlying function. In our case, recreational improvements based on water clarity benefits primarily from the nutrient reduction function, but also from wave energy (see below). However, at present we lack valuation studies of the appropriate geographic scale and detail to be able to isolate and defensibly estimate values for individual contributions of each function to the final economic good. For example, we need data on, among other things, how to apportion Swedish WTP values for water clarity on a per hectare basis.

An alternative approach for valuing nutrient reduction, used in this study, is to value the biophysical change directly (reduction in nitrogen) rather than relying on an ecological endpoint (e.g., improvement in water clarity), which is then used to value a subsequent economic good (e.g., recreation). While values for nitrogen reduction can be found from market prices for nitrogen offset credits (see e.g., Piehler and Smyth, 2011), we believe these prices to be too volative and potenticially distorted and thus rely instead on the costs of mitigation measures aimed at reducing nitrogen. This replacement cost approach has been used frequently in the literature (Gosselink et al., 1974; Notte et al., 2012; Hasler et al., 2014) and relies on cost as a proxy for the value of the economic benefits provided by nutrient uptake, which in our case may include recreational swimming, real estate values, and fish production for some species (**Table 1**). It examines the costs society incurs to avoid damages or, in our case, to replace services with man-made substitutes (e.g., wetland creation that reduces nitrogen concentration). It assumes that if people incur such costs, then the ES must be worth at least what people paid to replace them (or to avoid damages from losing them). Although less rigorous from a welfare economics perspective, cost may be a relevant proxy for value if (1) the man-made alternative replaces the same quantity or quality of services provided by nature, (2) it is the least cost option, and (3) the public would have been willing to incur this cost (Shabman and Batie, 1978; for a more accessible treatment see

Bockstael et al., 2000). We argue for this method for the eelgrass application based on the following context for each citeria: (1) as noted, eelgrass provides a unique and equivalent service related to the trapping and removal of nitrogen (see above); (2) the cost of providing the alternative (nitrogen mitigation measures) varies significantly by watershed, depending on what is feasible given the extent of nutrient pollution. Thus, while there is no single "least cost alternative" we believe that the average price for nitrogen used in this study (**Table 5**) provides a reasoanble approximation of a typical cost [We discuss adjustments to this value below, see "*Spatial (local) affects on values*"]. While economic benefits are also likely to vary between watersheds, we do not have data to determine whether this variation is symmetric with the observed variation in costs; (3) the implementation of a variety of nutrient abatement measures along Sweden's coast to meet Swedish and EU demands (Hasler et al., 2014) provides evidence of a willingness to invest in these types of services. In fact, eutrophication is considered a large problem in Sweden and all of the Swedish west coast is considered to be strongly affected by nutrient pollution and show less than acceptable *ecological status* according to monitoring data and assessment for the EU Water Framework Directive (HELCOM, 2010; SIME, 2014). To meet the requirements of national environmental goals and the EU directives to obtain *good ecological status,* measures to reduce nutrient supply to local watersheds are required in almost all of water bodies along the Swedish northwest coast (SIME, 2014; SwAM, 2014). We note, however, that the existence of relatively cheap man-made alternatives for reducing nitrogen may cause our approach to underestimate the true economic benefit of this service.

One locally important ecosystem function is the reduction of wave energy, which stabilizes sediment through the canopy and rhizome-root mat of the eelgrass bed. This contributes to at least two economic goods: *recreational values* may be enhanced due to reduced sediment resuspension and an *incremental* improvement in water clarity and *real estate values* may be enhanced by avoiding economic damage caused by coastal erosion (**Table 1**). Studies on coastal erosion prevention from eelgrass in Sweden is not available and thus we do not measure enhanced real estate values. However, studies on the Swedish northwest coast suggest that the loss of eelgrass beds has resulted in a local decrease of 1 m in secchi-depth (a measure of water clarity) due to increased sediment resuspension (Moksnes, *unpubl. data*). That is, in some local watersheds this eelgrass function may provide further improvements in secchi depth that are incremental to the water clarity improvements provided by nutrient uptake. However, at present there is a lack of valuation studies of the appropriate geographic scale and detail to identify these benefits on a local scale, and to allow a separation from the same good (recreation) being produced by nutrient uptake.

In summary, our approach values two final economic goods (fish production and reduced impacts from climate change) and one biophysical change directly (nitrogen storage, which is assumed to lead to several economic goods such as improved recreational experiences, fish production, among others). Below we describe our estimating of ecological endpoints and monetary values.

Estimating Ecological Endpoints

Fish Production

Eelgrass beds on the Swedish west coast constitute an important nursery and feeding habitat for a number of commercially and recreationally important species, including Atlantic cod (*Gadus morhua*), whiting (*Merlangius merlangus*), polloch (*Pollachius virens*), herring, eel, flounder, sea trout, and wrasses (Rönnbäck et al., 2007; Stål et al., 2008). In this study, only the gadoid fish and the wrasses were assessed due to limitation of data. Adult wrasses are fished commercially and sold to salmon farms in Norway where they are used to collect ectoparasites.

Due to overfishing, very few adult cod are found along the Swedish west coast today. The juveniles recruited to coastal habitat are primarily from offshore populations in Kattegat and the North Sea. These northwest coast juveniles migrate offshore as they mature and are mainly caught in the offshore fishery in Skagerrak and North Sea (Svedäng and Bardon, 2003; Cardinale and Svedäng, 2004). The total contribution of cod from eelgrass beds along the Swedish northwest coast to this offshore fishery today is estimated to be less than 3% (Stål et al., 2008). Thus, our valuation scenario (see Step #2 above) assumes only a marginal effect on the offshore fishery catch, the costs to the fishing industry, the behavior of the fishery, and the associated regulatory context.

To estimate the negative effect on the fish community from the loss of eelgrass in the study area, we used data from a study on the Swedish northwest coast that compared the community of fish in eelgrass beds with that found in soft bottom areas where an eelgrass beds had been lost in the last 20 years using semi quantitative beach seine samples taken both day and night in four areas (Pihl et al., 2006). We thus assume that the net difference in fish abundance between the two habitats represent a loss in production of gadoid fish and wrasses, and that other juvenile habitat is not available in the local area. This approach is similar to the one used in South Australia to estimate the enhancement of juvenile fish by seagrasses (Blandon and Zu Ermgassen, 2014). Comparing the differences between the habitats it was estimated that the loss of one hectare of eelgrass would results in a loss of 335 juvenile cod and 50 juveniles of other gadoid fish, and 685 adult wrasses (mainly goldsinny wrasse, *Ctenolabrus rupestris*) from the local area (**Table 2**).

To estimate how the loss of juvenile gadoids affected the production of adult fish caught in the fishery, we modeled the growth and survival of the juveniles, and the proportion caught in the fishery in each age-class, using data in the literature for average weight, natural mortality and fishing mortality for each age-class (**Table 3**). This provides a rough estimate of the total biomass of the gadoids caught over a 2–4 year period (until >95% of the biomass had been caught in the fishery). This approach is similar to the production by size-frequency method and modeling size-specific growth and mortality used in earlier studies to value fish production in seagrass beds (Watson et al., 1993; Blandon and Zu Ermgassen, 2014; Tuya et al., 2014). For Atlantic cod, natural mortality in juvenile cod during the first and second year (Age class 0-I) was based on mark-recapture studies along the Norwegian coast (Kristiansen, 2001). Estimates of natural and fishing mortality, and average weight per age-class

TABLE 2 | Estimated ecological endpoints related to commercial fish, carbon and nitrogen per hectare of lost eelgrass.

Variable	Eelgrass	Unveg.	Loss	Unit	Loss adults (kg ha^{-1})
Atlantic cod (juveniles)	365	30	335	No. ha^{-1}	26.6
Whiting (juveniles)	40	0	40	No. ha^{-1}	4.4
Polloch (juveniles)	10	0	10	No. ha^{-1}	0.3
Subtotal					31.3
Goldsinny wrasse (adults)	680	5	675	No. ha^{-1}	
Corkwing wrasse (adults)	10	0	10	No. ha^{-1}	
Subtotal			685		
One-time Carbon in living eelgrass			1490	kg ha^{-1}	
One-time Carbon in sediment (0–25 cm)			13,950	kg ha^{-1}	
Annual carbon sequestration			1664	kg ha^{-1} yr^{-1}	
One-time Nitrogen in living eelgrass			58	kg ha^{-1}	
One-time Nitrogen in sediment (0–5 cm)			162	kg ha^{-1}	
Annual nitrogen accumulation			12.3	kg ha^{-1} yr^{-1}	

Estimates are based on field studies that compare fish abundance and content of carbon and nitrogen in eelgrass beds and in unvegetated soft bottom habitats (see text for details and references).

were based on data from the International Bottom Trawl Survey in the North Sea Skagerrak area (ICES, 2013) using 10-years average values (2003–2012). Since all gadoid fish are caught in the same mixed fishing, the same estimate of fishing mortality was used for all tree species, but species-specific values of average weight per age-class. For whiting, estimated on natural mortality were based on studies in the Celtic Sea (Imelda, 2003), and the mortality of the 0-group was not included since the study indicate that whiting use eelgrasses mainly during their second year (Pihl et al., 2006). Due to the high juvenile natural mortality (e.g., 88% of the juvenile cod died before they were caught in the fishery), the 385 juvenile gadoid fish lost per hectare of eelgrass only resulted in a total loss of 31.3 kg of adult commercial cod, whiting and polloch (**Table 2**). Taken together, the loss of a hectare of eelgrass results in an annual loss of approximately 685 adults wrasses and 31.3 kg of commercial gadoid fish, equivalent to a nominal loss of 7,535 wrasses and 626 kg of gadoid fish over a 20 year period (the total loss of wrasses is adjusted for the multiple year classes of adult wrasses found in eelgrass).

Carbon Uptake and Storage

Seagrass meadows have a unique ability to produce, trap and store organic compounds, making them important sinks for carbon as well as nutrients. In good light conditions, excess photosynthetic carbon fixation is placed directly into the sediments as roots and rhizomes (Duarte and Cebrian, 1996). In addition to this direct source of carbon from seagrass tissues, organic matter from other sources accumulates in the sediments due to the ability of the seagrass canopy to trap particles from the water column (Hendriks et al., 2008). This results in exceptionally high burial rate of organic carbon, and an efficient preservation of the carbon in seagrass sediment is due to low oxygen levels and the dense canopy and rhizomes that protect the carbon deposits from erosion. The carbon buried in seagrass sediment can therefore be over a meter thick and preserved for 100s of years, making the sediment a critical component of seagrass carbon sink (Duarte et al., 2013). When a seagrass bed is lost, most of the seagrass is rapidly remineralized and the carbon returned to the ocean-atmosphere. All or part of the carbon-rich sediment is also eroded and it can be assumed that a large percentage of the carbon in the sediment is also reoxidized (Fourqurean et al., 2012), although proportion that is exchanged with the atmosphere still is unknown (Macreadie et al., 2014). Thus, the very large amount carbon found in the sediment should also be included when assessing carbon sink of seagrass beds (Pendleton et al., 2012; Duarte et al., 2013).

As there are no known studies of carbon sequestration rates, nor of the carbon content of live eelgrass or eelgrass sediment in Sweden, we rely on estimates from other areas. To approximate the carbon sequestration rate of Swedish eelgrass we used an average global rate of 1664 kg C ha^{-1} yr^{-1} (including carbon both from seagrass tissue and other sources) used for eelgrass in the north Atlantic in recent studies (Duarte et al., 2013). For estimates of the carbon content in living eelgrass (1490 kg C ha^{-1}) we used data from a recent study in Virginia, USA (McGlathery et al., 2012). This study also assesses carbon accumulation in a 9-year old restored eelgrass meadow. Using these values, and assuming that on average 25 cm of the carbon rich sediment will be eroded and the carbon reoxidized if the eelgrass bed is lost in northwestern Sweden, approximately 13,950 kg of carbon will be lost per hectare of eelgrass (**Table 2**). Thus, in our valuation scenario the loss of a hectare of eelgrass will lead to an immediate nominal loss of approximately 15.4 ton carbon from live eelgrass and sediment to atmospheric CO_2. Further, we assume an annual loss of carbon sequestration (1.66 ton C ha^{-1} yr^{-1}) that would have occurred had the mature bed survived for an additional 50 years, equivalent to an additional nominal loss of 83 ton carbon.

Nitrogen Uptake and Storage

Similar to carbon, also nitrogen is trapped and stored in eelgrass tissue and sediment. However, much less is known about burial

TABLE 3 | Mortality estimates for gaidoids for estimating ecological endpoints for fish production.

Species	Age (years)	Nat. Mort (Prop yr-1)	Fish Mort (Prop yr-1)	Weight at age (kg)
Atlantic cod	0	0.61	0.00	–
	1	0.52	0.15	0.307
	2	0.50	0.41	0.848
	3	0.38	0.51	2.081
	4	0.21	0.53	3.820
	5	0.18	0.53	5.686
	6	0.18	0.53	7.577
Whiting	0	–	–	–
	1	0.50	0.00	0.165
	2	0.20	0.41	0.264
	3	0.20	0.51	0.347
	4	0.20	0.53	0.472
	5	0.20	0.53	0.622
	6	0.20	0.53	0.687
Pollock	0	0.61	0.00	–
	1	0.52	0.15	0.103
	2	0.50	0.41	0.406
	3	0.38	0.51	0.736
	4	0.21	0.53	0.949
	5	0.18	0.53	1.337
	6	0.18	0.53	1.727

Values of natural mortality, fishing mortality and age-specific biomass used to estimate the production of three codfish that use eelgrass beds as juveniles.

rate and depth, and long-term storage of nitrogen in seagrass sediment (Romero et al., 2006), and no known studies exist from Swedish eelgrass beds. To approximate uptake rates of nitrogen of Swedish eelgrass, we used data from the same study as was used for carbon (McGlathery et al., 2012), which showed an average nitrogen content in living eelgrass of 58 kg C ha^{-1}. This study also showed that the accumulation of nitrogen in the top 5 cm of the sediment of a restored eelgrass bed after 9 years was three times higher than the nitrogen content in the sediment of an unvegetated adjacent area (162 and 51 kg nitrogen ha^{-1}, respectively). Using this data it can be approximated that the average nitrogen accumulation was 12.3 kg N ha^{-1} per year. This is likely a conservative value of the annual nitrogen accumulation since the meadows in the first years had lower shoot density and lower ability to trap and store nutrients than a mature meadow. Since little is known about the available of the nitrogen content at sediment depth below 5 cm in eelgrass beds, only the nitrogen content of the top 5 cm of the sediment is used in this estimate, although nitrogen is likely accumulated to the same depth as carbon, making also this estimate very conservative. A loss of one hectare eelgrass will thus result in an immediate nominal loss of approximately 220 kg nitrogen from live eelgrass and sediment followed by an annual loss of nitrogen uptake (12.3 kg N ha^{-1} yr^{-1}) that would have occurred had the mature bed survived for an additional 20 years, equivalent to an additional nominal loss of 246 kg.

Estimating Economic Values

Fish Production

The TEV of enhanced commercial fish harvest (from avoiding the loss of a hectare of eelgrass) is the sum of the producer and consumer surplus. The former can be captured through profits to the *commercial fishing industry* (e.g., increased catch for harvesters and increased sales for sellers, processors and distributors) and the latter can be captured as the benefit to *seafood consumers* of consuming additional seafood meals or the same meals at a lower price. However, we focus on the producer side as we assume a negligible affect on consumers in our valuation scenario (i.e., the avoided loss of a hectare of eelgrass is unlikely to affect price or quantity in local fish markets). We assume the fishing industry (producer) is not operating at capacity and thus costs associated with increasing production are marginal. Thus, our market-based valuation method relies on a (constant) price and proxies lost value to the commercial fishing industry based on price times quantity. This approach is commonly used in cases where a market-based good (e.g., fish) is dependent on an (eelgrass) ecosystem function such as habitat provision (McArthur and Boland, 2006; Blandon and Zu Ermgassen, 2014; Freeman et al., 2014).

To capture lost value through the supply chain for cod, whiting, and polloch, we rely on the final retail price per kilogram times the increased quantity of fish (adjusted from whole body to filet size). The value captured in the final market is assumed to capture the intermediate losses along the production chain (Just et al., 2005). To capture the lost value to wrasse fishermen, who sell their fish to the aquaculture industry in Norway, we rely on the first landing price per individual times the increased number of individual fish. Value losses to the aquaculture industry are assumed minimal as they can purchase from other suppliers on the margin.

Our valuation scenario assumes an annual nominal loss of 31.3 kg of cod, whiting, and polloch and 685 individuals of wrasse per hectare of eelgrass over the time period 2014–2034. Based on the price data we estimate the present value of future enhanced fish production from a hectare of eelgrass to be approximately 43,500 SEK (5,300 US$) or 3,200 SEK (400 US$) annualized (**Table 4**). Cod and Wrasse make up nearly 97% of the total value.

Carbon Uptake and Storage

To estimate the value eelgrass provides society in terms of absorbing greenhouse gases (including carbon), we rely on estimates for the SCC found in the valuation literature, i.e., a lost hectare of eelgrass can no longer provide carbon sequestration services and thus leads to economic damages. Economic estimates for the SCC, which are developed through Integrated Assessment Models, are based on our best understanding of how carbon emissions affect the climate (increased risk of droughts, floods, sea level rise, etc.) and how these climate changes affect society (e.g., crop damage, property damage, etc.).

TABLE 4 | Estimating per hectare value of eelgrass—Commercial Fish Production.

Fish	Total loss of fish (2014–2034)	Unit	Price[c] (SEK)	Total Nominal Loss 2014–2034 (SEK)	Total Discounted Loss[d] 2014–2034 (SEK)
Atlantic cod	532	kg ha^{-1}	101	23,076	15,681
Whiting	88		64	2433	1653
Polloch	6		89	229	155
Subtotal	626[a]			25,738	17,489
Goldsinny wrasse	7425	no. ha^{-1}	5	37,125	25,227
Corkwing wrasse	110		10	1100	747
Subtotal	7535[b]			38,225	25,975
Total	–	–	–	63,963	43,464
Annualized	–	–	–	–	3198

[a]*Loss of fish biomass (kg) adjusted from whole body size to filet (reduction in kg by 57%). Conversion factors based on (EUMOFA, 2013).*
[b]*Loss of wrasse individuals adjusted to reflect biannual harvest (2 years to maturity) and multiplied by landing price.*
[c]*Prices for cod and polloch based on actual retail prices from 2009–2014, while whiting is estimated based on ratio of landing value to retail price for the other two species (Sannino, Valentina, Personal Communication, European Market Observatory for Fisheries and Aquaculture products (MOFA). November 24 and 25). Wrasse prices based on personal communication, Swedish University of Agricultural Sciences, Department of Aquatic Resources, Lysekil, Sweden.*
[d]*Economic values based on 4% discount rate over a 20 year period. Annualized value spreads total impact over time (20 years) in constant annual amounts.*
1 US$ = 8 SEK.

TABLE 5 | Estimating per hectare value of eelgrass—Carbon and Nitrogen storage.

Input	Quantity *(t. C (ha^{-1}) (kg N (ha^{-1})*	Price (SEK/ton C SEK/kg N)	Time horizon	Total nominal loss (SEK)	Total discounted loss[c] (SEK)
Carbon in living eelgrass	1.49	948[a]	2014	1413	1413
Carbon in eelgrass sediment	13.95		2014	13,227	13,227
Annual carbon sequestration	1.66		2014–2064	78,885	35,248
Total (2014–2064)	98.6	–	–	93,524	49,887
Annualized	–	–	–	–	2322
Nitrogen in living eelgrass	58.0	193[b]	2014	11,194	11.194
Nitrogen in eelgrass sediment	162.0		2014	31,266	31,266
Annual nitrogen sequestration	12.3		2014–2034	47,478	33,553
Total (2014–2034)	466	–	–	89,938	76,013
Annualized	–	–	–	–	5593

[a]*Price of carbon is based on an average of values found in the literature for the Social Cost of Carbon, values ranged from $5 to $312 (Pearce, 2003; Stern, 2007; Tol, 2009; Macreadie et al., 2014; Revesz et al., 2014). We assume emission occurs in 2020 and damage occurs in the period 2014–2064.*
[b]*Price of nitrogen based on average annual cost of replacing the nitrogen-reducing function provided by eelgrass in watersheds on Sweden's west coast (Salöfjord, Askeröfjord, Marstrandfjorden, Hakefjord, Stigfjorden, Skärhamn, Kalvöfjorden, Malöströmmar), which range from 22–435 SEK. For watersheds with multiple nitrogen-reducing measures, we consider the cost of each measure individually and the associated annual effectiveness (Swedish Water Authority (SWA), 2015).*
[c]*Economic values based on 4% discount rate over a 50 year (carbon) or 20 year (nitrogen) period. Annualized value spreads total impact over time (20/50 years) in constant annual amounts.*
1 US$ = 8 SEK.

The SCC represents the present value of the annual future monetary damages resulting from emitting an extra ton of CO2, compared to a Business As Usual scenario (Revesz et al., 2014).

Based on a review of SCC estimates (see footnote **Table 5**), we apply an average value of the SCC of 948 SEK (127 US$) per ton of carbon absorbed. Given our assumed nominal loss of 98.6 t carbon storage capacity during the period 2014-2064, we estimate the present value of the future flow of carbon removal benefits derived from a hectare of eelgrass to be approximately 49,900 SEK (6,100 US$) or 2,300 SEK (280 US$) annualized (**Table 5**).

Nitrogen Uptake and Storage

To estimate the economic value associated with nitrogen uptake and storage provided by eelgrass, we rely on the actual costs of nitrogen reduction measures undertaken on Sweden's northwest coast. This replacement cost valuation method captures the difference in costs associated with reaching a nitrogen reduction target under two scenarios: (1) relying on the ecosystem function provided by eelgrass or (2) relying on a man-made alternative. Our target is kilograms of nitrogen stored in the sediment and in living eelgrass tissue and annually by a hectare of eelgrass. Since the cost of scenario (1) is zero, we estimate the difference (value) as the cost of implementing nitrogen-reducing measures in the study area, accounting for their annual effectiveness.

Using a database, we identify several nitrogen-reducing measures undertaken in coastal watersheds on the northwest coast of Sweden with documented eutrophication problems, including construction of wastewater treatment plant, wetland creation, and catch crops (Swedish Water Authority (SWA), 2015). Our dataset assumes managers select feasible measures for a given watershed and then select the least cost option, which is based on the average cost for that measure. The average annual cost effectiveness for removing nitrogen varies from 22 to 435 SEK per kilogram nitrogen per hectare per year (2010 SEK) depending on the measure, with an average cost of 193 SEK (25 US$), which was used in the calculation.

In our valuation scenario, eelgrass removes a (nominal) total of 466 kg of nitrogen over the period 2014–2034. Given that the average total cost to society of removing an annual equivalent amount of nitrogen (in present value terms) is 193 SEK, we estimate the value of nitrogen storage derived from a hectare of eelgrass to be approximately 76,000 SEK (9280 US$), or 5600 SEK (680 US$) annualized (**Table 5**).

RESULTS

We present total economic benefits that arise over the time frame of our analysis (nitrogen and fish benefits over 20 years, carbon over 50 years). Because the flow of future benefits associated with carbon, fish, and nitrogen occur at different times in the future we standardize them to *present value* through discounting. We also provide an annualized amount, which approximates an annual value by spreading the total impact over time in constant annual amounts. Based on a 4% discount rate we estimate the average marginal per hectare value of eelgrass services over time to be approximately 170,000 SEK in 2014 (20,700 US$), or 11,000 SEK (1300 US$) annualized (**Table 6**). Based on the economic goods valued in this analysis, nitrogen uptake and storage represents 46% of the total value, followed by climate mitigation (30%), and fish production (25%). The commercial value of cod (∼16,000 SEK) represents only 9% of the total value.

DISCUSSION

In this study we developed an interdisciplinary framework for valuing the contribution of eelgrass habitats to human well-being on the west coast of Sweden. Our approach considers the value of three ecosystem functions—structural habitat for fish and uptake of carbon and nitrogen—and aggregates the monetary values associated with the resulting economic goods. This approach differs from earlier valuation studies of seagrasses by capturing multiple economic values—reduced climate change impacts, increased commercial fish production, and reduced eutrophication—rather than focusing on a single economic good. Our results suggest that if a hectare of eelgrass is lost and the habitat transformed to unvegetated bottom where the top

TABLE 6 | Summary of the estimated economic value provided by a hectare of eelgrass on Sweden's West Coast.

Economic good	Biophysical change valued in analysis (Nominal)		Economic value captured	Total average value per hectare[a] (SEK Annualized)	Total average value per hectare[a] (2014-2064) SEK
Food (Commercial fishing)	626	Total loss of cod fishes for commercial production (2014–2034), kg per hectare	Based on lost value to the commercial fishing industry, including fishermen, processors, distributors, retailers from multiple cod fish species	1287	17,489
	7,535	Total loss of wrasse fishes (2014–2034), number of individuals per hectare	Based on lost value to the supplier of the aquaculture industry (fishermen)	1911	25,975
Climate mitigation	98.6	Total loss of carbon storage capacity (2014–2064), including a one-time loss (15.4 t C/ha^{-1}yr^{-1}) and re-occurring annual loss (1.66 t C/ha^{-1}yr^{-1})	Based on avoiding the global economic damages of climate change (floods, droughts, famine, sea level rise, etc), as captured by the "social cost of carbon" (SCC)	2322	49,887
Nutrient regulation	466	Total loss of nitrogen storage capacity (2014–2034) including a one-time loss (220 kg N/ha^{-1}yr^{-1}) and re-occurring annual loss (12.3 kg N/ha^{-1}yr^{-1})	Based on the cost to society of replacing the ecological service of nutrient regulation by eelgrass, where cost is a proxy for welfare benefits of this regulation	5593	76,103
			Totals	11,114	169,364

[a] *The table presents total economic impacts that arise over time (2014–2024 for nitrogen and fish; 2014–2064 for carbon), standardized to present value. Present value adjusts the value of an impact—e.g., a cost or benefit that accrues over time—to today's value to allow for comparison. We also provide an annualized amount, which spreads this total impact over time in constant annual amounts (50 years for carbon, 20 years for fish/nitrogen) using a 4% discount rate.*
1 US$ = 8 SEK.

5–25 cm of the sediment is eroded, it would result in a variety of losses including: a reduced yield of approximately 626 kg of gadoid fish and 7535 individual wrasses, a reduction of 99,000 kg (98.6 tons) of sequestered carbon and 466 kg of nitrogen over a 20–50 year period. Based on these ecological endpoints, we estimate the total present value of the flow of future benefits from the resulting economic goods to be approximately 170,000 SEK ha^{-1} (equivalent to ∼20,700 US$ ha^{-1}). This value is at the upper end of other monetary estimates in the literature for seagrasses, but may nonetheless be considered conservative given our cautious approach for estimating ecological endpoints and for aggregating values in our framework. As better ecological and economical data becomes available, and interdisciplinary valuation methods improve, we could expect this value to increase.

Intended Use of Economic Value Estimates

Our valuation framework is considered conservative because it acknowledges current limitations in our ability to translate *all* eelgrass functions into economic goods that impact our welfare. We believe that a conservative approach that strives to avoid double-counting of ecosystem benefits is preferable to inflated values that are hard to defend and are easily misinterpreted by policy makers and/or the public. A less conservative approach might try to include other values, in particular non-use or existence values associated with e.g., biodiversity and ecosystem resilience, which require stated preference valuation approaches. But combining these survey approaches with those in our framework raises challenging methodology issues because we cannot be sure whether survey respondents account for other seagrass benefits already captured in our framework, when stating their WTP for a given and defined seagrass improvement (or, theoretically, their WTA a seagrass decline). Parsing out and aggregating these types of values is the biggest challenge in a framework aiming to capture multiple economic values.

Our value estimates are useful for policy assessment by coastal managers as they help identify benefits that eelgrass provides society "on average at the margin." They may be used, for example, to decide whether to allow partial losses (from e.g., dredging) or to assess the value generated by off-setting compensation projects (e.g., eelgrass restoration). Valuation estimates can support arguments for establishing Marine Protection Areas when the benefits of such designations outweigh the costs and, more generally, can inform the "preservation vs. development" debate in coastal areas. The value associated with damaged resources is critical for implementing the Polluter Pays Principle (PPP), which underlies several EU Directives and suggests that operators, not the government, are responsible for internalizing the cost of environmental damage (e.g., European Commission, 2011). The PPP is particularly salient when motivating and improving the use of environmental compensation measures to achieve the Not Net Loss initiative in the EU (Cole, 2011; EEB, (European Environmental Bureau), 2014). Finally, the values in this study may also support market solutions such as Payment for Ecosystem Services schemes (Palmer and Filoso, 2009). For example, the lost value from damaged eelgrass beds may be a useful input in the future development of habitat banking markets to offset coastal development impacts.

A potentially useful and local application of our estimate is to improve existing Swedish policy related to compensatory offsets for negative impacts on eelgrass beds. Currently, operators that cause residual damage are required to pay a "fisheries fee" to compensate for the loss in fish production, which is then used to restore essential fish habitat. In theory, the fee represents a financial cost to operators that ostensibly captures the external cost on the fishery, i.e., the lost value in fish production that would otherwise be provided by eelgrass. However, there is little guidance on how to estimate values or to scale fair compensation payments. As a result, current approaches are *ad hoc,* with some payments based on estimates of secondary production of fish food and commercial market prices, some based on replacement cost of farmed-raised juvenile fish. Historically, compensation payments have varied from 10,000 to 100,000 SEK ha^{-1} or 1,400–14,000 US$; *pers. com.* Administrative County Board Västra Götalands Län). These fees likely underestimate the total environmental costs on society. An improved approach would scale a compensation payment based on the multiple economic benefits eelgrass provides (thus offsetting the welfare loss), rather than focusing exclusively on fish, which represents only 25% of the economic benefits estimated in this analysis.

Fish Production

Previous valuation studies of seagrasses have focused almost exclusively on a single function: provision of nursery and feeding habitat for fish production. Our study estimates the commercial value of Atlantic cod, whiting and polloch. Estimates may appear low but this is due to high natural mortality of juveniles and relatively low market prices (equivalent to ∼2100 US$ ha^{-1}). We also include value for small wrasses (∼3100 US$ ha^{-1}), which obtain a high price in the aquaculture market, where they are used to remove ectoparasites from salmon. We consider the value of fish production in Swedish eelgrass beds to be conservative for several reasons. First, we only value 5 of the 41 species of fish that rely on eelgrass beds on the Swedish northwest coast during some stage of their life-cycle (Pihl et al., 2006). Economically important eel, herring, and sea trout are excluded due to lack of data. We also exclude commercial species that do not use eelgrass habitats directly, but may benefit indirectly from the production of food in eelgrass beds, which is exported from the habitat during the winter when many species migrate to deeper unvegetated areas. Second, the estimated abundance and value of cod and the other gadoid fishes are likely low from an historic perspective considering that the biomass of these species has decreased by over 90% since the 1970s along the Swedish west coast due to overfishing (Svedäng and Bardon, 2003). Thus, the economic value provided by eelgrass beds' nursery function could increase substantially if these stocks recover. Importantly, we focus our assessment on commercial value, but a recent report suggests that if the enhanced fish production along the Swedish west coast (for e.g., cod and trout) were allocated instead to recreational sports fishermen (in which over 10% of the population participates), the benefits to society may be greater (Paulrud, 2008). Finally, our approach relies on price as a mechanism (proxy) for estimating economic value, which is only able to capture the portion of underlying value realized in a market (Fischer et al., 2011) and thus excludes non-use values the public may hold for fish. The fish production value could be improved by developing a bioeconomic model (see e.g., Rabassó and Hernández, 2015 for an example that empirically links seagrass degradation to commercial aquaculture value).

Using an annual value of fish production in Swedish eelgrass to compare the results with estimates from other seagrass systems, we find that the total commercial value of the five fish species valued in our analysis (equivalent to ∼400 US\$ ha^{-1} year^{-1}) is within the same range as the total commercial value of 25 fish species extracted from seagrass habitats at the island of Gran Canaria in Europe (866 € ha^{-1} year^{-1}; equivalent to 771 US\$ ha^{-1} year^{-1}; Tuya et al., 2014), the commercial value of three shrimp species found in seagrasses in Queensland, Australia [183–3687 A\$ ha^{-1} year^{-1} or 232–4675 US\$ ha^{-1} year^{-1}, inflated with the CPI; US Bureau of Labor Statistics (BLS), 2015; Watson et al., 1993], and the average value of commercial and recreational fish and invertebrates using seagrasses in South Australia (133 A\$ ha^{-1} year^{-1} or 129 US\$ ha^{-1} year^{-1}; inflated with the CPI, McArthur and Boland, 2006). However, our values are lower than a recent estimate of the total commercial value of 13 fish species using seagrasses in South Australia (230,000 A\$ ha^{-1} year^{-1} or 178,000 US\$ ha^{-1} year^{-1}; Blandon and Zu Ermgassen, 2014).

Carbon Sequestration and Climate Mitigation

The importance of seagrasses in the role for uptake and long-term storage of carbon has recently gained much attention, with most of the available literature focusing on sequestration rates of different species (e.g., Duarte et al., 2005; Fourqurean et al., 2012; Macreadie et al., 2014). However, relatively few studies have assessed the monetary value of carbon sequestration in seagrasses (but see Mangi et al., 2011; Pendleton et al., 2012; Luisetti et al., 2013), limiting comparison with the present. Mangi et al. (2011) assessed the value of climate mitigation by seagrasses on the Isles of Scilly, UK, based only on carbon fixation rates in seagrasses (as a proxy for sequestration), obtaining an annual monetary value of approximately 77 £ha^{-1} year^{-1} (or 130 US\$ ha^{-1} year^{-1} inflated using the CPI), which is similar to the present annualized value found in this study (∼280 US\$ ha^{-1} year^{-1}). However, in the present study we also took into account the carbon stored in the top 25 cm of the eelgrass, which constituted 82% of the annual value, and 33% of the total value over a 50 year time period (i.e., approximately 6000 US\$). Thus, the carbon found in the sediment of old eelgrass beds constitute significant part of the total carbon sink (Fourqurean et al., 2012; Duarte et al., 2013) and should be included in valuation studies of climate mitigation when the sediment is expected to erode (Pendleton et al., 2012). In the present study we used a conservative estimate assuming that only 25 cm of the sediment would erode, due to lack of data for eelgrass. In comparison, Pendleton et al. (2012) assumed that 100 cm of sediment would erode in a recent attempt to estimate the global emission of carbon from degraded seagrass beds. Thus, the value of climate regulation from Swedish eelgrass beds may increase as data become available on carbon content and erosion depth of the sediment.

The SCC is a well-accepted method for estimating welfare impacts from carbon emissions. Although it is subject to a variety of uncertain ecological and economic assumptions in existing climate models, it represents the best available monetary valuation approach (Revesz et al., 2014). Our value of 127 US\$ per ton of carbon lies within the interval seen in other studies, which range from 5 to 312 US\$ per ton of carbon (see **Table 5** footnotes).

Nitrogen Regulation

Our analysis indicates that nitrogen uptake provides the highest value of the ecosystem services assessed (equivalent to approximately 9500 US\$ ha^{-1} or 46% of total value). However, because little is presently known about burial and long-term storage of nitrogen in eelgrass sediment (Romero et al., 2006) we used a conservative approach for estimating the ecological endpoint that underlies this value, based on nitrogen accumulation estimates from recently restored eelgrass beds (which have lower capacity for trapping organic material and nutrients than an older beds) and only includes the top 5 cm of the sediment, due to limitation of available data. If we had data to support nitrogen accumulation and erosion down to 25 cm depth (as was used for carbon sequestration) the value of nitrogen regulation would nearly triple to over 24,500 US\$ ha^{-1}. Due to discounting, a 50 year horizon for nitrogen instead of the 20 used in this study would only increase our estimate by 25%, to ∼10,300 US\$ ha^{-1}. Importantly, the average cost of feasible nutrient abatement measures used in this study shows significant variation (22–435 SEK ha^{-1}) suggesting that *local values* for eelgrass beds may differ by 20-fold between watersheds. Since nutrient pollution and uptake by eelgrass often occur on a local scale, and since both the capacity of eelgrass to accumulate nutrients and the cost of undertaking equivalent measures can vary strongly between watersheds, it is important to consider qualitative adjustments based on how local factors influence our average value estimates for nutrient regulation [see "*Spatial (local) affects on values*" *below*]. Further, we assume managers select the least cost option from among the feasible alternatives, but if more costly options are selected due to e.g., ancillary recreational benefits, our approach may overestimate nitrogen values. Thus, the next step in improving our valuation approach could be to use a watershed-specific model for estimating spatially explicit "least cost" estimates (e.g., Hasler et al., 2014).

Given that nutrient regulation is the most valuable ecosystem service in our study, it is somewhat surprising that it has received so little attention in the seagrass literature, which has focused primarily on fish production and, more recently, carbon sequestration. To the best of our knowledge, the only other similar value estimate in the literature is a global estimate of nutrient cycling by seagrass/algae beds of approximately 26,200 US\$ ha^{-1} (Costanza et al., 2014), which also used a replacement cost approach. However, this may be considered a less robust estimate given the study's "local to global" extrapolation of values.

There are several implications of this study's cost-based approach for capturing nitrogen uptake. Besides being less rigorous from a welfare economics perspective, the values are somewhat challenging to interpret. Our estimates suggests that some local governments are willing to incur costs but does not say whether some individuals may, in fact, be willing to incur *greater* costs. If so, we are likely under-estimating nitrogen reduction values, all else equal. Alternatively, if the "political willingness to pay" costs used in this study are, in fact, higher than what individual citizens are willing to pay, we may be over-estimating nitrogen reduction values, all else equal.

Further, we cannot be sure which welfare benefits a government had in mind when deciding to undertake nitrogen-reduction measures (e.g., *direct benefits from* improved water clarity, and/or *indirect benefits* such as simultaneous reductions in other environmental contaminants). As such, cannot say for certain what type of value we have captured nor what it implies about our subsequent welfare after the measure has been undertaken. Note further that the current approach assumes that the value of nitrogen retention (1) goes up when we put more nitrogen in the system (the absolute cost of removal increases, even if marginal cost may decline) and (2) goes down when we remove nitrogen from the system or we become more technologically efficient at creating human substitutes for nitrogen-reduction (see e.g., Notte et al., 2012).

Importantly, even if we could isolate WTP for water clarity under the ideal valuation approach described in the Materials and Methods section, we cannot add this to our cost-based estimates due to double counting (nitrogen reduction measures likely capture water clarity improvement) and methodological concerns (e.g., mixing two fundamentally different valuation methods, see Freeman et al., 2014). However, if data existed it is possible to present "side-by-side" value-based and cost-based estimates as an informal validity check. That is, these valuation approaches complement each other in the sense that they both provide evidence of a WTP for economic goods that are dependent on eelgrass ecosystems.

Spatial (Local) Affects on Values

The estimated marginal values in this study are designed for use along the Swedish northwest coast to capture the value of losing/gaining one (marginal) additional hectare out of many. A truly robust marginal value, however, would require information on the current and future baseline condition of a resource, how a given action/policy may affect this over time, how individuals experience a specific valuation scenario and how spatial variables affect values (Turner et al., 2010). Therefore, our estimate is more accurately considered an "average marginal" value that attempts to "average-out" these various factors, which affect the benefits provided to society.

To improve the relevance and accuracy of our estimates for specific policy applications in specific coastal areas/harbors, we suggest consideration of some basic "rules of thumb." For eelgrass, there are several contextual variables that could have large effects on the local per-hectare value for all ecosystem functions, with carbon sequestration being an obvious exception given that it provides global benefits. In general, an eelgrass bed will have a higher economic value if the ecosystem function is "locally limiting" for the production of the ecosystem service, and/or if the economic good is in short supply. For example, if nursery habitats for juvenile cod are in short supply in a region and limiting for the recruitment of cod, the eelgrass bed will have a higher value than in an area with a surplus of nursery habitats. Similarly, the value of nutrient accumulation of eelgrass will be higher in a watershed that requires expensive nutrient abatement measures than in an area that does not require any measures (e.g., is already in compliance with water quality standards), or where the available abatement measures are less expensive to implement. An eelgrass bed that improves the water clarity locally (e.g., by decreasing sediment resuspension) will also have a higher value in an area where the demand for clear swimming water is high and in short supply, than in an area with little demand, far away from cities and tourists. This type of *qualitative adjustment* will strengthen environmental decision-making by identifying beds that provide disproportionally greater or lesser value than others.

Estimating Ecological Impact and Monetary Value of Historic Losses (1995–2015)

The per hectare estimates in this study can be used for a rough assessment of the impact on ecosystem functions and potential social welfare loss associated with the documented decline in eelgrass along the Swedish northwest coast. While we recommend that the monetary estimates in this study are used primarily for policy assessment at the margin rather than large-scale changes in the resource, applying the estimates to this historic loss nonetheless underscores society's dependence on this ecosystem and highlights the instrumental values at stake. Caution is warranted, however, when interpreting and using this historic value loss for two reasons: (1) we are valuing a large 60% change (loss) in the resource using an estimate that assumes a small marginal change and (2) we are extrapolating an average value that fails to capture context-dependent variables (see Bockstael et al., 2000 critique of the Costanza et al., 1997 paper).

The document 60% loss of eelgrass from the Swedish northwest coast since the 1980s (Baden et al., 2003; Nyqvist et al., 2009) is equivalent to approximately 11,500 ha (Moksnes et al., 2016). Assuming the loss occurs instantly in 1990 (the actual loss pattern is unknown), and using our per hectare ecological endpoints from **Table 2**, we estimate that the eelgrass decline between 1990 and 2015 resulted in a total loss of ∼9000 tons of gadoid fish catches, 197 million wrasses, and 422,000 and 6000 tons of sequestered carbon and nitrogen, respectively. To put these numbers in perspective, the total loss of cod catches resulting from the loss of eelgrass (7650 tons) is similar to the total 2013 annual catch of cod in Swedish waters (7895 tons), which includes the Baltic Sea (SwAM, 2012). The total loss of carbon and nitrogen storage is ∼10 and 3 times larger than the annual river supply of organic carbon and nitrogen to the Swedish northwest coast (∼44,000 and 2500 tons, respectively, Skagerrak in 2012; SIME, 2014). Thus, the loss of eelgrass has had a substantial impact on fish production and the recycling of carbon and nitrogen along Swedish northwest coast. That changes in eelgrass cover can have large effects on the recruitment of cod stocks is supported by an increasing number of studies (Warren et al., 2010; Lilley and Unsworth, 2014).

Based on our per hectare value estimates and the historic loss of eelgrass, the total nominal value associated with the lost economic goods is approximately 3.1 billion SEK (378 mil. US$). This includes a range of 0.62–8.3 SEK that accounts for varying assumptions about the price of fish, carbon, and nitrogen; and the actual size of the historic loss (Moksnes et al., 2016). However, this monetary estimate is not adjusted to reflect the fact that the impact on human well-being (i.e., value) depends, in part, on when an economic good or service is consumed or experienced. If we compensate for the time that these economic goods were not (historically) available to society by *compounding* the historic lost value at a 4% rate (the same rate used for discounting future values), the total net present value is approximately 5.2 billion SEK (with a range of 1.0–13.8 SEK) for the period 1990–2015.

Limitations and Future Research

In addition to the specific limitations relating to ecological endpoints and valuation of the three economic goods, there are also some general risks and uncertainties associated with our analysis. First, the use of a single monetary figure may suggest a false precision, which would under-state the uncertainty and lead decision-makers to mis-interpret the nuances and limitations of these estimates. Further, the current state of knowledge requires that we simplify complex ecological systems into single economic goods that we are able to value. By failing to capture the inherent complexity, such as tipping points and thresholds over or under which certain ES are no longer provided, our valuation estimates may represent proxies at best, or imprecise and variable estimates at worst. Finally, our valuation scenario—the conversion (loss) of one hectare of eelgrass to bare sediment—is a necessary but subjective assumption that affects our value estimate. It may overestimate losses if e.g., another vegetative habitat eventually colonizes the lost eelgrass area and can provide some non-zero level of services related to fish production or carbon/nitrogen uptake.

Continued work in this area will likely improve our ability to measure economic damages from climate change and the WTP to avoid nutrient pollution by better capturing the value of these externalities in the price of carbon and nitrogen. Future research should develop more information on (1) the geographic scale of eelgrass functions (e.g., fish habitat, nutrient uptake, sediment stabilization, etc.), (2) how these link and contribute in a meaningful way to our welfare and (3) how to defensibly aggregate the values of the multiple and subsequent economic goods.

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Factors Influencing Willingness to Donate to Marine Endangered Species Recovery in the Galapagos National Park, Ecuador

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Willingness to donate (WTD) money for the conservation of endangered species may depend on numerous factors. In this paper, we analyze data from a survey given to tourists visiting Ecuador's Galapagos National Park and Marine Reserve to investigate determinants of their WTD toward the conservation of two marine endangered species-the scalloped hammerhead shark (*Sphyrna lewini*) and the green sea turtle (*Chelonia mydas*). Specifically, we use regression analysis to analyze the influence of attitudes and beliefs toward species conservation, levels of concern for specific species, recreational motivations, and past donation patterns on WTD, while also controlling for individual characteristics such as age, gender, place of residence, and other demographics. Additionally, we evaluate the sensitivity of WTD to the species being protected by conservation efforts. Our results demonstrate that specific concern about the species, beliefs about donating to the protection program, and past donation behavior significantly influence the intention to donate money toward the recovery of the two marine endangered species. The likelihood of donating to green sea turtle conservation efforts is marginally higher than for hammerhead sharks, possibly due to its more charismatic nature. In contrast, visitors who are more willing to donate for shark conservation appear to be those with a strong desire to see them in the wild. The results provide useful information on the heterogeneity of tourist preferences toward donating to species conservation efforts, which has broad implications for resource agencies seeking ways to fund conservation actions.

Keywords: marine endangered species, donation behavior, conservation attitudes, attitude-behavior modeling, eco-tourism, Galapagos National Park, scalloped hammerhead shark, green sea turtle

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INTRODUCTION

A primary management tool proposed to reduce impacts of human behavior on the ocean is the marine protected area (MPA). To date, more than 11,300 MPAs encompass 2.12% of the world's oceans, with 0.94% in strongly protected no-take marine reserves (Marine Conservation Institute, 2015). The benefits of MPAs include the protection and rebuilding of commercial fish populations

(Gell and Roberts, 2003), the protection of vulnerable habitats and species (Rodrigues et al., 2004), and the provision of opportunities for tourism, recreation, and education (Ham and Weiler, 2012).

For endangered migratory marine megafauna (such as sea turtles, sharks, and whales), protection is required beyond the existing MPAs. Thus, marine conservationists have been advocating to increase the global coverage of MPAs and to create networks of MPAs (Balmford et al., 2005; IUCN, 2008).

Marine tourism is an emerging recreational activity around the world with the potential to contribute to conservation. Specifically, marine wildlife tourism, defined as any tourist activity with the primary purpose of watching, studying, or enjoying non-consumptive activities with marine wildlife (including diving and snorkeling), has been growing in recent decades (Masters, 1998; Stoeckl et al., 2010). Zeppel (2008) provides a summary of studies that show that marine mammals, sea turtles, seabirds, and sharks are key tourism attractions. Stoeckl et al. (2005) also emphasize the positive economic impact of wildlife tourism related to well-known species on coastal destinations. Some documented examples include watching whales, sea turtles, whale sharks, and dolphins, mainly in Australia and New Zealand (Davis et al., 2000; Hoyt, 2001; Wilson and Tisdell, 2001; Orams, 2003). Other studies have also found that recreational experiences with iconic marine species have contributed to pro-environmental attitudes and post-experience intention to engage in their conservation (Mayes et al., 2004; Zeppel and Muloin, 2008)[1]. For instance, visitor surveys after marine wildlife tours in Australia have shown that visitors are willing to help protect marine endangered species through personal conservation actions (e.g., report poaching or educate others) and through monetary donations for conservation (Tisdell and Wilson, 2001; Mayes et al., 2004).

Despite concerns about the impacts of increased marine tourism in some places (Hall, 2001; Dikou, 2011; Gladstone et al., 2013), the benefits from environmentally-friendly and well-managed tourism initiatives can promote and assist in coastal and marine conservation efforts. Thus, the growth of marine tourism represents a potential win-win for marine conservation and natural resource agencies. That is, the high cost associated with marine protection (Balmford et al., 2003) and limited funding sources (Gravestock et al., 2008) stand out as the main constraints to the creation of new MPAs and protection of existing MPAs. Tourists potentially could provide the resources needed to expand marine protection if resource agencies could design funding mechanisms that actively involve them. For this to happen, however, a better understanding of tourists' motivations, intentions, and behavior toward the support for marine endangered species is needed to design effective funding and conservation initiatives.

Over the past decades, researchers have examined social factors that influence people's interest in conserving a variety of environmental goods, including endangered species, and have advocated taking into account the social context for successful conservation strategies (DeCaro and Stokes, 2008; Choi and Fielding, 2013). Part of this research has involved testing conceptual frameworks that explain the way individuals link their values, beliefs, attitudes, and contextual factors to pro-environmental intentions and behaviors (Ajzen, 1991; Fulton et al., 1996; Stern, 2000).

When analyzing pro-environmental behavior, it is important to distinguish behavioral intentions from actual behavior. The Theory of Planned Behavior (Ajzen, 1991), or TPB, emphasizes the relationship between intention and behavior. Under this theoretical framework, the individual's intentions capture the motivational factors that influence a behavior (Ajzen, 1991). The distinction is particularly relevant in this paper because we focus on examining a person's willingness to donate (WTD) to the conservation of two endangered marine species, which is a stated intention that is a signal of, and precursor to, the actual behavior of contributing money for conservation.

Few studies have investigated the factors influencing pro-environmental intention and behavior to support marine endangered species conservation. Those that have focused on the determinants of preferences and values derived from stated preference economic valuation methods (Kotchen and Reiling, 2000; Aldrich et al., 2007)[2]. The focal point of these studies is on how environmental concern influences the willingness to pay (WTP) for conservation or protection. WTP is a *quantitative* measure of economic value. On the other hand, WTD is a *qualitative* measure of the desire or intent to contribute monetarily. Having a willingness to donate is indicative of, and a precursor to, having a positive WTP. Thus, they are related, but not identical concepts.

Since there are no studies, to our knowledge, that examine the determinants of WTD in a marine conservation context, and WTP is a related concept, we turned to that literature for insights[3,4]. Both Kotchen and Reiling (2000) and Aldrich et al. (2007) determined that environmental concern, as measured by the New Ecological Paradigm[5], has a strong effect on predicting WTP for the conservation of two endangered species, the peregrine falcon and shortnose sturgeon. Tisdell and Wilson

[1]However, few studies have examined whether visitors continue to support or engage in conservation efforts after these types of trips (see Ballantyne et al., 2011 for an exception).

[2]These methods typically involve asking people questions that reveal either directly or indirectly for their preferences or the value they place on a good or service, such as protection of an endangered species (see Lew, 2015).

[3]However, there are several studies that estimate WTP related to the conservation of marine species (e.g., Lew et al., 2010; Boxall et al., 2012; Wallmo and Lew, 2012; Lew, 2015; Wallmo and Lew, 2015) and marine parks (e.g., Peters and Hawkins, 2009) using stated preference methods.

[4]The literature also contains studies that examine factors influencing the intention to carry out environmental friendly activities for the conservation of marine species [e.g., manatees (Aipanjiguly et al., 2003), sea turtle (Kamrowski et al., 2014)]; as well as studies on behavioral intention for topics beyond marine conservation, including conservation of terrestrial threatened and rare fauna (e.g., Jacobson et al., 2003; Perry-Hill et al., 2014), water (e.g.,Yazdanpanah et al., 2014), soil (e.g.,Lynne and Rola, 1988), and energy (Abrahamse and Steg, 2009).

[5]Environmental concern is measured using the New Ecological Paradigm (NEP) scale. The NEP scale measures general environmental concern using responses to 15 likert-scale items (Dunlap et al., 2000). The NEP scale focuses on five core components of environmental concern: limits to economic growth, anti-anthropocentrism, the fragility of nature's balance, human exemptionalism, and the possibility of potential catastrophic environmental changes affecting people.

(2001) explored how socio-demographic factors affect the WTP of tourists visiting Mon Repos Beach near Bundaberg, Queensland, for the purpose of watching sea turtles. The study showed that on-site experiences with marine wildlife, and whether visitors saw sea turtles, significantly influenced their WTP for species protection.

In this paper, we explore factors influencing tourists' WTD for marine species conservation using survey data of tourists in Ecuador visiting the Galapagos National Park (GNP) and its Marine Reserve to gain insights about tourists' motivations, intentions, and behavior that can aid resource managers and decision makers design more effective ways of funding conservation programs. For this, the Galapagos is a useful region to study due to its economic and political significance in the Eastern Tropical Pacific region in terms of tourism related to marine species. It is the largest MPA in the region, is visited by the greatest number of tourists among archipelagos in the region, and has several marine endangered species found there. Additionally, tourism to the Galapagos has increased steadily over the last decade[6], making it a useful case study to explore tourists' intentions to support the recovery of marine endangered species in the region. Here we focus on tourists' WTD toward the conservation of two specific marine endangered species found in the Galapagos: the scalloped hammerhead shark (*Sphyrna lewini*) and the green sea turtle (*Chelonia mydas*). These are iconic migratory species whose protection would require the expansion of MPA networks and thus benefit the conservation of other species in those networks.

We use discrete choice models to analyze what factors influence the stated intention to donate for the recovery of the endangered green sea turtle and scalloped hammerhead shark. The data for the analysis are from a survey conducted in 2013 with Galapagos tourists. The survey included several questions to identify attitudes and beliefs toward species conservation, levels of concern for specific species, recreational motivations and past donation patterns, as well as individual characteristics, such as age, gender, tourist residency (whether the tourist resides in Ecuador and is therefore "domestic," or is from another country and is a "foreigner"), and other demographics.

MATERIALS AND METHODS

Study Setting

The Galapagos archipelago is one of several island groups in the Eastern Tropical Pacific marine region that extends along the Pacific Coast of the Americas, from the southern tip of the Baja California Peninsula in the north to northern Peru in the south. It consists of 13 major islands and over 100 islets and emergent rocks (Snell et al., 1996) and lies in the eastern tropical Pacific 1000 km from the coast of continental Ecuador. The Galapagos Marine Reserve (GMR) covers an area of approximately 138,000 km^2 (**Figure 1**).

Approximately 12% of the marine species in the Eastern Tropical Pacific are threatened with extinction due to overfishing, habitat loss, and changing climatic conditions (Polidoro et al., 2012). Of these threatened species, highly migratory marine species like green sea turtles and scalloped hammerhead sharks are of great concern in the region. Several studies suggest population declines for both species (Seminoff, 2004; Baum et al., 2007). Scalloped hammerhead sharks are facing increasing fishing pressure outside protected adult aggregation sites (Cocos Island in Costa Rica and the Galapagos Islands in Ecuador) and along the slopes of the continental shelf where catch rates of juveniles are high (Baum et al., 2007). Green sea turtles in the region are mainly threatened by coastal development, collection of eggs for consumption, and fisheries bycatch (Seminoff, 2004). As a result of population declines and continued threats, the Eastern Tropical Pacific populations of both species have been listed as Endangered by the International Union for Conservation for Nature (IUCN) on its "Red List" of endangered and threatened species[7] since 2004 (green sea turtle) and 2007 (scalloped hammerhead shark).

Survey

Data for the analysis were obtained from a survey of tourists visiting the GNP. There were two versions of the survey: one presented information and asked questions about the green sea turtle (TURTLE version) and the other presented information and questions about the scalloped hammerhead shark (SHARK version). The surveys were developed with input received through 8 focus groups and 12 cognitive interviews held in 2012, which involved 44 tourists and 12 tourist managers. Focus groups and interviews aided in refining the content and presentation of the information provided in the survey, as well as the survey questions themselves.

In the final survey instrument, respondents were presented with information about the IUCN Red List that lists and categorizes endangered species. The main goal was to determine how familiar respondents were with the concepts and mechanisms to define and list endangered species both in their home countries and internationally. Detailed information on the marine endangered species of interest, the green sea turtle or the scalloped hammerhead shark, was then introduced to respondents. This included information about the species' biology, feeding, and breeding behavior; habitat and distribution; and threats, current protection actions, and current extinction risk level and status of its population.

After reading this information, respondents were asked to indicate whether they knew about the different aspects of the species, their level of concern about the future status of the species, and their opinions about potential recovery programs. The surveys also collected personal information from respondents about their recreational motivations to see marine wildlife during their visit to the Galapagos; past donation

[6]The number of tourists to the Galapagos has increased at an average rate of 3.7% per year between 2007 and 2014 (Observatorio de Turismo de Galapagos (OTG), 2015).

[7]The IUCN, the International Union for Conservation of Nature, maintains a Red List, which is a comprehensive and objective list of plant and animal species that are at risk of extinction. Risk of extinction refers to the probability of a species becoming extinct in the future. Significant declines in population size and loss of habitat increase the risk of extinction. For both populations in this article, there is at least a 30% risk of becoming extinct in 60–80 years under current conditions.

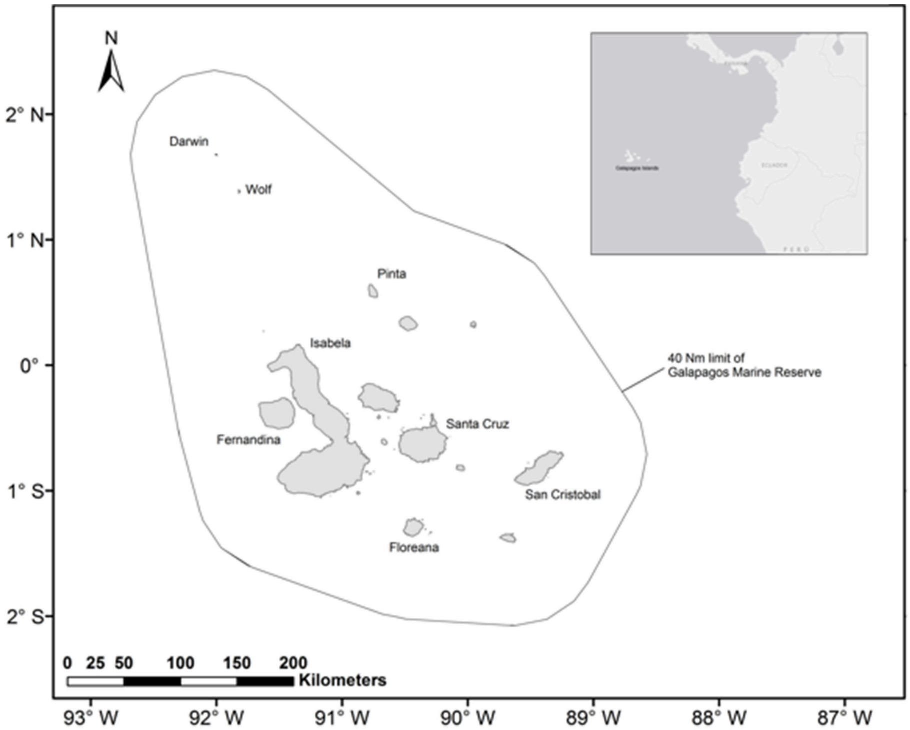

FIGURE 1 | Map of the Galapagos Islands and its marine reserve.

behavior for conservation and non-environmental issues; and socio-economic and demographic information[8].

The survey also presented a hypothetical, yet plausible, conservation scenario. It described a marine conservation program that would create new MPAs along the coasts of Costa Rica, Panama, Colombia, and Ecuador that would provide direct support for additional protection measures for the green sea turtle (in the TURTLE version) or the scalloped hammerhead shark (in the SHARK version). This new conservation program would complement existing offshore MPAs in the region, including the GNP and Marine Reserve. Due to the migratory nature of the endangered marine species considered here, extending protection to key nursery and feeding coastal areas would reduce their threat status (reduce the risk of extinction and lead to improvement in the IUCN status). An independent non-profit organization with representatives from participating governments and other local institutions in the region would be in charge of overseeing the funds raised through donations.

Respondents were then asked whether they would be willing to donate money to the new conservation program. The WTD response measures the respondents' behavioral intention to donate toward the conservation program and is the focal point of our analysis[9]. Respondents were asked to choose between three possible response alternatives (WTD responses): "no," "yes," and "do not know."[10]

Survey Implementation

Before the final survey was implemented, a formal pretest was conducted in 2012 to evaluate and test the survey administration procedures. Subsequently, the final survey was administered[11] in 2013 to a systematic random sample of tourists leaving the islands from the two airports[12] serving the Galapagos, Baltra, and San Cristobal. We surveyed during two main periods, March–April and July–August, to account for temporal variations in visitation. The survey was a self-administered intercept survey, where randomly selected tourists were intercepted and asked to fill out the survey on their own and return it to the interceptor upon completion.

The Analytic Approach

We focused our analysis on modeling respondents' WTD toward the recovery of the green sea turtle and scalloped hammerhead

[8]Many of these variables are described in more detail in The Analytic Approach section.

[9]The specific wording in the survey was: "Would you donate money to programs that improve the status of the threatened scalloped hammerhead shark?" (or green sea turtle depending on version).

[10]The "don't know" alternative was included after numerous focus group participants expressed that they might be willing to donate and support in the future but they cannot be certain about it at the time of the survey.

[11]In December 2012, the Institutional Review Board Administration (IRB) from the University California, Davis approved survey materials.

[12]Tourist operations have visitors enter and depart the Galapagos through either the Baltra airport or San Cristobal airport. Surveys were implemented at both airports and during different times of the year to minimize coverage bias and to optimize (with a limited budget) the chances of obtaining a sample with representation from tourists from the Northern Hemisphere and Southern Hemisphere who visit the Galapagos at different times of the year.

shark. To this end, we use random utility maximization (RUM) based discrete choice models to analyze responses that indicate respondents' intention to donate to the conservation scenario described in the survey. In the RUM approach, when faced with J alternatives, respondent i chooses the alternative that yields the largest utility from among the set of J alternatives (in this case $J = 3$: "yes," "no," "do not know"). The utility of the jth alternative (U_{ij}) is composed of an observable deterministic component (V_{ij}) and a stochastic component (ε_{ij}) that is known to the individual, but not the researcher (McFadden, 1974). Thus, we can model the probability that respondent i chooses the jth alternative as:

$$\Pr_i \left[choose\ j\right] = \Pr\left(U_{ij} \geq U_{ik}, \forall j \neq k\right) \tag{1}$$

Following the common assumption that errors are independently and identically Gumbel distributed, we get the familiar multinomial logit model (McFadden, 1974), with corresponding probabilities of the form (Greene, 2011):

$$\Pr_i \left[choose\ j\right] = \frac{\exp\left(V_{ij}\right)}{\sum_{k=1}^{J} \exp\left(V_{ik}\right)}, \forall j, k \in J \tag{2}$$

In this application, V_{ij} is assumed to be a linear additive function of the independent explanatory variables[13] characterizing respondent i's utility. Two main socio-psychological theoretical frameworks informed the selection of variables for the utility specification. The first is the Theory of Planned Behavior or TPB (Ajzen, 1991), which postulates that the intention to perform a certain behavior is the main predictor of that behavior. According to TPB, behavioral intention is determined by attitudes toward the behavior (positive or negative evaluation of performing the behavior), subjective norms (perception of social pressure from reference groups to perform the behavior), and perceived behavioral control (perceived ease or difficulty of performing the behavior). The other relevant conceptual framework is the Value-belief-norm (VPN) theory developed by Stern and colleagues (Stern et al., 1999; Stern, 2000), which postulates that pro-environmental behavior is determined by five factors: (a) general personal values (e.g., altruistic, egoistic); (b) ecological worldview[14]; (c) personal beliefs on adverse consequences for valued objects, (d) personal beliefs on perceived ability to reduce threat; and (e) personal norms for pro-environmental action. The VPN model explicitly accounts for beliefs about the consequences of human-environment relationships and how the individual can actually reduce threats. These particular beliefs could be shaped by information and findings from science.

These two theoretical frameworks suggest survey questions related to environmental attitudes and personal beliefs should help explain WTD. Both attitudes and beliefs are the core elements that will influence the intention to perform a behavior according to either the TPB or VPN. Environmental attitudes have been defined as a "psychological tendency expressed by evaluating the natural environment with some degree of favor or disfavor" (Milfont and Duckitt, 2010, p. 80). Environmental attitudes are usually represented by environmental concern[15] ($\boldsymbol{X}^{attitudes}$) and for this application we identify three measures: one describing how important protecting endangered species is in general; the level of concern about the specific marine endangered species in the survey; and the level of concern about the effectiveness of the conservation program. We also include two types of personal beliefs ($\boldsymbol{X}^{beliefs}$): a norm belief and a control belief. In general, norm beliefs are indicators of how the individual's behavior is influenced by *what should I do* or by *what others think I should do* (Schwartz, 1977; Ajzen, 1991). In the survey, tourists were asked to indicate their level of agreement to the norm belief that protection for the marine endangered species should be paid only by residents of the region. The control belief was framed according to the VPN theory (Stern et al., 1999) as a perceived ability to reduce an environmental threat; in this case, the extinction of an endangered species and respondents were asked to indicate their level of agreement with donating for the protection of the species even though it is threatened and may still become extinct in the future.

Besides psychological factors, Stern (2000) argues that studies to understand predisposition to behavior often overlook important factors, specifically personal characteristics (e.g., personal habits, interest for, and impact of experiences) and context-related factors (specific features of the environment where the behavior will take place; e.g., incentives and available information). To capture these individual-specific factors we include the following explanatory variables: previous knowledge on endangered species ($\boldsymbol{X}^{knowledge}$), past donation behavior ($\boldsymbol{X}^{donation}$), and personal motivations[16] to see marine species ($\boldsymbol{X}^{motivation}$), all of which are relevant to the conservation scenario presented. Respondents were asked about their knowledge of endangered species ($\boldsymbol{X}^{knowledge}$) at two scales: (a) general familiarity with the listing process and with regulations on endangered species in tourists' home countries; and (b) specific knowledge about the ecology, threats, and protection measures taken to protect the specific marine endangered species in the survey. In addition, several variables were included in the model to assess each respondent's experience with donating money ($\boldsymbol{X}^{donation}$), specifically whether the respondent had donated time or money in the past to a conservation organization, had donated money to a marine conservation program specifically, and had donated money to specific causes in the last 5 years. The specific causes included poverty, education, environment, and the arts. Finally, tourists were also asked about the importance of seeing marine animals as a motivation for visiting the Galapagos Islands ($\boldsymbol{X}^{motivation}$). All tourists were presented with four groups of marine species to rate their motivation to see them during the trip: sharks, sea turtles, sea lions, and marine iguanas (all of these

[13]Groups of explanatory variables are represented by X vectors in the text that follows.

[14]Environmental worldviews are commonly measured by the NEP Scale (described in earlier note).

[15] Environmental concern is a broad term that refers to beliefs and attitudes related to the seriousness and importance of environmental issues and commonly used to measure attitudes toward environment and conservation (Dunlap and Jones, 2002; Milfont and Duckitt, 2010).

[16]Several empirical studies assessing factors influencing predisposition to pro-environmental behavior in tourists have confirmed the significant influence of visitors' recreational interest toward the environment (Kerstetter et al., 2004; Thapa, 2010; Kil et al., 2014).

species groups consist of at least some species listed under the IUCN Red List).

Socio-economic and demographic data were also included as control variables ($X^{demographics}$). Likelihood ratio tests suggested that education, level of employment, region of residence, and whether the respondent was a retiree or not were not statistically significant and were thus excluded from the final models[17]. Household income, gender, age, and a binary variable for whether the tourist is from Ecuador or not (origin) were included as explanatory variables. Thus, the utility function for the *i*th individual and *j*th choice alternative was specified as:

$$V_{ij} = \alpha_j + \beta_j X_i^{attitudes} + \delta_j X_i^{beliefs} + \phi_j X_i^{knowledge} + \lambda_j X_i^{donation} + \gamma_j X_i^{motivation} + \varphi_j X_i^{demographics} \quad (3)$$

where, α is a scalar parameter (intercept), and β, δ, Φ, λ, γ, and φ are unknown coefficient vectors that are specific to the associated response ("no," "yes," and "do not know"); that is, there is a separate set of parameter vectors for each response. Identical explanatory variables are included in both the TURTLE and SHARK models.

We estimate separate multinomial logit models for each of the two survey versions (TURTLE and SHARK) using maximum likelihood estimation in STATA 14.0. A pooled version that combines data from the SHARK and TURTLE versions was also estimated with a dummy variable to identify whether or not WTD is affected by the version of the survey, which is a proxy for the effect due to the species[18].

RESULTS

Survey—Descriptive Statistics

The survey achieved an overall cooperation rate of 94% across the two main survey versions[19]. The total number of complete and valid[20] surveys used for the analysis was 701 (367 SHARK and 334 TURTLE surveys)[21]. Across the samples, the mean age of respondents was 44 years, and 42% of respondents were male (see descriptive statistics in **Table 1**). Approximately 63% were foreigners (reside outside Ecuador), and 37% came from mainland Ecuador. Respondents from North America, particularly the United States, accounted for 28% of all respondents. European respondents accounted for another 20%, while only 5% came from Asia, Africa, or Oceania. Eighty percent of the respondents had at least a 4-year university degree or higher, and more than 60% indicated having full-time employment. Across all respondents, mean household income was $73,000 USD with a standard deviation of $64,800 USD. There was a considerable difference in household income levels between foreign (mean annual income of $101,400 USD) and domestic (mean annual income of $21,800 USD[22]) respondents in the sample. The numbers suggest a common profile of tourists visiting the GNP: well-educated and higher income individuals. Student's *t*-tests confirm that the samples for each survey version (SHARK and TURTLE versions) were not statistically significantly different across the demographic characteristics.

TABLE 1 | Socio-demographic variables.

Socioeconomic variable	Turtle (N = 334)	Shark (N = 367)	T-test statistic
GENDER			−1.16
Female (%)	61	56	
EDUCATION CATEGORY			−0.34
High school (%)	7	11	
Some university (%)	12	11	
Undergraduate degree (%)	37	33	
Graduate work/degree (%)	44	45	
EMPLOYMENT CATEGORY			−0.47
Full-time employed (%)	61	60	
Part-time employed (%)	11	11	
Student (%)	5	5	
Retired (%)	12	12	
Unemployed/Unpaid (%)	4	6	
REGION OF ORIGIN			1.29
Asia/Africa (%)	2	2	
Europe (%)	21	17	
Latin America (%)	46	49	
North America (%)	28	29	
Oceania (%)	4	3	
ORIGIN GENERAL			1.44
Domestic (%)	33	38	
Foreigner (%)	67	62	
AGE CATEGORY			0.44
Median Age	41.5	44.0	
Mean Age	43.6	43.9	
INCOME (2012 $US DOLLARS)			−1.21
Median Income	60.0	47.2	
Mean Income	75.6	70.7	

[17]Although we expected that education and level of employment were predictors for WTD, their low statistical significance could be explained by their correlation with income or by a low variation across the sample if we consider the average profile of Galapagos tourists (**Table 2**). A binary variable for "Retired," as a specific level of an employment characteristic of some Galapagos tourists, was shown to be insignificant as well. These results might indicate that income captured most of the explained variation. In addition, variables related to whether the tourist actually saw the endangered marine species in the survey as part of their most recent trip to Galapagos were also found not to be statistically significant.

[18]However, a likelihood ratio test indicated that the data should be estimated separately instead of pooled (test statistic was 77.62, $p < 0.001$).

[19]Cooperation rates are calculated as the number of completed surveys divided by the number of tourists intercepted and asked to participate. Separate cooperation rates for each version were 94.5% (TURTLE version) and 93.5% (SHARK version).

[20]Valid surveys were those that were not missing observations to key variables for the analysis, and were not identified as "protest" respondents based on their responses to follow-up questions and open-ended comments.

[21]The margin of error for the two samples (and a binary response) are 5.36% (TURTLE) and 5.11% (SHARK) considering a tourist population of 204,000 for 2013 (official statistic) and a confidence level of 95%.

[22]According to official statistics from the Ecuadorian Institute of Censuses and Statistics, average annual household income in 2011 was approximately $9000 and has not increased significantly during the last 3 years. This figure shows that domestic tourists visiting the Galapagos have income levels that are much higher than the average income level in the country.

TABLE 2 | Factors influencing WTD—descriptive statistics.

	Sea turtle (n = 334)			Shark (n = 367)			Mann-Whitney test[a]
	Median	Mean	Std Dev	Median	Mean	Std Dev	
ATTITUDES (1-not at all to 5-extremely important or concerned)							
Protecting endangered species is important to me	5	4.67	0.51	5	4.70	0.51	0.854
Concerned for the endangered species in the survey	4	4.24	0.71	4	4.08	0.80	−2.417**
Concerned about effectiveness of the conservation program	4	3.60	1.02	4	3.65	1.05	−0.825
BELIEFS (1-strongly disagree to 5-strongly agree)							
Norm: Protection should be paid by residents only	2	2.14	1.12	2	2.21	1.16	0.761
Control: I do not want to donate for protection because the species will become extinct anyway	2	2.00	1.09	2	2.14	1.09	2.04**
KNOWLEDGE ABOUT ENDANGERED SPECIES (0 = no; 1 = yes)							
IUCN Red List categories for endangered species	0	0.31	0.46	0	0.28	0.45	0.799
Laws and regulations on endangered species in home country	1	0.72	0.45	1	0.73	0.45	0.176
KNOWLEDGE ABOUT SPECIES IN THE SURVEY (0 = no; 1 = yes)							
General facts and ecology of the species	1	0.69	0.46	1	0.56	0.50	3.697**
Threats to the populations of the species	1	0.76	0.43	1	0.81	0.39	1.663*
Protection measures to protect the species	0	0.44	0.50	0	0.19	0.40	−7.041**
Marine conservation programs to protect the species	0	0.11	0.31	0	0.08	0.27	−1.178
MOTIVATION TO SEE SPECIES (1-not at all to 5-extremely important)							
Importance to see sharks	4	3.62	1.15	4	3.59	1.21	0.935
Importance to sea turtles	4	4.23	0.76	4	4.35	0.73	2.127**
Importance to sea lions	4	4.17	0.87	4	4.29	0.82	1.830*
Importance to marine iguanas	4	4.17	0.84	4	4.30	0.81	2.209**
OBSERVATION OF SPECIES (0 = no; 1 = yes)							
Has observed the group of species (e.g. sea turtles/sharks in general)	1	0.82	0.38	1	0.76	0.42	−2.132**
Has observed the endangered species during trip to Galapagos	1	0.75	0.43	0	0.15	0.36	−16.025***
PAST DONATION BEHAVIOR (0 = no; 1 = yes)							
Has donated money to a marine conservation program	0	0.19	0.39	0	0.24	0.43	1.544
Has donated time/money to a conservation organization	0	0.43	0.50	0	0.41	0.49	−0.367

[a]*Reports the significance of the Mann-Whitney test; statistically significant differences between distributions are indicated at the 1% (***), 5% (**), and 10% (*) levels.*

Across the sample, only about 30% of survey respondents had heard of the global IUCN Red List, but 72% indicated that they were familiar with laws and regulations pertaining to endangered species in their home countries (**Table 2**). Survey respondents to both versions of the survey also indicated that protecting endangered species is important to them (mean of 4.68 on a 5-point Likert scale, where 1 represents not at all important and 5 extremely important). For both survey versions combined, the majority of respondents had heard about the species presented in the survey (62%) and about the natural and human-related threats they face (79%). The results show that statistically more respondents know about protection efforts for the green sea turtle compared to those for the scalloped hammerhead shark (44% compared to 19% for TURTLE and SHARK, respectively). On average, survey respondents were "very concerned" about the species' future status given the information provided about each species in the survey. The mean concern level for the endangered sea turtle (4.24) is statistically higher than that for the scalloped hammerhead shark (4.08) at the 5% level of significance.

The majority of respondents indicated they felt it was at least a "very important" (4 on a 5-point scale from 1 = not at all important to 5 = extremely important) motivation for their Galapagos trip for them to see sharks (59%), sea turtles (87%), sea lions (83%), and marine iguanas (84%). In terms of actually observing the species of interest in the surveys during their trip, more than 70% had observed the green sea turtle as part of the trip to the Galapagos, compared to only 17% who had observed scalloped hammerhead sharks[23].

WTD Modeling Results

For the SHARK sample, 25% responded "no" to the question asking whether they would be willing to donate to the conservation program, while 34% said "yes" and 38% stated they "do not know" in the question. Nineteen percent of the TURTLE sample responded "no," while 43% said "yes" and 38% indicated they "do not know" whether they would donate toward the program.

[23]As one reviewer noted, reported sightings of the green sea turtle may be inaccurate given the potential for respondents confusing the green turtles with other sea turtle species. However, the green sea turtle is the most common sea turtle seen in the Galapagos. The survey provides information and pictures of the species, which should also have aided in answering this question more accurately.

The "no" response was selected as the base alternative for both the TURTLE and SHARK models, resulting in parameters being estimated for the utility functions associated with the other two response functions "yes" and "do not know" (**Table 3**). The likelihood ratio index, a pseudo-R^2 measure of goodness-of-fit (Maddala, 1983), was 0.26 for the TURTLE model and 0.21 for the SHARK model. These LRI values suggest that both models are statistically significant (i.e., the parameters are jointly significant).

Except for the belief variables[24], positive coefficients indicate that higher levels of the variable lead to a higher probability of answering "yes" or "do not know" to the intention to donate toward the recovery of the species. Conversely, negative coefficients suggest that the explanatory variable reduces the likelihood of a "yes" or "do not know" response.

Factors that Influence a "Yes" Response on WTD

Psychological Factors

We find that environmental attitudes are statistically significant and influence WTD only when they measure the concern for the specific marine endangered species in the survey. Consistent with our expectations, respondents who are more concerned about the endangered species are more likely to donate toward its recovery, all else being equal (**Table 3**). Results also indicate that attitudes toward protecting endangered species in general do not influence the WTD for either endangered species. In addition, respondents' attitudes toward the conservation program, specifically the level of concern about its effectiveness, only influence the probability of being willing to donate in the SHARK model but not in the TURTLE model.

The estimated coefficients representing personal beliefs are negative and statistically significant for both survey versions. The more agreement with the norm belief that only residents should pay for protection of both marine endangered species, the lower the probability of a positive intention to donate for conserving the species. In other words, respondents who believe the species' protection should not be the sole responsibility of residents are more likely to donate. Moreover, respondents who believe that their donations can reduce the risk of extinction of the endangered species are more likely to be willing to donate in the future.

Seeing, or feeling it was important to see, marine endangered species in the Galapagos influenced intentions to support the marine conservation program depending on the species in the survey. Although survey respondents reported a high level of interest to see sea turtles while in the Galapagos (**Table 2**), this motivation does not appear to influence their decision to support recovery programs, as the parameter on "importance to see sea turtles" was not statistically different from zero (**Table 3**). Moreover, likelihood ratio tests failed to reject the null hypothesis that the parameters representing recreational motivations are jointly zero for the turtle model. In contrast, for the endangered scalloped hammerhead shark, the more importance respondents placed on seeing sharks, the higher the probability of answering "yes" when they were asked for their WTD for shark conservation. This is consistent with our prior that tourists who are willing to donate toward the recovery of endangered sharks are those who have a particular interest in the species; specifically, divers whose primary motivation is diving with schools of sharks in the archipelago.

Socio-Demographic Variables

The results of a likelihood ratio test suggest there is a statistically significant joint effect of socio-demographic variables on WTD for the recovery of the two marine endangered species (**Table 4**). However, the individual statistical significance of individual variables differs between the two models. Whether a tourist is from Ecuador or not (origin) has a statistically significant influence on the probability to donate for sea turtle conservation, but not for shark conservation (**Table 3**). Our results thus indicate that domestic (Ecuadorian) respondents have a significantly higher probability of being willing to donate to the recovery of sea turtles than foreigners, all else being equal. Household income[25], by contrast, appears to influence the intention to donate only for the recovery of the endangered hammerhead shark. Contrary to expectations and other studies involving endangered species (Aldrich et al., 2007; Choi and Fielding, 2013), the income effect on the probability to donate for shark conservation is negative. Thus, respondents with higher income levels were less likely to donate toward the recovery of the endangered hammerhead shark (**Table 3**); and this relationship is similar for domestic and international visitors. Other socio-demographic variables, including gender and age, do not influence the probability to donate toward the protection of either endangered species[26].

Other Individual-Specific Variables

Our results show that factors other than psychological and socio-economic characteristics of tourists in Galapagos have a statistically significant effect on WTD. Past donation behavior is a determinant of the intent to donate to the conservation of both marine endangered species (**Table 3**). The joint significance of all past donation-related variables is high (at the 1% level) for both the TURTLE and SHARK models. The results indicate that respondents who have specifically donated to causes related to environmental and animal welfare in the last 5 years are more likely to be willing to financially support the recovery of the endangered green sea turtle and the scalloped hammerhead shark, which is consistent with our *a priori* expectations. Interestingly, although more than 40% of the respondents visiting the Galapagos have participated or been a member of a conservation organization, this does not seem to influence their intention to donate specifically for marine endangered species conservation. Past donations specifically to marine conservation

[24] Due to the wording of these questions, the coefficients have interpretations that are different from those of other explanatory variables. Please refer to the section on "Psychological drivers" below for a detailed explanation.

[25] Multiple income variables, including interactions of income with region of residency and nationality, were used when testing model specification. However, these variables led to similar qualitative results. Likewise, interaction variables between income and nationality did not yield statistically significant results.

[26] Several model specifications were initially tried that included variables to account for effects due to tourists' region of residency (e.g., Europe, North America, or other), but these effects did not seem to be statistically significant.

TABLE 3 | Multinomial logit results.

EXPLANATORY VARIABLE	Model: Sea turtle		Model: Shark	
	Yes	Do not know	Yes	Do not know
Constant	−7.323**	−6.036**	2.127	1.535
ATTITUDES ($X^{attitudes}$)				
Protecting endangered species is important to me	0.186	0.481	0.063	0.022
Concerned about the endangered species in the study	0.981***	0.724***	0.534**	0.014
Concerned about effectiveness of conservation program	0.195	−0.085	*0.298***	0.339**
BELIEFS ($X^{beliefs}$)				
Norm:protection should be paid by residents only	−0.384**	0.050	−0.334**	−0.077
Control: I do not want to donate for protection because the species will become extinct anyway	−0.959***	−0.493***	−0.685***	−0.447***
MOTIVATION TO SEE MARINE SPECIES ($X^{motivation}$)				
Importance to see sharks	−0.296	−0.160	0.622***	0.030
Importance of sea turtles	0.299	0.151	−0.166	−0.015
Importance of sea lions	0.337	−0.080	−0.132	0.199
Importance of Marine iguanas	0.362	0.072	0.155	0.280
KNOWLEDGE ($X^{knowledge}$)				
IUCN Red List Categories	0.332	−0.065	0.167	0.272
Regulation on endangered species	0.215	0.532	−0.956**	−0.322
General facts and ecology species	−0.443	−0.388	−0.560	0.043
Threats to the species	0.014	0.216	0.600	0.000
Protection measures	−0.519	0.006	0.391	0.448
Marine conservation program	−0.206	−0.258	0.166	−0.316
PAST DONATION BEHAVIOR ($X^{donation}$)				
Has donated time/money to a conservation organization	−0.168	0.455	*0.952**	0.017
Has donated money to a marine conservation program	1.495**	1.801***	0.638	0.770
In the past 5 years, has donated for:				
Poverty	0.497	0.414	*−0.273*	−0.560*
Religion	−0.149	0.238	0.380	0.120
Education	0.478	−0.033	−0.352	−0.313
Environment/animal welfare	2.186***	1.602***	1.743***	0.973**
Health/ medical research	0.496	0.250	0.250	−0.069
Arts and culture	−0.399	−0.538	−1.276**	−0.348
Peace and Human rights	0.072	−0.694	0.445	0.105
Disaster relief	0.135	−0.118	0.376	0.061
DEMOGRAPHICS ($X^{demographics}$)				
Female	0.313	0.027	−0.158	0.260
Age	0.153	0.007	0.028	−0.007
Age-squared	−0.002	0.000	0.000	0.000
Origin: Ecuador	2.324***	1.614***	0.704	−0.635
Income	−0.253	0.166	−0.696***	−0.348**
N		334		367
MODEL FIT STATISTICS				
Log likelihood		−259.31		−315.42
LRI		0.26		0.21
AIC		642.62		754.84
BIC		878.91		996.97

*Statistical significance of parameters: *, statistically different from zero at the 10% level; **, statistically different from zero at the 5% level; ***, statistically different from zero at the 1% level.*

TABLE 4 | Joint significance of the explanatory variables of the multinomial logit model.

	Likelihood ratio test statistic*,**	
	Sea turtle model	Shark model
All parameters for attitudes and beliefs are zero	71.46***	47.02***
All parameters for motivations are zero	12.56	24.12***
All parameters for past donation behavior are zero	49.04***	50.04***
All parameters for knowledge are zero	8.42	15.37
All parameters for demographics are zero	33.35***	37.54***

Parameters are jointly different from zero at the 10% level (*), at the 5% level (**), at the 1% level (***).

programs significantly influence WTD only for the TURTLE model. Thus, the main driver of WTD is the actual past donation behavior, and particularly past donations related to the environment and its goods and services.

In contrast to personal donation habits, prior knowledge about endangered species does not significantly affect WTD. This is true for both levels of knowledge assessed in the study: general knowledge about listed IUCN categories and specific knowledge about the endangered green sea turtle and hammerhead shark. Likelihood ratio tests could not reject the null hypothesis that prior knowledge variables are jointly zero. The only exception to this result is in the SHARK model, where prior knowledge about regulations on endangered species (generally) in their home country has a statistically significant and negative influence on respondents' WTD.

Species Effects

In comparing preferences between the endangered sea turtle and hammerhead shark models, we find evidence of a species effect related to the green sea turtle. The pooled version of the model combines data from the TURTLE and SHARK versions and allows us to assess whether there is a difference between WTD between the versions (Appendix Table 1). Note that although the species effect parameter will capture all the differences between the survey versions, the two versions were identical except in specific information on each species; therefore, other differences between versions are expected to be negligible. The coefficient on a dummy variable for the green sea turtle version in the pooled model is positive and significant (coefficient 0.911, $p = 0.001$), which suggests a relative preference toward the green sea turtle over the scalloped hammerhead shark in terms of WTD.

Comparing the "Yes" and the "Do Not Know" Response Functions

Our findings suggest that factors that influence WTD are similar between respondents who are willing to donate ("yes") and respondents who are uncertain ("do not know") if they will donate toward the recovery of the endangered sea turtle. That is, there are qualitative similarities between parameters associated with the "Yes" and "Do not know" responses for each species model (**Table 3**). Statistical significance levels of the coefficients of the "yes" and "do not know" response functions for the TURTLE model are similar. Some differences do exist, however, between these estimated functions for the SHARK model. For instance, the effects of the explanatory variables, level of concern about the endangered species, the norm belief about "only residents should pay for protection," and knowledge about regulations on endangered species, are statistically significant only for those respondents who answer positively to the WTD question.

DISCUSSION

In this study, we investigated factors that influence tourists' intentions to donate toward the recovery of two marine endangered species in the GNP. Our results suggest that environmental attitudes, personal beliefs, and past donation behavior affect tourists' stated intentions. Consistent with the TPB (Ajzen, 1991) and VPN (Stern et al., 1999) socio-psychological theoretical frameworks, we found that specific attitudes and beliefs toward the environmental good (in this case marine endangered species) matter. Tourists who are more concerned about the extinction of the two marine endangered species in the future are more likely to be willing to donate toward their recovery. Moreover, the stronger the personal beliefs about the shared responsibility of protecting the species and that actions to protect the species should be done, the higher the probability of tourists' intention to donate for the conservation of these species.

The estimated effect socio-demographic factors had on WTD did not meet our a priori expectations. Contrary to other studies involving endangered species (Aldrich et al., 2007; Choi and Fielding, 2013), the income effect on the probability to donate for shark conservation is statistically significant but negative, meaning those who are more wealthy are less likely to donate. A possible explanation for this result is that tourists interested in supporting shark conservation are part of a small group of visitors with specific recreational interests and motivations for this particular group of marine species. Moreover, the majority of tourists, and those with higher income profiles, might not be interested in shark conservation specifically, but rather have broader conservation interests that would drive the model results. In the green sea turtle's case, the statistical insignificance of the income variable might be caused by the correlation between household income and origin of tourists in the sample[27]. Additionally, the parameter on tourist origin may be picking up whatever effect income has on WTD. Considering that tourists visiting the Galapagos are in general wealthier than average individuals, the low variation in (higher) income levels across the sample might explain the low statistical significance of household income.

This study confirms the importance of individual-specific explanatory variables emphasized in modern models of pro-environmental behavior (Stern, 2000), at least in this application. Specifically, we found that past donation behavior is a significant factor that positively influences WTD. The results of the

[27] Correlation coefficients between household income and tourist origin are −0.63 for the TURTLE version and −0.67 for the SHARK version.

behavioral model suggest that tourists who have donated in the past to causes related to the environment, animal welfare, and marine conservation are more likely to be willing to donate to conservation programs for the green sea turtle and the hammerhead shark, all else being equal. Surprisingly, neither knowledge about endangered species nor general recreational motivations to see marine species during their visit influences WTD in this study. This was contrary to our expectations and to modern psychological models (Stern, 2000), which emphasize the influence of context-dependent variables on the predisposition to perform a pro-environmental behavior. Nevertheless, the statistically insignificant role of knowledge about the marine endangered species is similar to results of previous empirical studies on endangered species (Kotchen and Reiling, 2000; Aldrich et al., 2007). The one exception to this finding suggests that tourists who are more informed about endangered species regulations in their home countries tend to be less willing to donate money toward the protection of the scalloped hammerhead shark in the Galapagos and Eastern Tropical Pacific. This knowledge does not have the same effect on WTD for the green sea turtle recovery. Together, the discussion above suggests that personal factors, such as specific past donation behavior and specific knowledge about laws or regulations to protect endangered species, seem to affect tourists' WTD to the conservation of the two species.

In addition to the several psychological, socio-demographic, and personal factors influencing stated intentions, we found evidence of a species effect on WTD[28]. This "species effect" suggests that tourist visiting the Galapagos and the Eastern Tropical Pacific have a stronger preference to donate to the recovery of the green sea turtle. One potential explanation of the species effect is the differential perception tourists may have of these species: sea turtles may be viewed as more charismatic and friendly sea animals, while sharks may be viewed as scary and dangerous. Additionally, tourists who are knowledgeable about or wish to see the scalloped hammerhead shark tend to be more willing to donate toward protection of this species. However, those factors do not appear to influence WTD toward protection of the green sea turtle. In combination, these things suggest that tourists vary in their personal preferences toward each marine endangered species and their protection. Consequently, it is important to recognize these differing preferences when assessing intended or actual behavior toward their conservation. Indeed, previous studies on U.S. endangered species have suggested that the charismatic nature of a species influences the amount spent on its protection (Metrick and Weitzman, 1996) and on people's WTP for protection efforts (Richardson and Loomis, 2009).

This study represents one of the few studies to investigate the factors influencing WTD for the recovery and protection of marine endangered species. It supports previous empirical evidence about the influence of environmental attitudes on a related concept, WTP for endangered species conservation (Kotchen and Reiling, 2000; Aldrich et al., 2007; Spash et al., 2009; Choi and Fielding, 2013). Besides confirming a significant relationship between attitudes and intention-to-donate for the recovery of two marine endangered species, the current study contributes to the empirical literature by evaluating other personal behavioral and context-dependent factors.

However, there are some limitations of the study. First, we limited the analysis to two marine endangered species and to a specific targeted population of these species, the Eastern Tropical Pacific populations. Thus, our results may not be generalizable to other species or to other populations in different marine regions. In addition, our modeling approach assumes that respondents were considering only one of two marine endangered species, either the green sea turtle or the scalloped hammerhead shark, when they answered the WTD question. This, however, may not be true for some respondents who might be linking the conservation program to other marine species. In this case, WTD responses might be based on more than an individual's concern for the species in question (this is commonly referred to as an embedding effect)[29]. However, we leave an investigation into this potential source of bias for future research. Geographically, the study focused only on Galapagos tourists. As such, to the extent visitors to other islands in the Eastern Tropical Pacific differ from Galapagos tourists in terms of their willingness to donate for species conservation, the results may not be generalizable beyond the targeted population. In addition, the study surveyed tourists about their intention to donate immediately after they finished their visit to Galapagos, which may potentially bias their answers toward a future donation behavior. However, recent studies have shown a smaller than expected positive long-term pro-environmental behavior after wildlife-watching trips. For instance, Ballantyne et al. (2011) found out that only 7% of visitors reported adopting a pro-environmental behavior as a result of a whale- and sea turtle-watching visit when surveyed 4 months after the visit. Therefore, our results may represent an upper bound on tourists' intention to donate. Finally, we note that this study focused on analyzing factors affecting tourists' behavioral intention to donate, not on how much they would be willing to donate (their willingness to pay), which is left for future research[30].

From a policy perspective, the current study highlights the potential application of behavioral results to efforts to fund conservation of marine endangered species in the Eastern Tropical Pacific marine region. Both the endangered green sea turtle and the scalloped hammerhead shark are considered "umbrella species" in conservation efforts—increasing protection of their expansive distribution and range will benefit other species (Roberge and Angelstam, 2004). Moreover, MPAs, and in particular the GNP, are important tourist attractions.

[28]Admittedly, the species effect is measured with a parameter that captures all differences between the survey versions. Therefore, it is possible that the species effect may embed other tourist preference differences affected by other differences in the surveys. However, given the two survey versions were constructed to be identical except for the species-specific information, the likelihood of this occurring is small.

[29]In our context, an embedding effect occurs when an individual's response to the WTD question is based on an assumption made by the individual that more than just the species in question (green sea turtle or scalloped hammerhead shark) will be helped by the conservation program.

[30]See Lew (2015) for further discussion of willingness to pay studies related to endangered marine species, the methods used in these studies, and willingness to pay estimates for other endangered marine species.

Increasing marine ecotourism represents an opportunity to provide funds for the conservation of marine biodiversity and coastal livelihoods through visitation fees or donations (Halpenny, 2003; Mayes et al., 2004). The findings of the study provide empirical evidence and insights about the factors that drive tourists visiting the Galapagos archipelago to be willing to contribute monetarily to marine conservation in the region, and specifically to the two marine endangered species under study. This information can be used by resource agencies to understand the true potential and feasibility of alternative funding mechanisms for conservation programs in the region.

As suggested by the study findings, certain profiles of visitors to the GNP are willing to contribute toward the recovery of the threatened populations of the green sea turtle and the scalloped hammerhead shark in the Eastern Tropical Pacific. In fact, our results show that there are heterogeneous preferences among tourists interested in donating for the two marine endangered species, which can be used when designing funding mechanisms for marine conservation. For instance, funding efforts can focus on tourists who have strong preferences for environmental-related causes by targeting them at more environmentally-friendly lodging places or cruises. Partnering with institutions working on marine conservation programs and with diving agencies is also a potential mechanism to enhance fundraising opportunities for resource agencies. In addition, resource agencies may wish to focus fundraising campaigns on protection of the endangered green sea turtle, given it has a stronger positive effect on stated donation behavior. Given the overlapping habitat of the green sea turtle with the scalloped hammerhead shark and other species, protection of the green sea turtle would still have a positive effect on conservation of other species.

At the broader regional level, the findings of this study are timely for the debate over alternative funding mechanisms being considered for the Eastern Tropical Pacific Marine Corridor (CMAR), a governmental initiative to create and promote the conservation of the archipelagos in Costa Rica, Panama, Colombia, and Ecuador. One of the main goals of the initiative is to enhance protection of key migratory and endangered marine species, including hammerhead sharks and sea turtles. Increasing tourism opportunities in these islands will likely increase the number of visitors who can and are willing to donate toward the recovery of these two "umbrella" marine endangered species. Potential revenue from tourism can therefore be a feasible avenue through which funding for the CMAR initiative can occur.

AUTHOR CONTRIBUTIONS

SC and DL conceived and designed the study, and wrote the paper. SC carried out fieldwork and data analysis under DL's supervision.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fmars.2016.00060

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Mitigating undesirable impacts in the marine environment: a review of market-based management measures

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Internationally, marine biodiversity conservation objectives are having an increasing influence on the management of commercial fisheries. While this is largely being implemented through Marine Protected Areas (MPAs) other management measures, such as market based instruments (MBIs), have proved to be effective at managing target species catch in fisheries and reducing environmental impacts in industries such as mining and tourism. Market-based management measures aim to mitigate the impacts of activities by better aligning the incentives their participants face with the objectives of management, changing their behavior as a consequence. In this paper, we review the potential of MBIs as management tools to mitigate undesirable environmental impacts associated with commercial fishing. Where they exist, examples of previous applications are described and the factors that influence their applicability and effectiveness are discussed. Several fishing methods and impacts are considered and suggest that whilst no single approach is most appropriate in all circumstances either replacing or complementing existing management arrangements with MBIs has the potential to improve environmental performance. This has a number of implications. From the environmental perspective they should enable levels of undesirable impacts such as damage to sensitive habitat or the bycatch of protected species of turtles, marine mammals, and seabirds to be reduced. The increased flexibility MBIs allow industry when developing solutions also has the potential to reduce costs to both the industry and managers, improving the cost-effectiveness of regulation as a result. Further, in the increasingly relevant case of MPAs the need for publicly funded compensation, often paid to industry when vessels are excluded from grounds, may also be significantly reduced if improved environmental performance makes it possible for some industry members to continue operating.

Keywords: MPAs, marine conservation, quotas, taxes, penalties, bonds, protected species

Introduction

Over the last two decades, the development of international[1] and regional[2] conventions to protect marine biodiversity has resulted in greater commitments to mitigate undesirable impacts in the marine environment, mostly through increasing the amount of habitat closed to fishing. For example, the Convention on Biological Diversity has a global target of 10% of the marine environment being included in marine protected areas (MPAs) by 2020. In the USA, legislation is being developed at both State and Federal levels with this objective (e.g., Hildreth, 2008). The Marine Strategy Framework Directive[3] has a similar goal in Europe. MPAs are also being implemented in developing nations for both conservation and economic reasons, such as tourism and to protect coastal community livelihoods (Francis et al., 2002). Similarly, a National Representative System of MPAs has been implemented in Australia with the main goals of protecting biological diversity and maintaining marine ecological processes and systems[4].

While the non-market benefits of MPAs are potentially numerous (Angulo-Valdés and Hatcher, 2010) the costs of setting them up may also be high. Establishing MPAs generally requires fishing effort to be reduced in the area under consideration, either through buy-back programs or by the displacement of fishing effort to other areas (Sen, 2010). While the true cost of such schemes is often difficult to accurately quantify, it can be substantial (Dowling et al., 2011). Where fishing effort has been bought out, this has also often involved publicly funded compensation for related industries. For example, compensation payments associated with expanding no take zones from 4 to 34% of the Great Barrier Reef marine park are estimated to have exceeded $250 million (MacIntosh et al., 2010). Much of this was paid to onshore businesses that claimed to be adversely affected by the change (Gunn et al., 2010). Consequently, identifying policies that can reduce these costs whilst still achieving management goals is an important component of developing cost-effective approaches to marine spatial planning and management.

There is little incentive for stewardship, or to actively prevent overexploitation, when a species or habitat is not privately owned, effectively making it a common property resource (Hardin, 1968; Gordon, 1991). Market-based management measures aim to create a situation where operators' incentives are better aligned with the objectives of management, changing their behavior to mitigate the impacts of activities as a consequence. In the context considered here, management objectives may include reducing protected species mortalities or preventing damage to sensitive habitat. This is in contrast to the more familiar fisheries management problem of preventing the target stock/s from being overexploited, although the central challenge is essentially the same in both cases.

If the environmental impact of a fishery can be adequately reduced by incentivizing behavioral change, marine conservation objectives may still be met without the need to fully remove fishing from an area. Potentially, fisheries could then continue to operate at some level within the bounds of declared reserves. Behavioral changes that reduce the need to prevent activities or displace effort also have the potential to reduce the costs of conservation to both management (e.g., compensation, administration) and industry (e.g., loss of income, increased competition on fishing grounds that remain open). If the overall cost per unit of benefit gained under MBIs is lower than the alternative (e.g., compensation for the complete exclusion of vessels from an area) they will also be a more cost-effective approach. This would allow the cost of achieving a given reduction in impact to be reduced or, depending upon the management objectives, greater areas of habitat or species range to be protected with the same level of funding.

In this paper, we review the potential of MBIs as management tools to mitigate the undesirable environmental impacts associated with commercial fishing and consider how this might reduce the need to exclude fisheries from MPAs. Where they exist, examples outlining previous applications of MBIs are described and the factors that influence their applicability and effectiveness are discussed. While our focus is on fishing, these tools are potentially applicable to other industries whose actions can impact the marine environment in undesirable ways (e.g., dredging for port development).

The paper is organized as follows: The next section, Fisheries Impacts on Marine Environments, outlines some of the key fisheries impacts that could potentially be reduced through the use of MBIs. This is followed by the section Market-based Instruments and Fisheries Management Measures, which outlines a range of potential MBIs, first considering measures based on financial incentives before discussing quota oriented approaches. The discussion section then addresses some additional factors for consideration and limitations that influence how these tools may be applied.

Fisheries Impacts on Marine Environments

The impacts fishing can have on the marine environment are well-documented (e.g., Tasker et al., 2000; Kaiser et al., 2002). In addition to catching their target species, fishing vessels can impact non-targeted species, some of which may be threatened, endangered, or protected species caught incidentally. In some cases, marine habitats are directly damaged, while associated ecological communities are impacted through ecosystem interactions (Hobday et al., 2011). These impacts may involve species of no direct commercial value but of considerable non-market value (e.g., iconic and often protected species such as turtles, dolphins, and seabirds). As the cost of this damage is often not borne by the fisher, levels of damage are typically greater than

[1]E.g., United Nations Conference on Development and Environment; the Convention on Biological Diversity; United Nations Conventions on the Law of the Sea.

[2]E.g., Convention on the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region (Nairobi Convention), Natura 2000 (EU).

[3]Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy.

[4]Whilst these reserves are now in place the management plans were set aside before coming into effect as part of the ongoing Commonwealth Marine Reserves Review.

the social optimum[5]. MPAs can limit or reduce this damage by preventing access to areas that are considered to have substantial non-market values (e.g., large populations of iconic species such as turtle nesting areas), but in doing so impose costs on fisheries and management agencies.

The specific impacts that a fishery has, how predictable these events are, and the frequency with which they occur vary by fishery and region. For example, in Australia the southern demersal trawl fisheries have heavier impacts on non-target demersal fish species and sharks, while seabirds are among the most heavily impacted bycatch species for the southern longline pelagic fishery (**Table 1**). Similarly, the impact on certain habitats varies with the type of fishing gear being applied. For example, habitats may be impacted by trawling but not by line fishing.

The frequency of an impact's occurrence is a function of the species or habitat present, the gear used, and how and when the gear is applied. The type of gear used directly influences the species or habitats a fishery is capable of interacting with, whilst the region and season it operates in influences the species and habitats that may potentially be impacted. MBIs attempt to reduce overall impact by making fishers accountable for the consequences of their actions which can influence behavior with respect to the gear type choice and configuration along with when the gear is applied. Last, fishers may have limited control over the degree of uncertainty associated with causing an impact and this is one of the factors that have a direct bearing on the applicability of specific MBIs.

Market-based Instruments and Fisheries Management Measures

Command-and-control measures generally dominate fisheries management internationally. These include forms of both input (e.g., gear or effort constraints) and output (total allowable catches) oriented measures. They are prescriptive by nature, so tend to be inflexible, not allowing individual solutions to problems, and are potentially inefficient as a consequence. Input oriented management measures are typically least favored by economists, as there is the risk of constrained inputs being wastefully substituted with unconstrained ones. While catch and conservation objectives may be achieved in some cases, these can be at the cost of high levels of inefficiency in the industry and consequent overuse of resources (Townsend, 1985). In many cases, even the key objectives are not achieved. Attempts to manage overall levels of target species catch (or bycatch) in fisheries by limiting effort via relatively easy to measure inputs such as hooks set or days fished typically fail as fishers increase their use of non-regulated inputs instead.

Output measures that focus directly upon monitoring and controlling the quantities of catch or bycatch a fishery takes provide greater certainty that management objectives relating to the fishery resource will be achieved. Further, when catch shares are individually allocated, they also have the ability to induce more efficient behavior (Grafton, 1996). For example, when a total allowable catch (TAC) is allocated to individuals, wasteful incentives to race to fish are reduced and replaced with individual incentives to minimize costs. If not constrained by input controls, output oriented approaches can also result in greater efficiency by being more flexible and allowing fishers to develop or apply methods of impact mitigation that work best in their specific circumstances. The most cost-effective way of catching target species or reducing impacts will potentially vary between fleets and even between individual vessels.

MBIs generally work by creating a price (either explicitly or implicitly) for the use of a non-market resource in the production process. This price reflects the cost imposed by the activity and primarily borne by society until then. Requiring operators to account for this cost creates an incentive to reduce their impact. In the case of fisheries, these undesirable impacts include bycatch of non-target species (including iconic/protected species) as well as habitat damage. MBIs differ from command-and-control measures in the way that they rely on price signals, applied at the individual or firm level, to incentivize changes in behavior and outcomes. They may be applied in addition to existing command-and-control measures (e.g., ITQs on top of spatial constraints) or instead of them (e.g., penalties instead of regulations specifying how and when to operate gear). Where vessels are heterogeneous in their ability to reduce impacts, tradable quotas facilitate further efficiency by creating additional financial incentives for quota to pass to vessels that can use it most efficiently, working in the same way as ITQs for target species.

A hierarchy of potential market-based management systems is presented in **Figure 1**. Incentives can be created by either placing constraints on the level of impact fishing activities have, or by influencing the rewards from fishing. Constraints such as bycatch quotas are flexible and differ from hard constraints, such as area closures, as fishers are potentially able to adjust the level of their individual constraint through quota trading. Non-tradable quotas are not strictly MBIs, in the sense that there is no market for them, but they can still create incentives to reduce impact at the individual level if not doing so would result in vessels having to prematurely stop fishing (and thus they have an implicit value associated with them). Financial incentives include the use of charges, subsidies or bonds. Charges and subsidies directly affect the returns from different fishing activities, thereby stimulating behavioral or technological change. Bonds incentivize similar types of change by providing the incentive for fishers to reduce their impacts sufficiently below some threshold. The expected behavioral response varies slightly depending on the type of policy instrument chosen. Tradable quotas and penalties are generally anticipated to result in individuals attempting to minimize the level of impact they create; as fewer penalties reduces costs and the ability to operate with a low level of quota either reduces costs or increases income. On the other hand, bonds and insurances are expected to create the incentive to ensure that impacts are limited to an agreed level. However, even

[5]Such market failure has been identified in a wide range of industries, and is not exclusive to fisheries. For example, the costs of pollution externalities are not generally considered by the polluter in their production process. The divergence between private and public optimum due to externalities has been long recognized in the economics literature (e.g., Pigou, 1924).

TABLE 1 | Examples of how the frequency of non-commercial bycatch can vary by gear type and region in some Australian fisheries[6].

Frequency of occurrence	Demersal trawl		Demersal longline	Pelagic longline		Demersal gillnet
	North	**South**		**North**	**South**	**South**
Infrequent		Whales				
	Seabirds	Turtles		Seabirds		
		Dolphins				
	Turtles	Seabirds				
	Sea snakes	Seals/Sealions		Turtles		Seals/sealions
	Sharks	Sharks	Sharks		Seabirds	Sharks
Frequent	Other fish	Other fish	Other fish	Sharks	Sharks	

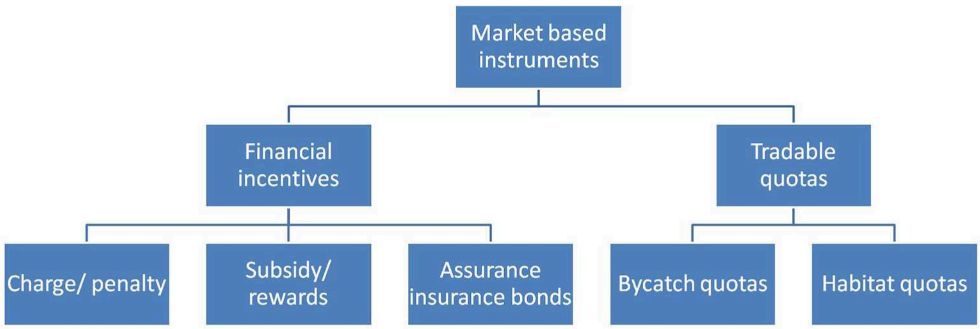

FIGURE 1 | Hierarchy of market-based fisheries management systems for reducing environmental damage.

in this case there is still some incentive to try and keep below this limit to avoid accidentally exceeding it.

Previous assessments of MBIs have also included eco-labels and trade-barriers (Pascoe et al., 2010). However, these are largely related to fishery-wide behavior and often require additional management measures to create the individual incentives required to achieve the desired outcomes. Similarly, when damages are known or even perceived to occur by the public, the loss of "social license to operate" may translate into financial cost in terms of reduced revenues (through lower demand for the product) and potentially greater regulation or restrictions. These again require fishery level solutions, but may utilize MBIs to create the appropriate individual incentives. For the sake of brevity, these fishery level issues are not considered here.

Finally, to be effective all policy instruments depend on adequate levels of compliance. A range of factors have been identified as influencing compliance in the fisheries context, central to which are economic incentives and deterrence (Sutinen et al., 1990; Nielsen and Mathiesen, 2003). Exactly what needs to be monitored to create a deterrent varies between alternative applications (i.e., the chosen MBI and the specific impact it is being applied to) but in all cases it is important that the likelihood of an impact being detected is high. When this condition is met MBIs can directly alter the economic incentives fishers face.

Financial Incentives

Charge or Penalty-based Systems

In addition to the potential opportunity cost associated with disposing of bycatch, damage to gear (e.g., from bycatch or habitat), reduced harvest due to bait and hooks being consumed by non-target species, or damaged and devalued target species[7] are all potential costs of poor environmental performance. However, as these costs are often relatively small or poorly accounted for there is consequently little incentive for fishers to limit their impact when operating. Bycatch and other environmental impacts can thus be considered unpriced inputs in the production process.

Placing an appropriate price on these environmental impacts provides incentives for fishers to modify their behavior (i.e., production and fishing effort allocation), and to adopt impact-reducing technologies that reduce these costs. Where such technologies do not exist, correctly set charges will encourage their development. For example, the use of carbon charges has been seen to influence both energy mix in manufacturing and total demand by households (Johansson, 2000; Bruvoll and Larsen, 2004; Tietenberg, 2013). Carbon charges have also been seen to induce technological change that substantially accelerates the substitution of carbon-free energy for fossil fuels (Gerlagh and Lise, 2005). An advantage of a penalty system is that, at least in theory, different impacts (and species) can attract different

[6] In addition to species specific assessments (e.g., Stewardson, 2007; Trebilco et al., 2010), the Australian Fisheries Management Authority (AFMA) Bycatch and Discard Program is a central source of information for bycatch in Commonwealth fisheries http://www.afma.gov.au/sustainability-environment/bycatch-discarding/

[7] For example, if fish can be bruised by interactions with bycatch whilst in the cod end, reducing their quality and value. Crab bycatch can also damage and devalue target species in shrimp trawls.

penalty rates, thereby ensuring the greatest protection to that which is most vulnerable.

Whilst there is no direct incentive to target non-commercial species, the value of any associated target species can result in them still being caught, especially if there is no explicit cost associated with doing so. The potential benefits of monetary penalties for reducing the level of bycatch of non-commercial species, particularly megafauna (e.g., seals, turtles, seabirds), have been demonstrated theoretically by a number of authors (Sanchirico, 2003; Diamond, 2004; Herrera, 2005; Singh and Weninger, 2009). Limited examples exist of charges being implemented on commercial but non-targeted species. Where applied, it has been with the intention of either encouraging fishers to avoid the species (Schrank et al., 2003), or providing a mechanism through which species without quota can be landed (Sanchirico et al., 2006).

Where fishers are able to avoid non-commercial species and incidents are observable, a bycatch charge is likely to influence their behavior and reduce the catch of these species. Similarly, penalties linked to operating in certain areas will create incentives for fishers to look elsewhere without permanently locking fishing activity out of these areas. Given seasonality in ecological systems, such a charge can be readily adjusted to provide a greater disincentive to operate in an area in times of high sensitivity (e.g., spawning seasons), and a weaker disincentive in less critical times. However, as with many of the policies discussed in this review, the effectiveness of the MBI will depend on the actual ability of the fisher to avoid the species or areas of concern.

The implementation of both habitat use and bycatch penalties requires information on fishing activities and catches. Historically, these data have been relatively expensive to collect, due to the need for independent observers on each vessel during fishing operations. However, the continued refinement of technology such as electronic monitoring systems (EMS) has the potential to make it increasingly cost-effective when compared to observer coverage, and to revolutionize the use of this type of incentive (Bryan et al., 2011; Kindt-Larsen et al., 2011; Piasente et al., 2011; Seafish, 2012). Vessel monitoring systems (VMS), which can track individual vessel location, are now common among larger fisheries and can provide a means of determining when and where a vessel is fishing (Witt and Godley, 2007). With such information, charges can be readily applied if vessels operate in ecologically sensitive areas.

Penalties are not likely to be appropriate for impacts that are highly stochastic in nature (i.e., essentially random) as it makes them difficult to predict and consequently hard to avoid. In this situation, imposing penalties that are large enough to create a strong incentive to reduce impact may also result in operators facing untenable financial risk every time they go fishing. A further factor for consideration with penalties is that to create a strong incentive to reduce impact, they would need to be payable soon after issue. Allowing penalties to accrue increases the risk of default and diminishes the impact/cost association. From a practical perspective, this means that penalties are likely to function better in cases where impacts are reasonably predictable and infrequent rather than situations characterized by high levels of uncertainty or high frequency of occurrence, the same problem Holland (2010) identifies for bycatch ITQs. Impracticably large numbers of penalties would need to be issued in the latter case creating unnecessary additional costs for both industry and regulators, making the measure inefficient.

Direct Subsidies and Payments

The use of subsidies to reduce environmental impacts is limited in fisheries. Where subsidies exist, these are usually related to reducing the cost of less damaging fishing gear to encourage its adoption (Cox and Schmidt, 2006). However, even so called "environmentally friendly" subsidies can result in increased exploitation by reducing the cost of fishing (Cox and Schmidt, 2006), and potentially increase total damage as a result.

Payments to individuals to ensure the protection or enhancement of ecosystem goods or services are an established market-based instrument for habitat and species protection in terrestrial conservation (Farley and Costanza, 2010; Muradian et al., 2010). The potential of such an approach to managing the impacts of fisheries is still emerging as it requires well-defined and secure property rights over the good or service being protected[8] and effective enforcement (Bladon et al., 2014). Critics of the approach also suggest that payment for such services can undermine the moral sentiments for conservation, moving it from ethical consideration to economic self-interest (Gómez-Baggethun et al., 2010). Gains may also be lost over the longer term if agreements cannot be maintained over time and are allowed to lapse. The resumption of dolphin hunting in the Solomon Islands in 2013 after the breakdown of an agreement between villagers and a conservation group that had been providing financial support to develop alternative activities illustrates this point (Oremus et al., 2015). Whilst the breakdown of any MBI discussed in this paper will potentially result in their benefits being lost, the risk of this occurring is greater with payment schemes as participation is typically voluntary from the perspective of the provider and not mandated as is the case in other MBIs.

Assurance Bonds and Insurance

Assurance or performance bonds are economic instruments commonly used in environmental management (Shogren et al., 1993; Cornwell and Costanza, 1994; Ferreira and Suslick, 2001; Bagstad et al., 2007). The aim of the bond is to ensure that the worst case cost of any damage that remains once an activity has been completed is covered (Perrings, 1989; Costanza and Perrings, 1990). This does not necessarily require an upfront payment[9] , and may instead involve a bank-backed guarantee of payment in the event that the restoration is not satisfactorily undertaken by those that caused the damage, or damage is incurred that cannot be rectified. In addition to incentivizing producers to limit impacts it also ensures funds are available to rectify any damage once the activity has been undertaken. Assurance bonds have been used in a wide range of industries

[8]Well-defined and secure property rights typically ensure exclusivity, durability, transferability, divisibility, flexible in nature, and good quality of title.

[9]Some earlier schemes required an upfront posting of the bond, creating liquidity constraints in cases where the producer could not raise the bond (Shogren et al., 1993).

to ensure appropriate environmental outcomes (Costanza and Perrings, 1990; Cornwell and Costanza, 1994; Gerard and Wilson, 2009), including terrestrial based mining operators in Australia and New Zealand (White et al., 2012) and elsewhere (Shogren et al., 1993; Gerard, 2000). Bonding programs in the US have been set up to incentivize compliance with environmental requirements when closing oil and gas operations. These appear to have been relatively successful, with non-compliance rates between only 1 and 9% (Gerard and Wilson, 2009). In Western Australia, <2% of mining bonds are called in White et al. (2012).

Assurance bonds have a number of perceived advantages. First, they ensure that sufficient resources are available for rehabilitation in the case that a firm becomes insolvent before restoration is undertaken (White et al., 2012). Further, as the money is already held by the enforcing agency it is up to the firm creating the damage to demonstrate no net loss, rather than for the enforcing agency to prove the contrary. Such a shift in the burden of proof also creates the incentive for firms to research the future environmental costs of their activities if they want to challenge the level of bond set by government (Costanza and Perrings, 1990).

Within the marine environment examples of applications of bonds are currently limited. One example is the Great Barrier Reef Marine Park, where they are a key instrument in the management of approved development activities (e.g., marina development or associated dredge disposal), requiring either a cash bond or bank guarantee ranging from $50,000 to $500,000 depending on the scale of the development (Great Barrier Reef Marine Park Authority, 2010). These funds have been accessed on a number of occasions to remove abandoned equipment from activities such as tourism and pearl aquaculture (ABARE, 1993; Lal and Brown, 1996; Smith et al., 2005). Financial assurance is also required for oil and gas development in the Australian marine environment, with the level of assurance based on a combination of factors, including the type of hydrocarbons, the potential spill volume and the potential area of shore impacted (APPEA, 2014). Similarly, offshore renewable energy installations in the US require a bond to ensure that decommissioning requirements are satisfactorily met after the facilities (e.g., pipelines, cables, and other structures and obstructions) are no longer required (Hill, 2011; Kaiser and Snyder, 2012). Comparable arrangements are also in place in most other countries for offshore energy developments, particularly in relation to appropriate decommissioning of offshore oil and gas facilities (Ferreira and Suslick, 2001).

Pascoe et al. (2010) outlined how bonds could be applied to manage fisheries interactions in the marine environment, with the bond returned provided fishers achieved a pre-determined performance target in terms of bycatch rates or avoidance of habitat impacts. Access to different areas of the fishery could be subject to different bond levels depending on environmental sensitivity. Individual fishers could choose to either pay the bond to access a particular area or fish elsewhere. The bond creates an incentive to either adopt technologies to minimize the chance of violation (if operating in the bonded area), or to avoid the sensitive area entirely (Pascoe et al., 2010). This provides an alternative to total exclusion, the counterfactual when implementing an MPA. Less fishing in the bonded area reduces the likelihood of the adverse environmental impact occurring, and may also benefit any fishers that remain as less pressure on the resource has the potential to result in higher catch rates (at least in the short run). Allowing non-impacting operators to remain in an area will also reduce the level of effort that is ultimately displaced to other areas or that requires buying out.

Monitoring and enforcement of such a system is potentially challenging. VMS enable identification of whether and for how long vessels are fishing in a bonded area. However, attributing any observed damage to individuals is problematic, particularly if multiple vessels are fishing in a sensitive area at the same time. While estimates of habitat damage could be derived from monitoring the amount of time fished in an area and the particular type of gear, the uncertainty around this is likely to result in legal challenges if attempts to seize bonds are made (Pascoe et al., 2010). While fishery level (rather than individual level) bonds are also an option, these may provide adverse incentives, as if fishers anticipate that the bond will be lost through the action of others there will be little incentive to limit their own impacts (Pascoe et al., 2010).

An alternative to assurance bonds is requiring developers or proponents of other activities in the marine environment to insure against the costs of restoration of (or compensation for) potential environmental damage. A potential benefit of an insurance-based system is that the risk could be sold on the insurance market, with industry members paying a premium to the insurer which reflects the insured's past performance and adoption of mitigation technologies (Pascoe et al., 2010). As with assurance bonds, the aim of insurance is to provide incentives to avoid damage, as those that are most successful (through their actions or technologies employed) will face lower premiums. Insurance markets have been used in the management of pollution in a number of countries (OECD, 2003), and there is generally a mandatory requirement for oil tankers to have appropriate insurance against oil spills in the marine environment (Chiau, 2005; Zhu, 2007). Ahvenharju et al. (2011) found that insurance-based systems were most suitable where adverse outcomes may involve high costs which individuals were unlikely to be able to meet, but the likelihood and consequences of these outcomes were highly uncertain. An advantage of insurance in this respect is that the cover is potentially open-ended, unlike bonds which are set at a predetermined level.

In the case of marine interactions, insurance schemes are likely to be most effective when the chance of an impact is relatively small (Holland, 2010) and highly observable, but where the consequences of the impact are relatively significant from an ecological perspective. In this case the insurance may cover the costs associated with having to exclude all other vessels from an area or close the fishery should the impact occur. For example, when vessels are monitored (e.g., via observers, electronic monitoring) the bycatch of turtles and marine mammals are readily observable. They are also potentially more avoidable than some other bycatch such as non-commercial finfish species. By and large, most fishers aim to avoid the bycatch of these species, although there is evidence that different groups within a fishery adopt bycatch reducing technologies at different rates (Jenkins and Garrison, 2013). Lower insurance costs for the use of more environmentally friendly fishing gear provide added incentive for their earlier adoption and development.

Quota Systems

Quota systems involve setting a total permissible level of impact, and are typically employed on the basis that when the quota is reached the vessel or fishery in question must cease operating for the remainder of that season/period. These systems impose a hard cap on the level of impact and can be applied at either the fishery level (common pool) or to individual operators. The incentives created differ depending on which level is implemented. There is a strong theoretical basis for assigning quotas at the individual level and then allowing them to be traded between participants as in a well-functioning market this makes it possible for quota to pass to operators that can use it most efficiently (Moloney and Pearse, 1979; Clark, 1980; Grafton et al., 2000). A number of quota based systems have been proposed and this section considers those based on bycatch and habitat.

Bycatch Quotas

Bycatch quotas are aimed at limiting the total incidental catch of specific species (commercial or non-commercial). A detailed review of the advantages and disadvantages of each system in terms of reducing fishing bycatch has been provided by Pascoe et al. (2010) so only the key points are discussed here.

There are a number of cases where bycatch limits for non-commercial species are imposed on fisheries, although these have mostly been related to bycatch of megafauna, particularly the more charismatic species. In the US, a total allowable catch of turtles or marine mammals is in place in several fisheries (NOAA, 2004), while limits on dolphin mortality in international tuna fisheries are also common (e.g., IATTC, 2008). New Zealand also uses output controls to manage bycatch of Hooker's sea lions in the arrow squid trawl fishery (Bache, 2003; Diamond, 2004; Chilvers, 2008). In Australia, a seabird bycatch threat abatement plan relating to bycatch during oceanic longline fishing operations currently imposes a catch rate limit (Department of Environment and Heritage, 2006). Whilst not a quota *per se* this approach aims to constrain the level of impact the fishery imposes and when these limits have been reached the fishery is either closed or substantial parts of its grounds are shut down (Pascoe et al., 2011). Dunn et al. (2011) suggest that such spatial and temporal closures in themselves may be appropriate management measures to limit bycatch of species irrespective of the observed level of catch.

The unintended economic impacts of common pool quotas can be substantial and greater than alternative management approaches (Pascoe et al., 2011, 2013). Abbott and Wilen (2009a) suggest that such quotas result in a "race for fish" and fisheries characterized by excessive rates of bycatch, shortened seasons, and foregone target species harvest, even when efficient (i.e., low bycatch) fishing gear is used. Delays in information collection may also make the restriction ineffective. For example, a spike in the level of demand for swordfish in 2006 resulted in a race to fish, with a large increase in the number of hooks set early in the year and the expectation that the bycatch quota would result in the fishery being closed early (Gilman et al., 2007). Alaskan bottom trawlers were also observed to have had limited success at mitigating halibut bycatch when this was managed under a common pool cap that relied on voluntary cooperation between vessels to prevent it closing the fishery before the commercial TACs were taken (Abbott and Wilen, 2010). The introduction of a formal cooperative that allocated individual quotas for target and prohibited species to its members was far more successful at altering fisher behavior though by making fishers individually responsible for their own bycatch and altering their incentives in the process (Abbott et al., 2015).

Several authors have investigated the use of individual transferable bycatch quotas (ITBQs) as a means of reducing bycatch for both megafauna (Bisack and Sutinen, 2006; Hannesson, 2006; Bisack, 2008; Ning et al., 2009) as well as fish species—either commercial (by-products) or non-commercial (Boyce, 1996; Diamond, 2004). However, relatively few real life examples of ITBQs can be found, and those that exist are focused on bycatch of commercial species. In 1996, Canada instituted an individual vessel bycatch quota (IVBQ) for its groundfish trawl fleet (Diamond, 2004), while several shark species caught as bycatch are included in the NZ quota management system. A system of individual bycatch quotas for US fisheries was found to be less successful, particularly when total quantities of bycatch were low and effectively a random event (Holland, 2010). In such cases, individual quota allocations are low and can result in illiquidity and high transactions costs. A potential consequence of this is that a fisher who is unfortunate enough to exceed their quota on a trip may find it costly to source and purchase additional quota if the unpredictable nature of the impact results in other (risk averse) fishers being reluctant to sell due to concerns that they will subsequently need it themselves. In such cases where impacts are infrequent and uncertain, greater benefits may be obtained by fishers pooling their individual quotas, reducing both financial risk and transactions costs for individuals (Holland and Jannot, 2012).

A potential limitation of quota pools is that the inefficiencies associated with common pool quotas may arise if the TAC is reached prematurely as a consequence of moral hazard, where operators can still benefit at the individual level from racing to fish (Abbott and Wilen, 2009b). Mechanisms such as revenue pooling are potential solutions to this issue but can introduce other efficiency problems due to free riding (Uchida and Baba, 2008). The formation of smaller quota sharing groups, where participants know one another and there is greater trust, may be more effective if this social capital incentivizes collaborative behavior (Pretty, 2003).

Individual Habitat/Spatial Effort Quotas

An alternative quota is the individual habitat quota which takes the form of an effort control (Holland and Schnier, 2006). These are spatial management instruments where different levels of effort penalty are applied based on the level of damage created by fishing in those areas. These quotas are tradable, allowing vessels to plan and adjust their fishing activities to minimize their own damage. Fishers "consume" their quota based on where, when and how they fish, with the penalty system providing incentives to either operate in areas where less damage will be incurred, or adopt fishing gear that will have a lower impact. Ideally, such a system would impose differential penalties based on gear used. Such a system provides an incentive to either reduce effort, or use more environmentally friendly gear, in sensitive habitats without the need to impose a total closure. Perhaps the only current example of this type of measure is the use of Habitat Bycatch Conservation Limits (HCBL) in the BC groundfish fishery, which imposes individual bycatch limits for cold-water corals and sponges. Initial reports appear promising and suggest that this approach has resulted in management targets being met as a result of immediate behavioral responses and substantial reductions in the total quantity of bycatch (Wallace et al., 2015).

While not designed with bycatch in mind, such a system can also be adapted as a bycatch management system. Modeling of a variable effort unit system based on fishing location proposed for an Australian tuna fishery to reduce bycatch of seabirds found that such a system could effectively control bycatch at lower cost to the industry than the current area closures (Pascoe et al., 2013).

Discussion

The preceding section illustrates the range of MBIs available as tools to alter the incentives commercial fishers face to reduce their impacts on the environment. Whilst examples of MBIs being applied in this context are still relatively limited in number and often in their infancy when compared to other industries, considering the lessons learnt from experiences to date in conjunction with known practical and theoretical limitations is informative. The continued movement toward fisheries management from a broader ecosystem based perspective and the associated requirements to reduce impacts suggest that the importance of MBIs in helping realize these goals is likely to increase into the foreseeable future. Similarly, the adoption of multiple-use zoning systems in MPAs, where fishing is permitted in some areas (e.g., Day, 2002; Boyes et al., 2007), demonstrates that there is a need for more flexible and complimentary management arrangements to ensure conservation objectives are achieved.

Whilst transferable quotas are arguably the most familiar MBI in the context of fisheries management at this point, their primary application remains as a means of managing target species catches. From a theoretical perspective, quotas, and specifically ITQs, are an economically attractive approach to effectively constrain undesirable outputs. The level of information and therefore cost that is necessary for estimating appropriate penalties to reach a particular quota is likely to be greater compared to setting a quota and adjusting. However, to attain socially optimal quota levels or optimal penalties for equivalent impact reductions, information relating to the full costs and benefits of impact abatement are necessary. Assuming acceptable compliance, the primary limiting factor of ITQs resides in their reliance on conditions that facilitate well-functioning markets. They are consequently likely to function best in situations where multiple participants and relatively frequent impacts result in high volumes of quota and trade.

Poor levels of market participation and illiquidity, limited or asymmetric information, or the existence of participants with excessive market power can result in high transaction costs, insufficient trades occurring and market failure (Farrell, 1987; Stavins, 1995). In these situations the long-run efficiency gains potentially available with ITQs will be diminished, preventing an efficient distribution of quota from being achieved (Anderson, 1991, 2008). Fisheries with ITQs for relatively infrequent and stochastic bycatch have been observed to be inefficient as uncertainty creates strong incentives to retain quota, resulting in thin and poorly functioning quota markets as a consequence (Holland, 2010). In such cases, greater formal cooperation between fishers and the pooling of quota is preferable, so that operators are less dependent on markets but can still access quota to mitigate risk efficiently (Holland and Jannot, 2012).

The examples provided in the previous section illustrate how the case specific characteristics of an environmental damage problem can influence the capacity of fishers to adapt; these are important factors for consideration since they can influence the practicality and consequent effectiveness of particular management measures. The importance of case specific characteristics is also highlighted by Holland and Jannot (2012) when discussing the appropriateness of either individual or pooled quotas but it appears that these factors often influence the choice of MBI more generally. They list frequency of bycatch by species, variance and distribution of events, numbers of events per vessel, whether risk of bycatch and profit are correlated, whether bycatch is heterogeneous across vessels, and whether real time information would reduce bycatch. For example, whilst ITQs may outperform penalties when impacts are either frequent or stochastic, if impacts are likely to occur infrequently and have a degree of predictability a system of penalties will generally be the more appropriate approach.

Penalties are flexible and can easily be adjusted to meet management objectives. If desired, penalties that increase with the level of bycatch or damage can also be implemented, increasing either progressively or in a stepwise fashion once defined thresholds are reached. To prevent them being considered just another cost of operating the level penalties are set at, or the rate at which they increase, would need to reflect the severity of the impact. In doing this, minimum levels of bycatch or damage may effectively be realized at least cost to the industry. Fishers who do what is possible to reduce bycatch but occasionally catch some will receive generally low penalties, while those who do not take measures to reduce their impact will end up with higher penalties.

For MBIs to be successful, compliance is necessary; if creating an impact does not result in the charge being imposed, quota consumed, or bond forfeited, there is no incentive to alter behavior. Ensuring compliance under systems of payments is just as important due to the obvious incentives for individual to try and game the system for their own gain. In most situations, some form of surveillance is required if compliance is to be ensured and in many fisheries the most effective way of monitoring what vessels actually catch has been through the use of onboard observers. Observer schemes can be costly though, and whilst it is possible to require that industry funds it this might not be practicable in reality. Lack of space can also prevent observers from working on smaller vessels. Potentially lower cost alternatives to onboard observers, such as electronic monitoring may be required and the continuing development of this technology is making it increasingly feasible (Kindt-Larsen et al., 2011; Petter Johnsen and Eliasen, 2011; Piasente et al., 2011; Seafish, 2012). The benefits of this are potentially twofold in that it could both reduce costs as well as allow vessels too small to physically accommodate an observer to effectively demonstrate compliance (and thereby continue to operate in certain areas).

For marine habitats, the use of individual habitat quotas may be the most effective means of limiting damage inside marine reserves (other than complete exclusion). These have the potential to be applied both in cases of undifferentiated habitat types where impacts are to be reduced, or in patchy environments where certain habitats need to be avoided. An attractive feature of this measure is that compliance can be easily assessed using VMS data, especially in combination with a video system that monitors fishing activity. A key challenge is to determine the total level or area of impact deemed as acceptable over any given period of time (e.g., season/year/indefinitely). If the ultimate aim is to progressively reduce aggregate impact, the total level of permissible impact may be reduced over time so that fishers must either apply less effort in that area or become more environmentally efficient (e.g., via the development of gears that result in lower levels of impact per unit of effort applied). Variants of this type of spatially related effort measure may also be applied to tackle bycatch when the areas in which the bycatch occur are discrete and do not overlap the majority of the target species distribution. A limitation to the gradual implementation of habitat quotas is in low energy environments, especially the deep sea, where habitat regeneration times may be measured in decades or centuries rather than years.

Both penalties and tradable quotas have the potential additional benefit of raising revenue that can be used for a variety of purposes, including funding conservation activities. Payments for the consumption of non-market resources by fishers to other groups are an alternative approach to offset their environmental impact. Removing predators has been seen as an economically feasible conservation action to protect turtles (Engeman et al., 2002, 2010) and seabirds (Wilcox and Donlan, 2007; Donlan and Wilcox, 2008; Pascoe et al., 2011), and fishery funded nesting site protection has been demonstrated to be a cost-effective and successful means of reducing impacts on turtles (Gjertsen et al., 2014).

An alternative to habitat quotas is the use of bonds or insurance that are either forfeited or claimed, respectively, if predetermined levels of impact are exceeded within a defined period (again typically a season or year). The level of a bond could be based on the cost of replacing damaged habitat, the cost to the rest of the fishery due to these grounds being closed for a period of time, or both. When critical impacts are likely to occur in a relatively small geographical area and additional controls outside these areas are not deemed necessary for conservation purposes bonds may be a more appropriate approach than habitat quotas as these situations are likely to results in low volumes of quota and trade. A situation that is much the same as how the management of infrequent and predictable bycatch are better suited to penalties than quotas.

Concluding Remarks

The focus of this paper has been on fishing impacts as these are prevalent in the marine environment. However, many of the instruments considered are also applicable to other marine industries, particularly the use of assurance bonds and requirements for appropriate levels of insurance against environmental damage.

A key message from the review is that no single approach is most appropriate in all circumstances and that the defining characteristics of the situation need to be identified and understood. Characteristics such as frequency of occurrence, the extent to which an impact may be predicted, and the seriousness of an impact occurring can then be used to help guide the process of determining which measure should be most effective. For example, if impacts occur infrequently and there is capacity to avoid them then penalties may be efficient; but if impacts are frequent and unpredictable then this approach is unlikely to work. Conversely, insurance markets may not be appropriate in small fisheries due to the limited ability of the insurers to spread the risk but these may be optimal measures in large fisheries.

It is well-recognized that fishers and other users of the marine environment respond to the set of incentives created by the management system within which they operate. Using this, an appropriate set of incentives can be created that limit environmental impacts. Real world experience with many of these instruments is still limited, particularly in the fisheries context where many examples remain more theoretical than empirical. However, real world experiences in the absence of adequate incentives have been demonstrated to result in undesirable outcomes such as poor environmental performance or high costs being imposed on resource users.

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Understanding Non-compliance With Protected Species Regulations in the Northeast USA Gillnet Fishery

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Marine mammals and sea turtles in the United States are protected from commercial fishery interactions under the Marine Mammal Protection Act and the Endangered Species Act. To reduce harbor porpoise bycatch in the northeast sink gillnet fishery, fishermen are mandated to attach pingers to their nets in regulated areas. Although, pinger regulations have been in place for over a decade, in practice, enforcement is weak and the penalty for a violation is almost non-existent. In this scenario, the presence of normative factors may motivate a fisherman to comply with the pinger regulation. This study considers both economic and normative factors within a probit framework to explain a fisherman's compliance decision. Model results indicate fishermen who previously violated pinger regulations, who are not completely dependent on gillnet gear and face a lower chance of being detected by an observer, are more likely to violate. Understanding the influence of normative factors on compliance decisions is a key component for higher compliance. That is, incorporation of these factors in the design of policy instruments may achieve higher compliance rates and thus more success in protecting these species. Our model findings were ground-truthed by conducting focus group research with fishermen using pingers; some preliminary findings are shared in the discussion in support of our model results. Finally, these results also suggest observer data can be used to support compliance and enforcement mechanisms in this fishery and possibly other fisheries as well.

Keywords: non-compliance, fisheries, normative factors, law enforcement, observer effect, U.S. endangered species, marine mammals

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INTRODUCTION

Non-compliance with regulatory requirements can derail resource management objectives. Biological assessments used to monitor the health of a stock can trigger management responses and regulatory actions when stocks are in danger of over-fishing. In most cases, fisheries and marine mammal management rely on regulatory instruments such as a command-and-control approach, in the form of fishing effort reductions and gear standards to protect the stock. Regulatory instruments *direc*t individuals how to behave; while economic instruments, market based, can be designed with incentives to *influence* an individual's behavior, to achieve the same desired goal. Therefore, choosing a policy instrument is a strategic choice. Resource managers can use any combination of instruments, however, if goals are not met, non-compliance may be the source of failure and not the policy instrument itself; additional policy instruments may not rectify the problem and cause further economic harm. Hence understanding the underlying motivation of behavioral responses to regulations is crucial and may allow us to design more successful policy instruments. In this paper, we examine economic and normative factors that may motivate compliance behavior in the sink gillnet fishery in relation to required gear standards in order to protect porpoise under the United States (U.S.) Marine Mammal Protection Act (MMPA) of 1972 (16 U.S.C. 1361 et seq.).

The MMPA established a long-term regime for governing interactions between marine mammals and commercial fishing operations; the potential biological removal (PBR) control rule enacted under the MMPA Amendments of 1994, specifies the allowable level of human-induced mortality for a marine mammal stock (MMPA 1972, section 1386). In the northeastern United States (US), the National Marine Fisheries Service (NMFS) is primarily responsible for protecting populations of harbor porpoise (*Phocoena phocoena)*, northern right whales (*Eubalaena glacialis)*, coastal bottlenose dolphins (*Tursiops truncates),* and loggerhead sea turtles (*Caretta caretta)* via the MMPA and Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531) (Resolve, 1996; NMFS, 2002, 2005; NOAA, 2006a,b,c). One of the major threats to their survival is lethal injuries from interactions with commercial fishing gear, including sink gillnet gear.

Most policy instruments the National Oceanic and Atmospheric Administration (NOAA) has implemented to protect marine mammals under its authorities have been a "command and control" approach. In general, time and/or area closures reduce or shift fishing effort out of a high bycatch area by prohibiting fishing completely; gear standards reduce the bycatch rate and allow vessels to continue fishing. While closures can be monitored remotely (e.g., electronic vessel monitoring systems) or by patrolling the area, monitoring gear compliance involves hauling gear at-sea for inspection; it can be more labor intensive and thus costly.[1] Consequently, in terms of compliance detection (e.g., monitoring) and cost, a closure may be the preferred policy instrument for the regulator while the individual being regulated may prefer gear modifications since they can continue fishing in the proposed closed area.

In 2007, harbor porpoise bycatch exceeded PBR (U.S. Department of Commerce, 2010) and based on the statutory requirements contained in Section 118 of the MMPA, NMFS was required to take action. Closures and acoustical devices (pingers), a gear standard, were the two primary policy instruments chosen to reduce the harbor porpoise bycatch in the northeast sink gillnet fishery to levels below PBR under the 1999 Harbor Porpoise Take Reduction Plan (HPTRP). Non-compliance with pinger regulations was as high as 65%, from 1999 to 2007, in some regulated areas in the northeast, based on data collected in the Northeast Fisheries Observer Program (NEFOP) (Palka et al., 2009).

Regulators often rely on strict enforcement and penalties to achieve high levels of compliance. An individual will violate a regulation if the expected illegal gain exceeds the penalty, which is a function of the size of the fine for non-compliant behavior and the detection rate of a violation (Becker, 1968). Sutinen and Anderson's (1985) seminal conceptual work on law enforcement was followed with empirical papers confirming Becker's original hypothesis (Sutinen and Gauvin, 1989; Bean, 1990; Sutinen et al., 1990; Furlong, 1991; Kuperan and Sutinen, 1998; Hatcher and Gordon, 2005; Shaw, 2005), demonstrating the economic gain often outweighs the penalty. King and Sutinen's (2010) survey of the northeast United States groundfish fleet indicate the deterrence effect of the existing enforcement system is weak; violations had a 32.5% probability of being detected, and if detected, a 33.1% chance of being prosecuted and resulting in a penalty. Economic gains from violating fishing regulations are nearly five times the economic value of expected penalties. The incentive to not comply is high.

Sink gillnet vessels, members of the northeast groundfish fleet may find it more practical to take the risk of receiving an unintentional first offense of $200 (NOAA, 2014) vs. purchasing pingers; what's more, the maximum statutory penalty for a MMPA violation is equivalent to the initial cost of pingers, $8000 (NMFS, 2009). Thus, the likelihood of a pinger violation leading to an arrest, prosecution and a *fine* is extremely low. However, evidence in various fisheries indicates the majority of fishermen seemed to comply even when the expected illegal gain did exceed the penalty (Kuperan and Sutinen, 1998; Sutinen and Kuperan, 1999). Normative influences may motivate an individual to comply. That is, social norms (obligatory, shared or forbidden behaviors) mediate the way in which people in society behave (Ostrom, 1990, 2000; Wiber et al., 2004). Moral, ethical, legitimacy, and social influences can induce an individual to comply even when the economic incentives for non-compliance are high.

Sutinen and Kuperan (1999) extended Becker's crime model; they developed a theoretical framework which adopts work by Adam Smith (1759) that explicitly portrays human economic motivation as being multidimensional, arguing the psychic well-being is based on acting morally and receiving the approval of others, as well as enhancing wealth. Kuperan and Sutinen's empirical work (1998) found that compliance in a Malaysian fishery depended on the tangible gains and losses, as well as the moral development, legitimacy, and behavior of others in the fishery (Sutinen et al., 1990). Hatcher et al. (2000) made a similar conclusion in regard to fishermen's compliance with quota in the United Kingdom fisheries; a significant positive relationship between perceptions of fairness and levels of compliance was reported though a follow up study confirmed the deterrence effect but found less evidence of normative factors influencing compliance (Hatcher and Gordon, 2005). Similarly, Keane et al. (2008), Nielsen (2003), and Nielsen and Mathiesen (2003), communicated how normative factors (e.g., legitimacy of the imposed regulations) influences individual's compliance decisions while Eggert and Lokina (2008) showed the importance

[1] A dock-side gear inspection program is a lower cost alternative, however, the effectiveness of monitoring compliance may be species dependent; while a vessel may pass a dock-inspection for Turtle Excluder Devices or pingers, that does not enforce proper use at sea. http://www.greateratlantic.fisheries.noaa.gov/protected/porptrp/ptci.html

of normative variables in addition to deterrence variables in explaining compliance behavior of the Tanzanian Lake Victoria fishers. A deterrent, which in practice usually means greater enforcement, is not the only way to improve compliance. Sutinen (2010) argues policy makers should pay more attention to the institutional design to strengthen perceived fairness and legitimacy of the management process.

Normative influences may motivate a fishermen to comply with protected species regulations. The existence of laws and policies such as the MMPA and ESA, imply society values these animals. According to Lavigne et al. (1999), North American attitudes toward marine mammals have in many respects paralleled the evolution of attitudes toward the environment, endangered species and wilderness (Richardson and Loomis, 2009; Wallmo and Lew, 2012). Marine mammals are part of a healthy marine ecosystem and may factor into a fishermen's livelihood. There is an inherent incentive for fishermen to protect their income; fisheries regulations directly impact their day to day earning decision. In a 2012 meeting of fishermen discussing pinger compliance, similar values were echoed: "All I know is in this room there is not a guy in here that wants to hurt a porpoise or whale" (Appendix in Supplementary Material, comment 1). Hence, normative factors may explain compliance decisions with harbor porpoise pinger regulations in the presence of economic incentives to not comply.

We develop a behavioral model which incorporates deterrent (e.g., perception of detection), economic and normative factors (e.g., moral, legitimacy, and social influences) to investigate compliance decisions. Specifically, the compliance behavior of fishermen in the northeast sink gillnet fishery under the 1999 TRP with regard to pinger compliance is examined from 2007 to 2010. Proxy variables are developed from NMFS observer data, NEFOP, to model normative factors. Potential biases with observer data were identified as a concern because forewarned captains may fix problems before the observed trip; however, we are not measuring compliance rates but instead attempting to understand compliance behavior. Our model findings were ground-truthed by conducting focus group research with fishermen using pingers; some preliminary findings are shared in the discussion in support of our model results. The percentage of outcomes correctly predicted is 92% based on model estimates. Our results also suggest observer data such as the NEFOP can be used to support compliance and enforcement mechanisms in this fishery, though this is likely applicable to other fishery compliance problems as well. The intent of this study is to identify the importance of understanding and including normative and economic factors that may influence fishermen's compliance decisions, in order to design effective regulations to protect harbor porpoise.

BACKGROUND

Gillnet Fishery

Sink gillnet gears are used by vessels targeting commercially sought species such as, cod (*Gauds morgue*), spiny dogfish (*Squalls acanthi as*), pollock (*Pollachius virens*), goosefish (*Lophius americanus*), and flounder (*Pleuronectiform*). These vessels operate from Maine to North Carolina. The mix of species landed varies by season and area. In season-areas where groundfish landings, such as cod, pollock, flounder and goosefish are prevalent, dogfish landings are generally absent. Typically, gillnet vessels leave their ports in the early hours of the morning, haul their catches, reset their gears, and return to port the same day. A vessel usually hauls four to eight strings of gear per trip, where one string is around 3000 feet in length. Gear is set in the water to soak for 24–72 h, after which it is hauled and reset. During the long soaking period of gillnets, harbor porpoise (*Phocoena phocoena*) become entangled in the gear and suffocate.

Harbor Porpoise Management

During the last 25 years there have been cycles with harbor porpoise bycatch above or below PBR (Waring et al., 2012). The MMPA indicates that when the 5-year average annual bycatch estimate is greater than PBR (Wade and Angliss, 1997), the following process is initiated to reduce bycatch. First, the stock is designated "Strategic," which requires convening a *Take Reduction Team* (TRT). The TRT has 6 months to develop a plan that will reduce bycatch below PBR within 6 months of implementation of the plan, with a long-term goal of reducing bycatch to an insignificant level approaching zero. The HPTRP implemented the pinger requirement on 1st January 1999 (63 Federal Register 66464, 2 December 1998) after the 1994-1998 average bycatch rate exceeded PBR.

In December 2007, NMFS reconvened the team to consider additional modifications to the HPTRP to reduce harbor porpoise bycatch in New England and Mid-Atlantic gillnet fisheries to levels below the stock's PBR and approaching ZMRG. High non-compliance rates with pinger regulations was one of the reasons bycatch levels exceeded PBR. Enforcement presence was lacking. Since 2012, two pinger violation cases have been prosecuted by NOAA's Office of General Council in the northeast. A $4000 fine was issued in 2014 to a vessel found in "contravention of applicable regulations designed to prevent harbor porpoise from interacting with fishing gear" (NOAA, 2015); and in 2012, a written warning was issued to another vessel for "fishing in the closed offshore area without pingers" (NOAA, 2013).

The focus of this study is the 2010 fishing year (June 2009–May 2010), when gillnet vessels were operating under that 1998 HPTRP plan (NMFS, 1998). According to this plan, vessels could continue fishing if they attach pingers to their gear in the following areas: the Mid-Coast, Mass Bay, Offshore and Cape Cod South Area, north of 40°N (**Figure 1**).

METHODOLOGY

Conceptual Framework

Although pingers regulations are in effect for over a decade, a systematic way to monitor compliance does not exist. Under the current institutional structure, researchers detect and assess pinger violations via NMFS's NEFOP. In general, violations are recorded by NEFOP observers; however, observers do not report

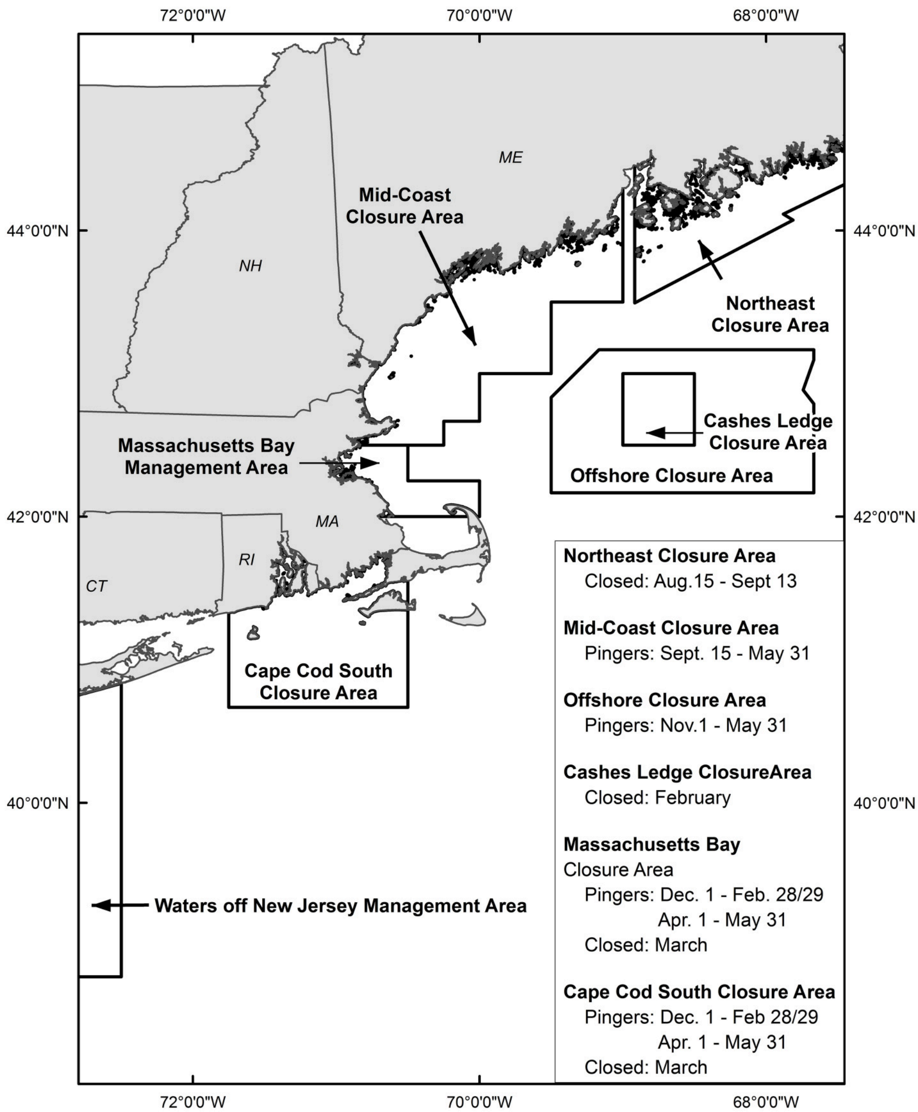

FIGURE 1 | 1998 Harbor porpoise TRT management areas in the northeast (NMFS, 2009).

to enforcement[2]. As a consequence, the likelihood of an observed violation leading to punishment is rare. Moreover, a common belief among fishermen is pingers adversely impact catch and thus revenue; pingers are known to act as dinner bells for harbor seals that eat the warm bellies of cod caught in the gear (Bisack and Clay, 2012). Although there has been no experiment to study the impact of pingers on catch since 1997 (Kraus et al., 1997), the 2007–2010 NMFS observer data show significant differences in cod and pollock catch rates between strings with 100% and zero pingers present [$p = 0.0040$ (equal variance) or $p = 0.0159$ (unequal variance)][3]. An average gillnet trip (8 strings with 10 nets per string soaking for 24 h) fishing with pingers could incur a revenue loss on average of \$1535 per trip (= 1190 pounds less cod and Pollock *\$ 1.29 per pound). With the potential perception that the economic benefit of compliance is lower than non-compliance and a low likelihood of a fine being issued, the economic incentive for non-compliance is assumed to be high. Under this environment, a fisherman's compliance behavior may be explained by normative influences.

We consider three broad types of normative variables: individuals' moral values, social influences and their perceived legitimacy of the regulations. We hypothesize that a fishermen's attitude toward compliance can differ due to differences in their moral standards. An individual's behaviors are often motivated by their personal moral values (Frank, 1996; Nielsen, 2003; Nielsen and Mathiesen, 2003). That is, an individual concerned

[2]NOAA's Office of Law Enforcement (OLE) has requested this information from NMFS, yet only two cases have been prosecuted between 2012 and 2014.

[3]NMFS observed hauls targeting cod (Gadus morhua) and pollock (Pollachius virens) had a mean catch rate of 2.38 pounds of fish per net soak hour ($n = 749$, std = 0.098) while hauls with zero pingers had a mean catch rate of 3.00 pounds of fish per net soak hour ($n = 316$, *std* = 0.238).

with the principles of right and wrong behavior, may feel obligated to obey the law, and thus gain a greater sense of satisfaction by behaving an honorable way.

Social interactions can also influence an individual's attitude toward compliance. A person is likely to be more non-compliant the more his community and peer groups are non-compliant (Vogel, 1974; Geerken and Gove, 1975; Witte and Woodbury, 1985; Robinson and O'Leary-Kelly, 1998; Beams et al., 2003). O'Fallon and Butterfield (2012) explain the occurrence of unethical behavior through three different theories: social learning (i.e., "I behaved unethically because I observed my peers doing it and being rewarded for it"), social identity (i.e., "I behaved unethically because unethical behavior is the social norm"), and social comparison theory ("If I do not engage in unethical behavior, I will fall behind my peers"). There are many reasons a person may be persuaded to make a decision in a particular direction; formal unions are a peer pressure mechanism for example. However, peer pressure may or may not make a difference since it is just one of several factors to consider.

The legitimacy of regulations can also impact an individual's decision to comply. Their perception of the problem and solution can impact their compliance decision; they may question the need for protection and whether the solution works (e.g., whether pingers repel porpoise). The literature on local management or co-management approach to fisheries governance suggests that a greater involvement of fishermen in the management process will lead to increased levels of compliance because regulations will then be accorded greater legitimacy. To be precise, participation by fishers in the management process is considered by many as "essential" for achieving more sustainable, equitable, and efficient management outcomes (Ostrom, 1990; McCay and Jentoft, 1995; Pinto da Silva and Kitts, 2006; Rountree et al., 2008; Yochum et al., 2011). We tend to support solutions with greater satisfaction if we participate in the development of the solution.

Many factors contribute to an individual's personal decision on an issue. The objective of this study is to analyze the influence of these economic and normative factors, in addition to deterrents and a set of vessel characteristics, on an individual's compliance behavior. A formal model of the decision process is given below.

Model Specification

A binary choice modeling framework is used to explain a fisherman's compliance behavior. We assume a fisherman will decide to violate the pinger regulation if their expected utility from non-compliance exceeds the expected utility from compliance. In this scenario, the difference in the expected utilities of the individual is modeled as follows:

$$y_i^* = \beta' x_i + \varepsilon_i$$

Where, x represents a vector of variables that effect a fisherman's compliance decision, β is the vector of unknown parameters and ε_i is the error term. In practice, we do not observe utilities, or y_i^*. What we observe instead is the binary choice variable V_i, which indicates whether a violation has occurred or not. The relationship between y_i^* and V_i can then be defined as follows:

$$\begin{aligned} V_i &= 1 \quad \textit{if } {y_i}^* > 0 \\ V_i &= 0 \quad \textit{otherwise.} \end{aligned}$$

The probability of violation is written as:

$$\begin{aligned} Prob\,(V_i = 1) &= Prob\left(\varepsilon_i > -\beta' x_i\right) \\ &= F(\beta' x_i) \end{aligned}$$

Where F is the cumulative distribution function of ε. If we assume ε is independent and an identically distributed standard normal, we obtain a probit model which can be expressed as:

$$Prob\,(V_i = 1) = \Phi(\beta' x_i)$$

Where, Φ(.) is a standard normal distribution function. The parameters of this binary probit model are estimated via a maximum likelihood method. In a probit model, the estimated coefficients cannot be interpreted as marginal effects; rather they are calculated as follows (Greene, 2000):

$$\frac{\partial E[v|x_i]}{\partial x_i} = \varphi(\beta' x_i)\beta$$

The dependent variable, violation V_i is equal to 1 if vessel i violated the pinger regulations under the 1998 HPTRP plan at least once in the 2010 fishing year (May 2009–April 2010). A gillnet haul was considered in violation of the pinger regulation if the vessel did not have the correct number of pingers attached to the gillnet gear (Palka et al., 2009).

Our independent variable vector x, includes a set of vessel characteristics, deterrence and normative variables. Characteristic variables consisted of a vessel's registered gross tons (*GT*), the ratio between the engine horsepower to vessel length representing the vessel's capital stock (*HPLEN*), the number of years the captain has been fishing with gillnet gear (*CYRS*) and gross revenues (*GREV*) the vessel earned within the last year. We assume the expected fine is less of a deterrent to high earning vessels and test whether the probability of violating pinger regulations is related to high earning revenue vessels. We also examine whether vessels fished gillnet gear exclusively within the last year; vessels may have *less* flexibility to adjust their behavior in response to changes in regulations specific to gillnets if they fish the gear exclusively (GGE = 1), and therefore more likely to comply.

Fishermen that perceive low detection probabilities may consider this factor in their compliance decision. NMFS observer data are used to identify pinger violations in order to assess compliance rates for management. We consider the idea that NMFS observers can be a substitute or complement to enforcement. That is, does the presence of an observer deter non-compliant behavior similar to an enforcement agent? We include a detection variable that captures the vessel's history of being observed over several years to test whether being observed in previous year's influences their compliance in the current year. A person may be compliant whether an observer has been on board or not. Specifically, the detection variable is positive (*DETECT = 1*) if a NMFS observer was aboard the vessel while fishing in pinger areas, at least once in *each* of the previous two fishing years (May 2007–April 2009). Vessels can be observed more than once within a year. However, by adding the additional requirement of sampling two consecutive years for our deterrent variable (DETECT), we test whether consistent annual observer sampling of a vessel influences their compliance behavior.

The normative variables considered take account of both intrinsic and extrinsic values that may influence behavioral outcomes, such as a compliance decision. Our assumption regarding these factors is lower moral values for example, are associated with lower compliance rates while higher values with higher compliance. We construct proxy variables using existing data due to a lack of direct observable data (e.g., interview survey data) for these factors. We assume a vessel's previous violation history captures the decision maker's moral behavior. That is, persons with a history of repeat violations are associated with lower moral values compared to persons with no violations; some individuals follow the law no matter what. The variable recording the vessel's violation history is positive (V_OLD=1), if the vessel has two (2) or more observed pinger violations in the previous 2 years; a violation did not have to occur in consecutive years. We therefore examine whether vessels that have a violation in the current year are more likely to have violated in previous years. Individuals with two or more violations (V_OLD=1) observed in two consecutive years (DETECT=1), may be lackadaisical or casual about regulations, may have low moral standards, but are classified as repeat violators.

Social influences can affect compliance decisions. An individual may feel compelled to not comply with regulations if others are not complying. There were no apparent groupings of sink gillnet vessels fishing with pingers at the time of this study in any particular area. Nonetheless, vessels fishing from the same port of landing are likely to have more opportunities to communicate about prices, regulations etc., compared to vessels fishing in different ports. As a consequence we attempt to understand this factor by including a proxy social variable; we include a "port behavior" variable which indicates whether another vessel in an individual's landing port also had a pinger violation (*PBEHAV = 1*). Specifically, our model tests whether port effects are present; are vessels more likely to not comply if other vessels in their port do not comply as well? Vessels landing in multiple ports were assigned to their highest revenue port.

Our proxy legitimacy variable tests whether a fishermen's involvement in the management process influences compliance with regulations. Specifically, we determine the decision maker's affiliation with a HPTRP team member within their port; members include gillnet fishermen from Maine to Rhode Island, though members are not in every port. A fisherman having direct access to a TRT member may allow information sharing, cooperation, and potential collaboration with the development of the HPTRP. We test whether a fishermen is more likely to comply if they have an active TRT member in their port ($TRT = 1$) *or not* ($TRT = 0$).

Many factors can enter an individual's decision process. We develop proxy normative factors in the absence of a formal compliance survey. Our intent is to investigate alternative normative factors in addition to the expected economic factors that influence a person's decision. This may lead us to consider developing a more formal compliance survey in the future.

DATA

Model Data

Pinger violations, non-compliance, are observed and calculated by using data from the NEFOP, the only available data source to estimate compliance rates. Several data bases are used to build our compliance model data set. The Northeast Commercial Fisheries database and the Northeast Vessel Tracking and Reporting database were used to estimate a vessel's gross revenue (*GREV*) and number of different gears types used within a fishing year (*GGE*). The NMFS Northeast Regional Office's (NERO) Vessel Permit database identifies a vessel's characteristics such as horse power, length and gross tons.

The first step involves identifying all *observed* gillnet vessels fishing in pinger regulated season-areas during our current fishing year, June 2009 through May 2010. The NEFOP observed 52% of the gillnet vessels fishing in areas that require pingers during the current year. Using this unique observed vessel list, we track each vessel from June 2007 to May 2009, two previous fishing years, to calculate a vessel's violation and detection history. Several different databases are accessed over the study period to construct our set of independent variables (**Table 1**) to identify statistically a set of factors that may explain a vessel's compliance behavior in the current year (dependent variable).

During the current fishing year (2009-2010), 248 gillnet vessels took 15,022 trips north of the 40 degree latitude line, earning revenues of $45.6 million dollars. Of these, 107 vessels (43%) fished in areas that required pingers and earned revenues of $8.3 million in pinger managed areas (18% of the total revenue earned by all 248 active gillnet vessels). The NEFOP observed 56 gillnet vessels that had the same operator during the entire study period (2007–2010); this is important because our independent variables include fishing history. We assume the individual making the

TABLE 1 | Description of independent variables.

Variable	Description
CYRS	Number of captain years fishing
HPLEN	Ratio of engine horsepower to vessel length
GREV	Gross revenues of vessel in the previous year (in $1000)
GTONS	Gross tons
DETECT	Perceived probability of detection (observed in each previous 2 years at least once = 1; else = 0)
GGE	Fish gillnet gear exclusively yes = 1; no = 0
V_OLD	Previous violations (at least 2 observed violations in the previous 2 years =1; else = 0)
PBEHAV	Port Behavior of other vessels (yes, others violated = 1; no = 0)
TRT	TRT member belonged to this port? (yes = 1; no = 0)

compliance decision is the vessel operator; therefore our sample consists of vessels that have the same decision maker (operator or captain) over the entire study period. Our data and model results therefore represents 52% (=56/107 vessels) of the fleet fishing with pingers during our study period.

Focus Group Data

Following the completion of this model, researchers held several focus group meetings with the objective of ground-truthing the compliance model results reported in this paper. NOAA Fisheries (NMFS) frequently uses qualitative research such as focus groups and cognitive interviews to facilitate the development of survey instruments. This qualitative research was conducted in facilities that allow observations of the discussion or interview and provide a professional atmosphere for the research. Four focus group sessions with 15 invited gillnet fishermen from Rhode Island to Maine participated (Bisack and Clay, 2012) during the week of 4–8 March 2012. Focus group sessions were facilitated by the researchers. We share some preliminary findings regarding fishermen's perceptions in our discussion section to interpret some of our statistical model findings that follow. Invited fishermen were asked to express their opinions on several normative factors considered here and about regulations in general. Some selected comments by the participants are presented in the appendix. Our model results are based on 2007 to 2010 data and though the focus group meetings were held 2 years after these period, selected comments are robust and independent of the time delay.

RESULTS

During the 2009–2010 fishing year there was at least 1 observed violation on 66% of the vessels (=39/56 vessels) and on 51% of the observed trips in our sample. On average, observed vessels fishing in pinger management areas weighed 21 tons, had a measure of 8 horse power units per vessel foot, earned $228,325 in annual revenues and had captains with 24 years of experience in gillnetting (**Table 2**). Data indicate 79% of the vessels used gillnet gear exclusively. Based on their 2 year history, 48% had an observer on-board their vessel for 2 consecutive years while fishing in pinger areas. Previous violations were present for 55% of the vessels, our moral proxy variable. Our proxy legitimacy and social influence variables indicate 38% of the vessel operators were affiliated with a local TRT member in their port, and 54% resided in a landing port where other vessels had a violation.

Table 3 reports the estimated probit coefficients (estimated with SEs) for the incidence of non-compliance, violations, with pinger regulations among vessels. The log-likelihood test rejects the zero-coefficient hypothesis implying that the model fits the data well ($p < 0.0001$). The percentage of outcomes correctly predicted is 92% based on the model estimates. This suggests an overall good fit of the model. All variables, except the port behavior (*PBEHAVE)* and *TRT*, were significant at 95% level or higher (**Table 3**). Marginal effects were calculated at the individual observation and then averaged over the sample. The marginal effects show a particularly strong influence on VIOL in the estimated model for some variables such as DETECT and GGE.

TABLE 2 | Summary statistics and frequency distribution of the independent variables.

Continuous variables	Mean	Standard deviation	Min	Max
CYRS	24	10	3	45
HPLEN	7.98	2.14	4.6	13.81
GREV ($1000)	228.33	124.23	17.57	644.69
GTONS	20.89	10.15	4.00	65.00
Dummy variables	**Frequency**		**Percent**	
VIOL	39		66.10	
DETECT	27		48.21	
GGE	44		78.57	
V_OLD	31		55.36	
PBEHAV	30		53.57	
TRT	21		37.50	

No. Observations:56.

TABLE 3 | Factors of a vessel's decision to violate pinger regulations.

Variable	Coefficient estimates	Marginal effects*100
INTERCEPT	8.62 (2.46)***	–
CYRS	0.08 (2.36)**	1.22
HPLEN	−1.31 (3.00)***	−18.74
GTONS	0.09 (2.44)**	1.30
DETECT	−2.55 (−2.39)**	−36.43
GREV	0.01 (2.03)**	0.14
GGE	−5.14 (2.02)***	−73.55
V_OLD	3.11 (2.87)***	44.45
PBEHAV	1.42 (1.31)	20.32
TRT	−0.61 (−0.60)	−8.72
Log Likelihood	−14.59	
Zero-slope chi-square (9 df)	42.59 ($p < 0.0001$)	
Percent correctly predicted	92.3%	
No. Observations	56	

*The t-statistics based on SEs are in parentheses. Marginal effects, predicted probabilities, are evaluated at the individual observation and then averaged over the sample. *** and ** indicate significance at 1% and 5% level, respectively. Variance inflation indices and correlation checks indicate multi-collinearity was not present.*

The deterrent factor *DETECT* was inversely related with the probability of a violation, suggesting a higher expectation of being observed will lead to fewer violations. Individuals observed in previous years were on average 36% less likely to violate the pinger regulation. The sign of *GGE* indicates vessels that fish multiple gears, or vessels that do not fish gillnet exclusively, are more likely to violate. The marginal effect for this variable is 74%.

Among the vessel characteristics, those with lower horse power per feet (HPLEN), or under powered vessels, are more likely to violate; this variable has the largest marginal effects among the set of vessel characteristic variables Results also indicate vessels that had more experienced captains (CYRS), were heavier (GTONS) and earned higher revenues (GREV) are more likely to violate. However, the magnitudes of these impacts are low; the marginal effects are close to 1%.

Among the normative variables, vessels with a history of violations, our moral variable (*V_OLD),* have a positive significant relation. Individuals who violated previously are, on average, 45% more likely to violate in the current year. This implies a large number of vessels are, in fact, repeat violators. Vessels are more likely to violate if they did not have a NMFS observer on board in the previous 2 years and they have a history of violations. While the social and legitimacy variables, port behavior (*PBEHAV*) and *TRT*, both have expected signs for their parameter estimates, they are statistically insignificant. The sign for the port behavior (*PBEHAV)* coefficient may suggest the compliance decision of the vessel operator tends to be positively related to the compliance decision of the other port members. The negative sign for the *TRT* coefficient proposes that fishermen's involvement in the development of the TRT plan may lead to lower violations.

In summary, our model estimates suggests, vessels more likely to violate the 1998 TRT harbor porpoise pinger regulations are characterized by lower horse power per foot, higher gross tons, multiple gear use, a positive violation history, and were not carrying a NMFS observer in the previous 2 years while fishing in pinger management areas.

DISCUSSION

Policy planning requires a sound understanding of compliance behavior to achieve successful regulatory goals. Commercial fishing gear standards along with closures are the typical regulatory instruments chosen to reduce the take of protected species such as marine mammals to PBR goals; however, pinger regulations, for example, are successful *only if* there is a high level of compliance. In 2007, non-compliance was one of the primary reasons the TRT reconvened when the porpoise bycatch levels exceeded PBR; compliance was not addressed in the 1999 HPTRP development. NMFS works with various partners, including NOAA's OLE, the U.S. Coast Guard, and individual states to monitor compliance and enforce regulatory components of the HPTRP; this includes coordinating special operations patrols to conduct more focused at-sea monitoring and enforcement of HPTRP requirements (NMFS, 2010). Becker's (1968) basic deterrence framework assumes detection probabilities and fines can be set to improve compliance with regulations; however, requests for more enforcement and higher penalties may not be cost-effective for monitoring pinger gear compliance and though observers record violations in NEFOP, they are not enforcement agents. Subsequently low detection rates can lead to an extremely low probability of being caught and prosecuted; hence, the economic incentive for pinger non-compliance is high. We need to strengthen and expand our compliance framework; HPTRP compliance measures continue to rely primarily on NMFS observer data. Enforcement may not be the only remedy to curb the compliance problem; the observer program may be a substitute or a complement for enforcement. However, our intent in this paper is to understand what factors may influence a fisherman to comply in the absence of incentives.

We follow Sutinen's seminal work along with others and consider normative, economic and perceived detection variables to explain compliance behavior with pinger regulations in the northeast sink gillnet fishery to shed light on other approaches we can pursue to improve compliance with gear standards. Using a probit framework we incorporate economic and normative factors to examine compliance behavior of fishermen with regard to pinger regulations. Results indicate a fisherman who had a history of violations, a low detection rate the previous year, and were characterized as high revenue earners fishing multiple gears were more likely to be non-compliant with pinger regulations. High revenue earners fishing multiple gears may be associated with more capital and hence willing to take more risks with violation consequences.

To ground-truth these model results focus group discussions were held with fishermen using pingers who reside in Connecticut to Maine ports. We weave some preliminary focus group findings about fishermen's perceptions of the normative factors considered in this paper. Participant's views support our model hypothesis and findings. In general, fishermen believed pingers deter harbor porpoise; however, they agreed the economic incentive to comply is absent (Appendix in Supplementary Material, comment 2).

Compliance model results suggest vessels more likely to violate pinger regulations had lower detection rates by NMFS observers. Our deterrent variable DETECT, may indicate the presence of NMFS observers have an influence on compliance decisions. Some focus group participants stated 40% of their 2012 trips were being observed and therefore "non-compliance was not an option." However, they also discussed among themselves who the "bad apples" are and stated the coast guard knows them as well (Appendix in Supplementary Material, comment 3). They went on to share their perception of how these "bad apples" make their decisions; "you can land flounder revenues of "\$4000 and your MMPA fine is \$500, you break the law every day" (Appendix in Supplementary Material, comment 4). Participant's sense or believe the chance of getting caught is low, and if you do get caught, the fines are acceptable.

Violators may often be repeat offenders. We assume a vessel's violation history captures their moral behavior. Our model results show vessels more likely to violate pinger regulations had a history of violations. Fishermen's statements during the focus group meeting echoed King and Sutinen's (2010) findings that most fishermen comply, and within a typical population, there is a small core subgroup that tends to violate routinely (Appendix in Supplementary Material, comment 7–8). Participants talked about "Smart Compliance" in general which recommends different types of enforcement strategies and penalties for different groups of fishermen based on their compliance history (King and Sutinen, 2010); specifically, more aggressive targeting of frequent violators and for certain types of violations, criminal penalties and the forfeiture of all fishing privileges should be considered. Participants recognize the need to increase the penalties.

The presence of a TRT member in a vessel's residing port was not statistically related to their recorded pinger violations. The social science literature asserts we should see improved levels of compliance when individuals have more opportunity to participate in the design and discussion of regulations. We suggested a fishermen's involvement in the development of the HPTRP via a TRT member residing in their port, may lead to lower violations. However, the statistically insignificant finding is consistent with focus group participants' comments. Frankly, only a third of the participants knew who their TRT representative was and some of these participants had that knowledge because they in fact, were members of the 2007 harbor porpoise TRT and participated at TRT meetings. Meetings are infrequent; the TRT met in 1998 and then nine (9) years later in 2007 when bycatch levels exceeded PBR. An increase in face-to-face communication could improve compliance behavior.

The proxy social (*PBEHAV*) variable was not statistically significant; we tested whether other vessels in the same port had violations or not. Focus group participants stated fairly strongly, that their decision to comply is not influenced by other's behavior (*PBEHAV*) (Appendix in Supplementary Material, comment 6). Why would we be expected to know other people's behavior? It was clear fishermen may have an impractical assessment of their peer's behavior. A participant made the following comment when asked whether they know who is and is not complying with the pinger regulations: "So I mean our gillnet fleet *I think* is, (long pause), I know he's a complier (pointing to another participant)" (Appendix in Supplementary Material, comment 5). The response was not surprising. Gillnet vessels reside in approximately 22 different ports along a large New England coastline from Maine to Connecticut; they describe their day-to-day fishing operations as a somewhat solitary existence. Given that fishermen are in short supply of face-to-face TRT meetings to discuss MMPA regulations and have a limited awareness of their peer's compliance; these environmental conditions may possibly provide an explanation of the insignificant finding for our legitimacy and social proxy variables.

Models and data in general are not flawless; we do not have perfect information and consequently, shortcomings and potential biases exist. We followed Hatcher's et al. (2000) compliance model with some adaptations. First, though a penalty structure was present in the sense that MMPA fines exist, only one recorded pinger violation has been prosecuted with a resulting fine. For that reason we could not investigate Becker's original crime model relationships; that is, empirically estimate whether the expected illegal gain exceeds the penalty. Second, while Hatcher et al. (2000) relied on face-to-face interview survey data to investigate normative factors, our model relies on historical data recorded by NMFS observers. Our model data are based on recorded observations vs. an individual's perception of their history. Using both data types, interview surveys and NMFS observed data, may improve our ability to understand compliance behavior. For example, comparing differences between an individual's "actual" vs. "perceived" history of violations may uncover whether an individual's awareness of their own compliance behavior is accurate. Third, while the non-significance of the social and legitimacy proxy variables to some extent was expected, including these variables sheds light on the importance these factors can have on compliance decisions. In contrast, our *moral* variable was significant. While anecdotal, the hot topic with focus groups participants was "repeat violators"; everyone knows who the repeat violators are including enforcement.

Responses from focus group participant seem to authenticate our normative variable findings and these variables remain in our study with a long term goal of improving these data in future research. Finally, the appropriateness of using observer data was raised; there is a perception that vessels may be forewarned and repair broken pingers prior to a NMFS observer boarding a vessel for official data recording. If this were the case, the observed violation rate would be negatively biased; however, we are researching factors that may influence compliance decision and not the compliance rate itself.

Our research findings will hopefully provide resource managers some valuable knowledge and insights to include while developing regulations. Observers could simply inform vessel owners that OLE does access their records. Thus, NMFS's observer program can complement or supplement enforcement. With that mind, increasing or balancing observer coverage in low sampling areas could result in high compliance returns; under sampling can induce non-compliant behavior. Alternatively, only vessels fishing gillnets exclusively be allowed to fish in pinger areas, was suggested by an anonymous reviewer. In addition, increasing observer presence which collects multi-disciplined research data simultaneously is likely more cost-effective than increasing enforcement levels in-order to conduct at-sea gear compliance checks for a single species. Second, profile and target repeat violators for compliance inspections. This may induce a sense of fairness among fishermen which may also lead to improved compliance. While these findings are not a surprise, the validation thru a formal model may provide enough scientific support to turn these recommendations into management actions.

Consequential closures, entire fishing areas would be closed for several months and years, threatening a vessel's livelihood if non-compliance exceeded a benchmark porpoise bycatch rate for two consecutive years in pinger management areas (75 Federal Register 7383, 19 February 2010). This incentive in the form of a "threat" was not implemented during this study (2007–2010) but immediately after in May 2010. Approximately half of the gillnet fleet started operating under sector management in the northeast groundfish fishery in May 2010 simultaneously. Future research will investigate pinger compliance under a new incentive structure, consequential closures and sector management. We anticipate these new data, along with additional focus group socio-economic research data, will enrich our model. Our results are not conclusive but deserve more attention. We anticipate this research can help us understand the internal motivation embedded in the compliance decision of the individual being regulated, ultimately leading to more successful regulations.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fmars.2015.00091

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Measuring Management Success for Protected Species: Looking beyond Biological Outcomes

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The complexity of the ocean ecosystem, including the human component, is such that a single fishery may require multiple policy instruments to support recovery and conservation of protected species, in addition to those for fisheries management. As regulations multiply, the need for retrospective analysis and evaluation grows in order to inform future policy. To accurately evaluate policy instruments, clear objectives and their link to outcomes are necessary, as well as identifying criteria to evaluate outcomes. The Northeast United States sink gillnet groundfish fishery provides a case study of the complexity of regulations and policy instruments implemented under the Marine Mammal Protection Act (MMPA) and the Endangered Species Act (ESA) to address bycatch of marine mammals. The case study illustrates a range of possible objectives for the policy instruments including *biological, economic, social-normative,* and *longevity* factors. We highlight links between possible objectives, outcomes and criteria for the four factors, as well as areas for consideration when undertaking ex-post analyses. To support learning from past actions, we call for a coordinated effort involving multiple disciplines and jurisdictions to undertake retrospective analyses and evaluations of key groups of policy instruments used for protected species.

Keywords: policy instruments, marine mammals, bycatch, social norms, compliance, retrospective analysis, ex-post analysis

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The complexity of the ocean ecosystem, including the human component, is such that a single fishery may require multiple policy instruments to support recovery and conservation of protected species. Many policy instruments are assessed prior to implementation (i.e., prospective or ex-ante analysis) when we have limited information; however, we seldom go back to undertake evaluation after implementation (i.e., retrospective or ex-post) when we have more information (Greenstone, 2009). As regulations multiply, the need for ex-post analysis grows, as it allows us to identify what works and what does not. After 20 years of regulating under the Marine Mammal Protection Act (MMPA) and the Endangered Species Act (ESA) in the United States (US), regional Protected Resources (PR) leaders for National Oceanic and Atmospheric Administration (NOAA) have voiced their desire to learn how well the policy instruments in place are working, as well as how accurate our estimates of impacts made prior to implementation (ex-ante) are compared to actual economic and biological outcomes (ex-post) (Bisack et al., 2015). In order to undertake such instrument evaluation, evaluation criteria based on measurable outcomes must be identified (Rossi et al., 2004), which in turn are defined by the objectives of the instrument.

Traditionally the performance of protected species management has been measured using biological criteria as proxies for the larger policy objectives and outcomes. For example, under the MMPA the biological objective is to conserve marine mammals as significant functional elements of marine ecosystems, which is primarily undertaken with moratoriums on their direct take. The 1994 potential biological removal (PBR) control rule under the MMPA sets the criteria for how much bycatch is allowed. Yet policy instruments for protected species recovery generally have multiple objectives, suggesting the need for multiple criteria or measures of performance outcomes. Proposed regulations for policy instruments must meet economic and social objectives; evaluation criteria are necessary for these objectives as well. For example, a regulation must ensure that national benefits exceed costs [i.e., under Executive Order (EO) 12866 in the US or the Cabinet Directive on Regulatory Management (CDRM) in Canada] and consider distributional impacts [e.g., among small businesses, minority groups and/or low-income populations under EO 12898, the Regulatory Flexibility Act (RFA) and CDRM]. A regulation may also be required to illustrate that future compliance, monitoring, and enforcement costs have been considered (e.g., under the CDRM), although even when this occurs motivation and incentives to comply are seldom addressed.

We advocate for a coordinated effort involving multiple disciplines and jurisdictions to develop an evaluation strategy for protected species policy instruments. Further, we advocate for the use of multiple evaluation criteria based on *biological, economic, social-normative,* and *longevity* objectives and outcomes, to name a few. The biological and economic efficiency objectives may be more recognizable, and potentially easier to attain, than the distributional concerns of participants in a fishery, which may be captured in social-normative objectives. Instruments that explicitly consider social-normative factors may be better situated to address the distributional issues (e.g., access/exclusion from fishing opportunities), issues which can delay or impede implementation. Since the design and implementation of policy instruments is costly, it may be desirable to include design features that extend the useful life of an instrument by allowing it to adapt to a changing environment (i.e., longevity). With this group of factors in mind, we use the Northeast United States (NE US) sink gillnet groundfish fishery as a case study to illustrate considerations when identifying evaluation criteria. While we recognize that the success of a policy instrument in achieving its objectives may be, in part, unique to the setting, we believe that assessing the strengths and weaknesses of a range of policy instruments is essential to developing successful plans for protection of species in the future.

The NE US sink gillnet groundfish fishery has been regulated under multiple legislative authorities for over 20 years. The MMPA provides the authority to address bycatch of marine mammals such as harbor porpoise in commercial fisheries, while the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) provides the authority to manage commercially fished species in US waters. During the early 1990's, high harbor porpoise mortalities motivated innovative cooperation between industry, scientists and government which resulted in the development of acoustical devices (pingers) that attach to gillnets to deter porpoise interactions (Kraus et al., 1997). The first Take Reduction Plan (TRP) under the MMPA combined pinger requirements and gillnet gear closures to protect harbor porpoise (National Marine Fisheries Service (NMFS), 1998), with monthly rolling closures under the Northeast Multispecies Fishery Management Plan (FMP) (63 Federal Register 66464, December 2, 1998). By 2000, harbor porpoise mortalities/takes were below the PBR level (Waring et al., 2002). However, the reduction in takes under the plan was temporary (Waring et al., 2006), even though restrictions on effort in the commercial groundfish fishery were ongoing to achieve stock rebuilding goals.

By 2004, Days-At-Sea (DAS), initially established in 1994 to limit the number of days a vessel owner could fish, had dropped between 67 and 100% for any given vessel and catch trip limits had tightened (New England Fisheries Management Council (NEFMC), 2006). In addition, a sector allocation program (similar to a harvest cooperative) was introduced, which allocated a share of a groundfish stock to a group of vessel owners that voluntarily joined a sector group. Only one sector formed, the Cape Cod Hook Sector, which was allocated a share of Georges Bank cod. In response to overfishing of several stocks, including Gulf of Maine cod, a 2006 emergency rule implemented differential DAS counting (National Marine Fisheries Service (NMFS), 2006), and NMFS approved a second voluntary sector which was a gillnet gear sector (71 Federal Register 48903, August 22, 2006). In 2010, a revised voluntary sector allocation program was implemented for the entire groundfish fishery. Vessels that did not join a sector fished under the effort controls (DAS) and an Annual Catch Limit (ACL) for all the vessels in the "common pool" (75 Federal Register 18356, April 9, 2010). About 55% of the northeast gillnet vessels joined one of seventeen initial sectors. At about the same time pinger non-compliance was identified as a major source of high bycatch of harbor porpoise, and a revised TRP was implemented. The TRP increased, spatially and temporally, the areas that required pingers to fish, and created an incentive for pinger compliance in the form of a threat—indefinite closures over a large area if compliance remained below defined levels (National Marine Fisheries Service (NMFS), 2009). Industry agreed to the plan that would largely rest on individual responsibility for compliance (i.e., self-policing).

The choice of a policy instrument may influence the objectives that can be considered during design, and consequently during evaluation. As illustrated with the harbor porpoise example, most policy instruments NOAA has implemented for marine protected species under its authorities have used a "command and control" (C&C) approach directed toward fishermen (also see National Marine Fisheries Service (NMFS), 1998; National Oceanic Atmospheric Administration (NOAA), 2006, 2009, 2012). Policy instruments under the C&C approach include controls on inputs (fishing effort, DAS) and outputs (catch, ACLs), as well as technical standards (gear modifications, pingers). Under the C&C approach, the governing agency requires individuals to undertake specific activities to meet specific standards to achieve a specific objective; this approach can limit the ability of individuals to achieve economically efficient outcomes.

In general, the more specific is the requirement, the fewer opportunities exist for individuals to modify their behavior or processes for economic efficiency. The specificity of C&C instruments may encourage the use of sunset clauses, to address concerns of cost and effectiveness. The objectives of a C&C policy instrument for protected species tends to be narrowly focused on a biological outcome, with economic considerations focused on a least-cost or cost-effective objective. While other factors may be considered during policy development, they are seldom explicit.

Economists have long supported policy instruments where market signals create incentives for desired behavioral changes; this may require consideration of additional objectives such as social-normative objectives. Incentives can be classified as positive ("carrots"), such as property rights, or negative ("sticks"), such as taxes, fines or sanctions. In the harbor porpoise case study, the threat of an indefinite closure if pinger compliance did not meet a target was a "stick." Generally positive rewards are preferred to negative punishments, given political and user difficulties with imposing and enforcing sanctions (Polasky and Segerson, 2009). Market-based instruments allow individuals to voluntarily choose how to meet an objective, with prices and other economic variables providing signals to reduce or eliminate negative externalities (e.g., harbor porpoise bycatch). This flexibility may allow the instrument to adapt to changes in economic or biological environments. Market-based instruments, explicitly or implicitly, establish some degree of property right characteristics (exclusivity, divisibility, transferability, duration, and enforcement), that allow for better planning by users, owners and managers. There is a growing literature on the implications of various market-based instruments in fisheries management (e.g., Pascoe et al., 2010; Squires et al., 2013; Innes et al., 2015). Yet, even with these approaches, some forms of technical standards or controls are typically retained to support or complement market measures, further supporting the need for evaluation of C&C instruments.

Theoretical and empirical analyses of policy instruments for protected species have largely focused on biological and economic outcomes. However, objectives based on social norms (e.g., fairness) may also be implicit in an instrument, and an understanding of those norms is important to successful implementation of either C&C or market-based instruments. Social norms include the unwritten, yet mutually understood rules that govern acceptable behaviors and coordinate interactions with others within a society. Human societies use norms of acceptable behavior among their members with the threat of punishment encouraging compliance. There is generally a range within which acceptable behaviors fall, but also a consensus as to when behavior falls within and outside the range of "acceptable skirting" of the rules (e.g., Toner et al., 2014). Misperceptions in group norms, as well as perceptions of a lack of adherence to norms such as fairness, can result in the creation of a new social norm that may run counter to the intensions of the policy instrument; non-compliance may be a potential outcome. Investing in stakeholder meetings during development of a new policy instrument is an approach to understand customary rules of behavior and factors of importance, as well as provide a baseline of existing norms. At times, minor changes in regulations can eliminate small incentives for non-compliance, nudging the average fisherman toward compliance. While details on methods to identify norms go beyond the scope of this paper, non-compliance may be a signal that the norms implicitly assumed by the designers of the policy instrument do not align well with those of the community the instrument impacts.

Few evaluations of protected species policy instruments have been undertaken. A coordinated approach to analysis may create synergies, although such an approach will require agreement on a number of factors such as identification of baselines and evaluation criteria. A few considerations for such an approach are examined below; in particular, we suggest four general criteria as the initial focus. **Table 1** uses examples from the case study to illustrate potential means to identify and measure the proposed criteria. For retrospective analyses and evaluations to be useful the objectives of a policy instrument must be clearly linked to its outcomes or results. As well there needs to be a way to determine if the change in outcome was due to the instrument or other forces. This is done using a baseline which describes what would have happened if the policy instrument had not been implemented. Simulation is frequently used to develop a baseline for retrospective biological and economic analyses. Alternatively, experimental or quasi-experimental design may be used to identify the outcomes of similar situations where the policy instrument was not implemented; these may be called counterfactuals (Greenstone, 2009). Experimental-based counterfactuals for protected species may be difficult to identify due to their imperiled state or legislated requirements; however, alternative locations or jurisdictions and species may provide relevant examples.

The *biological* objective of most actions directed toward protected species is conservation; however, the criteria to evaluate biological objectives may vary depending on the population status and condition of the species such as endangered, threatened (ESA, Species at Risk Act of 2002) or depleted (MMPA). That is, criteria to measure the success in meeting the biological objective may relate to bycatch (incidental take), abundance, distribution, or the probability of extinction of a species. Often determining the biological objective does not automatically translate into measurable criteria to evaluate the outcome. Fisheries observer data have provided fertile ground for ex-ante analysis, prior to implementation of the instrument. There are enough direct interactions observed for species such as harbor porpoise (**Table 1**), loggerhead sea turtles and bottlenose dolphins for ex-ante analyses to attain predictive statistical power (National Marine Fisheries Service (NMFS), 2009), suggesting sufficient data may also exist for ex-post analyses. In contrast, species with limited observed interactions such as the North Atlantic Right Whales (NARW) require non-standard approaches. There are no direct interactions recorded by observers of NARW bycatch in the gillnet fishery; rather, mortality, along with a cause determination, is typically determined post-mortem after carcass recovery. Thus, for a species such as the NARW, performing ex-ante analysis on the implications for a regulation to achieve a conservation objective

TABLE 1 | Examples of evaluation objectives and criteria based on harbor porpoise case study of command and control instruments of pingers and closures.

Objective class	Policy instruments evaluated	Act	Evaluation criteria	Identified during instrument design?	Evaluation criteria used	Implementation strengths/shortfalls
Biological	Pingers and Closures	MMPA	PBR	Yes	Reduce bycatch below PBR	Northeast Fisheries Observer Program data primary and fertile source for ex-ante and ex-post analysis.
Economic	Pingers and Closures	E.O.12866	Net National Benefits (NNB)	Yes	Cost-effectiveness-Analysis (CEA): vessel profit losses per harbor porpoise saved[a]	Benefits not assessed—CEA second best to NNB[b]. Difficult to untangle impacts of non-MMPA and MMPA policy instruments.
Social-Normative	Pingers	RFA, MSFCMA National Standards (NS)	RFA - SBA impacts on small businesses; NS 4[c], 8, 10 - No discrimination among states, consider community impacts, vessel safety	Yes	RFA and NSs: Profit loss and percent impacted by SBA	Stakeholder buy-in not assessed[d]. Self-policing impractical—instrument fails to meet PBR due to non-compliance. NEFOP data—observer effect could bias compliance assessments in ex-post analysis.
Longevity	Pingers	MMPA	New TRP when bycatch exceeds PBR	Somewhat	Frequency of change in TRP (1998 and 2010)	*Ad-hoc* review for MMPA policy instruments vs. periodic appraisal.

While the TRT consists of a team of stakeholders that work collectively to develop conservation measures, it is the fishermen who are asked to comply; focus group participants were unaware of who their TRT representatives are suggesting communication and buy in to comply is unknown.

[a]*The cost of saving one harbor porpoise using closures exclusively vs. pingers exclusively was \$3398 vs. \$583, respectively (National Marine Fisheries Service (NMFS), 2009).*

[b]*See Bisack et al. (2015) for discussion by NMFS economists on pros and cons of CBA and CEA.*

[c]*Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA) requires U.S. fisheries to adhere to 10 National Standards (NSs) (16 U.S.C. 1851 §301), including many directly related to social and economic outcomes: NS4, do not discriminate between residents of different states; NS5, consider economic efficiency; NS7, minimize costs; NS8, take into account the importance of fishery resources to fishing communities; and NS10, promote safety at sea.*

[d]*Stakeholder meetings are a strategic way to derive usability objectives from business objectives, and to gain commitment to usability. They also collects information about the purpose of the system and its overall context of use. In contrast, scoping meetings are held to give the public an opportunity to get involved in the development of conservation measures (e.g., proposed policy instruments such as closures or pingers).*

may be challenging. The data however, may be sufficient for the development of ex-post evaluation criteria. For example, to assess the effectiveness of regulations, Pace et al. (2014) developed a novel method that relied on opportunistic entanglement data from 1999 through 2009. The study determined gear modifications as outlined in the Large Whale TRP did not result in a detectable decrease in waiting time (the number of days) between entanglement events. Thus, they concluded management measures implemented during the study period to reduce large whale mortalities were generally ineffective in abating whale deaths from fishing gear entanglements; hence, more action was required. It is worth noting, human behavior was not included in this model. Perhaps a multi-disciplinary approach would have identified the source of the failure and potential solutions.

The *economic* objectives used to inform the selection of a policy instrument for protected species, unlike the biological objectives, are seldom articulated during the development phase of the instrument. However, most developed nations require some sort of ex-ante cost-benefit analysis to support regulatory proposals, although allowance for cost-effective analysis may exist in some guidance documents (e.g., Treasury Board of Canada Secretariat (TBS), 2007). For example, in the US, Executive Order 12866 requires an evaluation of costs and benefits of regulatory proposals to US society and a determination of net benefits to the Nation (net national benefits). In Canada, the CDRM requires an evaluation of social and economic impacts, and directs authors of a regulation to identify the "instrument that maximizes net benefits for [Canadian] society." Most economic analyses for protected species are ex-ante analyses, and economic measures of benefits are often not available. In such cases, net benefits analysis may be replaced with cost-effectiveness analysis, such as the cost of saving a porpoise estimated in the 2010 TRT plan (**Table 1**; National Marine Fisheries Service (NMFS), 2009). Instruments, however, may not always require regulation; instrument actions may be voluntary or negotiated between parties (Segerson, 2010). In such cases ex-ante analysis may not be undertaken and retrospective analysis may be more challenging. Examples of ex-post economic analyses are relatively rare and focus on regulatory change. While not specific to protected species, Lee and Thunberg (2013) showed the benefit of moving to catch shares by evaluating the additional cost if the US Northeast groundfish fishery had instead remained under DAS. In that scenario, the US society would have been $33 million worse off ($25 million in consumer surplus and $7.5 million in producer surplus). Squires showed a $75 million loss in US consumer surplus as a result of increased sea turtle bycatch in foreign waters following driftnet fishery area closures to protect sea turtles in the US (Bisack et al., 2015).

Explicit incorporation of *social-normative* objectives in policy instrument development is rare, and yet these factors may have a significant impact on the implementation and outcomes of policy instruments (Revesz and Stavins, 2007). Both norms surrounding compliance and level of participation in the creation of regulations are important determinants of eventual compliance behavior (e.g., Dalton, 2005a,b; Pomeroy and Douvere, 2008). Not considering these factors can reduce compliance and result in unmet goals and objectives. Social pressure (community), perceived legitimacy, fairness, and morals (stewardship) are all examples of normative factors. The case study of the gillnet fleet and high non-compliance with pinger regulations illustrates the importance of social-normative factors (**Table 1**).

The fundamental premises of consequential closures in the 2007 TRP were: (1) as a result of the threat, non-compliance with pinger requirements would decrease and ensure bycatch rates would not exceed benchmark limits; and, (2) the threat of indefinite seasonal closures would encourage fishermen to enforce compliance with pinger requirements among their communities (i.e., self-police). However, successful "self-policing" requires a small group or community that conducts activities in a confined setting with members that have face-to-face contact (Dietz et al., 2003). Northeast sink gillnet vessels reside in ∼22 different ports on the long New England coastline from Maine to Connecticut, making face-to-face contact problematic. During focus groups, sink gillnet fishermen who are members of groundfish sector groups self-report that they have a high level of compliance with pinger requirements (Bisack and Clay, 2012). Sectors are typically limited to a small number of members and for gillnet, the negotiated contract identifies pinger violations as one cause for expulsion. Focus group participants provided insights into pinger non-compliance including: they knew who the "violators" were in their (local) communities, saw punishment as non-existent (lack of fairness), and, while they believed pingers deter porpoise (legitimacy of the solution), they also believed the stock was healthy and therefore management was unnecessary (legitimacy of problem). Work such as this may provide a framework for future stakeholder meetings to gather information on social norms when developing new policy instruments. This information may improve understanding of potential outcomes and assist with retrospective analyses, as well as support the development of methods and systems to gather baseline information on norms or identifying counterfactuals for retrospective analysis.

Lastly, one objective of instrument design seldom discussed is *longevity*, which considers whether the instrument is able to continue to achieve the intended outcomes over time, given changes in human behavior and environmental conditions. That is, given the biological, economic, and social-normative factors associated with the instrument, how long should we expect that instrument to continue to meet the purpose and need for the policy? The RFA requires a periodic review of some regulations to consider this question, while one of the benefits of market-based policy instruments is their ability to allow participants to respond to changing conditions. Diametrically opposed to longevity are sunset clauses, which are often added simply as a means to get disparate groups to agree to a policy. While such clauses may purport to be concerned with outcomes and effectiveness, their actual timing may occur before results are anticipated and may not include measures to evaluate effectiveness, an issue for data-poor and long-lived species such as NARW (78 FR 73726, December 9, 2013). Synergistic and cumulative impacts with other management actions are likely to have an impact on the longevity of a policy instrument, and need to be considered as well. For instance, under the MMPA closures may coincide with changes in effort on commercial stocks targeted by sink gillnet vessels. Achieving the MMPA biological PBR criteria is feasible when there is no change in the fish management actions that suppress effort. However, if fishing effort increases as fish allocations increase, the objectives of the closure may be defeated as takes of PR increase in the open areas. Instrument effectiveness may also decline due to biological factors. For example, concerns have been raised regarding the potential for harbor porpoise to habituate to pingers. While one field experiment found porpoises in the Bay of Fundy habituated to a specific pinger and were not alerted to echolocate by pingers (Cox et al., 2001), alternative analysis using interaction data from the NEFOP concluded there did not appear to be habituation (Palka et al., 2008). In general, concerns about changes in biological or environmental conditions are addressed by reactively adding additional instruments onto existing measures. While sunset clauses and retirement plans should be considered during design, possible evaluation criteria include measures of the frequency of modifications or additions to the instrument (**Table 1**).

The need for retrospective analysis of individual policy instruments and evaluation across instruments and settings, for marine protected species is clear, but the way forward is less so. The management of marine fisheries with protected species interactions is set within a complex system. Ecosystem based management (EBM) can provide a natural bridge between single species assessments and management. However, current EBM models are frequently missing the economic and social components, which would consider interactions between ecological and human systems. Retrospective analysis and evaluation can guide us. We need to identify a common language for a multi-disciplinary approach and select a small number of data rich examples for an initial analysis and evaluation. We encourage looking beyond national borders for potential counterfactuals, increasing data collection on non-biological factors for baseline development and suggest further consideration for quasi-experimental design opportunities. The information gleaned from retrospective analysis and evaluations can help identify the key factors to consider when choosing an instrument (e.g., biological, economic, social-normative, and longevity). The goal is more effective use of policy instruments from all perspectives.

AUTHOR CONTRIBUTIONS

All authors listed, have made substantial, direct and intellectual contribution to the work, and approved it for publication.

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Uncertainty, Irreversibility and the Optimal Timing of Large-Scale Investments in Protected Species Habitat Restoration

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We present a model of the optimal timing of a large-scale habitat restoration project. The model is a dynamic benefit optimization that includes ecosystem costs caused by the presence of a large dam. We use a single stochastic variable to incorporate two sources of uncertainty: uncertainty about how ecosystem costs will evolve over time and the possibility of the ecosystem jumping to an undesirable state. We use our model to illustrate two main results. First, variability in ecosystem costs creates an incentive to delay a project intended to restore ecosystem health. The uncertainty regarding ecosystem costs creates an option value to waiting to invest in restoration at a later date. Second, the possibility of jumping to an irreversible and unacceptably bad ecosystem state (such as species extinction) creates an incentive to hasten restoration. These results formalize the countervailing incentives faced by policy makers when multiple uncertainties and irreversibilities are present in managed ecosystems.

Keywords: habitat restoration, uncertainty, irreversibility, real options, extinction risk

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INTRODUCTION

Investments in ecosystem restoration projects are often subject to economic analyses to help determine which investments to make. Decisions on restoring critical habitat for protected species are complicated by the facts that costs are largely irretrievable and that delay in undertaking restoration can lead to further irreversible ecosystem damage (e.g., species extinction). When considering large-scale habitat restoration or species recovery projects, decision-makers typically face a variety of uncertainties such as current and future ecosystem conditions, the costs and efficacy of restoration efforts, and the presence of tipping points that must be weighed in the decision making process. Uncertainty regarding restoration efficacy, creating sunk restoration costs, and waiting for new information are valid reasons for delaying restoration. Conversely, the possibility ongoing ecosystem degradation that leads to higher restoration costs or irreversible system damage may lead decision-makers to move forward on restoration projects. These countervailing incentives are pervasive in restoration planning; hence there is a need for decision makers to be able to understand both reasons to hasten restoration efforts and reasons to slow them down within a single framework.

In this article we construct a model that highlights how uncertainty and the irreversible nature of many protected resource recovery investments create countervailing incentives that complicate decisions regarding whether and when to undertake an expensive project. Importantly, our model captures the tensions in such a decision: delaying action postpones costly expenditures and allows one to wait for more information, but delay also carries the risk of serious consequences such as species extinction. We formulate a continuous-time, continuous-state optimal stopping model of the decision of when to remove a dam and restore the ecosystem. We solve the model using stochastic dynamic programming methods. These types of models are prevalent in the "real options" literature (see Dixit and Pindyck, 1994) and in the investment management literature for pricing options.

One good example of a large investment in ecosystem restoration when uncertainty is present is the decision regarding when (and whether) to remove a large dam to improve habitat conditions for anadromous fish. This problem exists, for example, when undertaking recovery action for protected Evolutionarily Significant Units of Pacific salmon species on the west coast of North America. Dams create a number of problems for aquatic ecosystems (Ligon et al., 1995; Bednarek, 2001; Bunn and Arthington, 2002; Pringle, 2003; Beechie et al., 2006; Pess et al., 2008). Dams block access to upstream habitat thus often greatly reducing the carrying capacity for anadromous fish. Dams also disrupt natural hydrologic regimes, which can lead to degradation of water quality, lack of nutrients, and adverse changes in stream geomorphology. We use the example of an expensive dam removal project to motivate our model, but other similar examples are possible, such as the purchase of a large plot of land that is uniquely important habitat for protected species or constructing a new wastewater treatment plant to improve water quality (Connon et al., 2011; Parker et al., 2012; Medellín-Azuara et al., 2013).

A key feature of ecosystem recovery decisions is that there are risks involved. For example, fish population viability is affected by human-induced stressors that can be alleviated through mitigation or restoration measures. In our case, a dam may block access to spawning habitat and alter hydrologic regimes. Fish populations, however, are also subject to random fluctuations caused by stochastic environmental factors. Therefore, we do not know with certainty *ex ante* whether fish populations will maintain their current levels, increase, or decline even after the dam is removed. The path that future ecosystem costs will take is uncertain. At any time, there is a risk that populations of concern could drop below a threshold leading to extinction. In fact, results from the ecology literature indicate that small populations are at greater risk of crossing these thresholds because their small size increases the relative variability in the population (Lande, 1993; McElhany et al., 2000). In a decision model context, reaching a point of extinction can be thought of as an extreme high-cost state.

Real options analysis is an attractive framework for analyzing large-scale ecosystem restoration projects because it captures three important features. First, restoration is costly and can be irreversible. For example, the cost of a restoration project on the Elwha River in Washington, including removal of two large dams, was estimated to be $324.7 million (National Park Service, 2005). Second, there is significant uncertainty regarding future ecosystem costs if major restoration is not undertaken. For example, the evolution of fish populations is uncertain and affected by restoration or the lack thereof. Declining fish populations and possible extinction create societal costs. Third, damage to ecosystems, such as species extinction, can become irreversible if restoration is delayed for too long. For example, one recent article extrapolates current trends in fish population dynamics and concludes that 9 out of 21 anadromous salmonid taxa in California are "in danger of extinction in the near future" (Katz et al., 2013). The authors note that large-scale habitat restoration projects such as dam removal will become increasingly important in the face of warming temperatures and more variable rainfall.

Economic analysis that evaluates expenditures on protected resources conservation can help decide whether, where, and how much to invest. Previous studies constructed models to solve for cost-effective allocation of limited resources for habitat restoration (Duke et al., 2013). One example is work that chooses the optimal spatial allocation of riparian habitat restoration to help in the recovery of protected steelhead trout (Wu et al., 2000; Wu and Skelton-Groth, 2002). Benefit-cost analysis, which provides a test of whether projects are likely to improve social welfare, is often used and cited in decisions regarding whether or not to undertake ecosystem restoration projects (Pearce, 1998; Hanley, 2001; Hammitt, 2013). However, traditional benefit-cost analysis misses important aspects of the investment decision when planning for large, irreversible investments in cases where protected species are at risk of extinction. Verbruggen (2013) discusses the concept of irreversibility and its importance in decision-making. Verbruggen (2013) also highlights some limitations of traditional cost-benefit analysis. Our modeling efforts address some of these concerns by directly incorporating irreversibility (both in terms of sunk investment costs and irreparable harm to ecosystems), uncertainty (in terms of stochastically evolving biological resources), and timing issues.

The decision of when to restore an ecosystem is also similar to the question of when to invest in expensive pollution control. Pindyck (2000, 2002) considers a model of when to invest in pollution control, focusing on irreversibility and uncertainty. By investing immediately in pollution control, Pindyck notes that society pays sunk costs in pollution control investments (for example scrubbers on coal plants). These types of sunk costs (i.e., irreversible investments) lead one to favor delaying investment. Conversely, potential permanent environmental damages that may be very costly or impossible to reverse (e.g., permanent temperature changes from greenhouse gas emissions) lead one to hasten the decision. Pindyck characterizes these types of damages as sunk benefits. These two types of irreversibilities, sunk benefits and sunk costs, are countervailing—one hastens the decisions to act and the other delays the decision to act.

Some previous work uses a real options framework to examine natural resource management problems when uncertainty and irreversibility exist. Saphores and Shogren (2005) formulate a model of invasive species and pest control. In this model, damages from pests are irreversible and there is uncertainty in how fast the pest population will grow. Saphores and Shogren's (2005) model is interesting in our context because invasive species control is an ecosystem improvement investment. Conrad (2000) presents a model of when to develop (or extract resources from) a wilderness area. The social value of the wilderness area, the value of the extractable resource, and the benefit flow from development in the future are all uncertain while development is irreversible. The irreversibility of the development creates an option value that creates an incentive to preserve the wilderness area. Similarly, Leroux et al. (2009) develop a real options model of land conversion with stochastic future environmental damages. Saphores (2003) formulates a model to determine the harvest size of a renewable resource. Similar to our approach here, Saphores (2003) incorporates the risk of extinction in the decision model, showing that potential extinction creates an incentive to reduce harvest levels.

Our model includes a single ecosystem cost function that incorporates two different sources of uncertainty that are important to the results: uncertainty (variability) in the future time path of ecosystem costs that are generated by a large dam and uncertainty regarding whether the ecosystem will jump to an undesirable state (such as species extinction). Results from our model show that the sunk costs associated with a large investment combined with uncertainty regarding future ecosystem costs create an incentive to delay action that might help protected species recovery. The results also show a countervailing incentive to hasten the same investment when there is a risk that a species may become extinct. These results are predictable given the results of previous work (particularly Pindyck on pollution control cited above). Any irreversible cost in the presence of uncertain future benefits will create option value, while incorporating risk of moving to any extremely undesirable state creates an incentive to take action earlier. However, our work is novel in that it applies option pricing model to the issue of what action to take when attempting to recover threatened species. In contrast to the previous work on species extinction and conservation, the irreversibility in the model comes from the sunk cost associated with the expensive restoration action, rather than in the irreversible loss of ecosystem function. Our specification makes the problem applicable to species recovery efforts in a way not considered previously. In the Model section below, we construct a dynamic model that incorporates the uncertainties and irreversible outcomes just discussed. This is followed by a Results section that describes the outcomes of the model under different assumptions and demonstrates how different uncertainties and irreversible outcomes affect the decision. The final section is a discussion of these results and provides concluding comments.

MODEL

Here we formulate a model of whether to remove a dam before the end of its productive life. The objective is to maximize social welfare generated by a large dam. Social welfare is determined by the net benefits from the dam's operation—services such as hydropower, water supply and flood control less ecological costs such as negative benefits associated with reduced fish populations. In managing an ecological recovery decision like this, a decision-maker will monitor a variable or variables of interest to trigger a decision. In our model specification, the decision-maker monitors ecosystem costs (damages) imposed by the dam. We formulate an optimal stopping model where a decision-maker monitors the flow of net benefits from operating the dam over time and determines the conditions under which it is optimal to remove the dam prior to the end of its production life. In solving the dynamic problem we have specified, we assume the dam has T years of production life left. Upon reaching T years the dam must be removed, at cost K, if it has not been removed prior to reaching T years. The dam will be removed earlier than time T if the costs exceed the benefits by a threshold that is determined within the model. When the dam is removed we assume society absorbs a lump cost, equal to removal costs plus discounted residual ecological costs and foregone net benefits from operating the dam.

The benefits of dam removal in our model are due to reduced ecosystem costs. That is, we do not include a variable for ecosystem health level directly, but rather include dam-related ecosystem damages as a cost. This avoids having to map ecosystem health to a cost or to a utility received from poor ecosystem health. We specify ecosystem costs as a flow per unit of time, similar to the cash flow of an investment.

We present our model in two stages. First, we specify a model that assumes no uncertainty in ecosystem costs. Second, we add to the model stochastic ecosystem costs (described by Brownian motion) and observe how the optimal decision rule differs from the deterministic case. Within this second specification that includes stochastic ecosystem costs we further add the probability function for moving to an extreme cost state. This extreme state represents a condition where restoration of the ecosystem is too costly or impossible, like extinction of a key species. This specification contains one stochastic variable, ecosystem costs, in which we are able to model two types of uncertainty. The stochastic process that defines the ecosystem costs conveys uncertainty about how these costs will evolve and drives the option value result. The addition of a jump process incorporates the uncertainty regarding the ecosystem state uncertainty that drives the extinction risk result.

Deterministic Approach—No Uncertainty in Ecosystem Costs

Given an existing dam, we model the case when society bases a removal decision on forecasted costs and benefits of the dam with no uncertainty considered. In many finance texts this is referred to as the discounted cash flow approach. If the dam is torn down at the current time, $t = 0$, society receives no further benefits from the dam, accepts the residual flow of ecological costs, and pays a onetime removal cost, K. We represent the net value to society if the dam is removed at time 0 as $G(0)$.

$$G(0) = -(K + \int_{t=0}^{\infty} \overline{X}\,(t)\, e^{t(-\alpha) + t(-\rho)} dt) \tag{1}$$

In Equation (1) K is the cost to tear down the dam—the one-time investment in restoration $\overline{X}(t, \alpha)$ is the residual ecosystem costs. Residual costs are ecological costs that remain and continue to accrue after the dam is removed. These ecological costs diminish over time following dam removal as ecosystem function returns to its pre-dam condition. Examples of such residual costs may include the difference between actual and potential fish production as fish populations recover, or the short term ecological effects of the release of accumulated sediment behind the dam. We represent the residual cost as the discounted present value of a flow of costs that decays with time at a rate of α per year. We will refer to α as the speed of recovery parameter. The parameter ρ is the discount rate.

If the decision is delayed until a future time y, society receives a flow of net value (benefits—costs) from operating the dam over the time period between $t = 0$ and $t = y$, pays post-removal residual costs over the time period between $t = y$ and $t = \infty$, and delays paying removal costs until time y. The value of the dam if the removal is delayed until $t = y$ is the net value of the benefits generated by the dam minus the discounted removal costs that will occur in the future, as shown in Equation (2).

$$G(y) = \int_{t=0}^{t=y} \pi(t)e^{-\rho t}dt - (Ke^{-\rho y} + \int_{t=y}^{t=\infty} \overline{X}(t)\, e^{t(-\alpha)+t(-\rho)}dt) \tag{2}$$

In Equation (2), $\pi(t)$ is the net value flow from the dam. It is defined as the benefits from dam operation (e.g., the net value of hydropower generated at the dam) less ecosystem costs that accrue as long as the dam is in place (e.g., fish production that lost due to inaccessible spawning habitat above the dam). If $\pi(t)<0$ the dam is a net negative to social welfare: society could avoid paying a net cost by tearing down the dam at time t. If $\pi(t) > 0$ then society will lose a positive net benefit flow if the dam is removed at time t. Note that the dam has a fixed useful lifespan, so that t must be less than the maximum useful life, T.

In this deterministic setting a decision-maker can evaluate removing the dam at different time points based on forecasted net benefits and pick the time that yields the highest benefit. However, this decision approach does not consider uncertainty—the possibility that realized costs and benefits may differ from current forecasts. This uncertainty often leads one to delay the decision. We explore this case of considering uncertain costs next.

Stochastic Approach—Uncertainty in the Ecosystem Costs of the Dam

In this sub-section we add uncertainty in the evolution of ecosystem costs[1]. We specify ecosystem costs using a stochastic differential equation that combines Geometric Brownian Motion (GBM) with a jump process, as shown in Equation (3).

$$\begin{aligned} dX &= X\mu\, dt + X\sigma\, dz, \text{ if } I(X) = 0 \\ dX &= CE - X, \text{ if } I(X) = 1 \end{aligned} \tag{3}$$

In Equation (3), X is the ecosystem cost due to the dam per unit of time and μ is the expected rate of change in ecosystem cost per unit of time. Variability in the change in ecosystem cost per unit of time is represented by the parameter σ and larger values for σ indicates that there is more uncertainty about the future flow of ecosystem costs. dt is a time increment and dz is the increment of a Wiener process[2]. C_E is a fixed jump in the cost flow per unit of time, dX, if the ecosystem crashes to a state that makes restoration prohibitively costly. $I(X)$ is an indicator function that is equal to 1 if the ecosystem moves to the extremely high cost state over the next unit of time (dt) and zero otherwise.

At each time step in the model, depending on the level of ecosystem cost flow, X, there is a probability of the ecosystem's cost flow jumping to an extreme cost state, that is, species extinction or some other type of irreversible damage to the ecosystem. We use a Gompertz equation, shown in Equation (4), to model the probability of making such a jump in ecosystem costs as a function of the ecosystem cost, X.

$$p_E = exp\{-be^{-cX}\}, \text{ where } b \text{ and } c \text{ are positive parameters.} \tag{4}$$

As X increases, p_E moves closer to 1 and approaches it asymptotically for large values of X. The Gompertz equation is a useful functional form for the jump probability for three reasons: (1) it produces a low probability of a jump when the cost flow is low, (2) the probability of a jump rises as the ecosystem cost increases, and (3) it to asymptotes to 1 (i.e., the probability of a jump must remain less than 1). These properties accurately represent our belief that as the ecosystem degrades, the probability of jumping to a high cost state increases. The Gompertz equation meets these criteria, allowing us to evaluate how these criteria influence the removal decision.

At each point in time a decision maker chooses to either remove the dam or to leave it in place. Dam removal yields a net value to society, given by Equation (1), in the form of a reduction in the flow of future ecosystem damages. Dam retention leads to an ongoing flow of benefits, given by Equation (2), which could be negative if the ecosystem costs generated by the dam exceed the benefits from dam operation. The decision to remove the dam is based on whether the expected value of removing the dam outweighs the expected value of delaying the removal.

The decision-maker wants to maximize the value of the dam to society. The dam's value as a function of ecosystem costs and time, $F(X,t)$ can be expressed using the Bellman equation, shown in Equation (5), where the choice variable u represents the binary choice between removing the dam ($u = 1$) or not ($u = 0$).

$$F(X, t) = max_u \left\{\Omega(X, t), \pi(X, t) + (1 + \rho)^{-1} E[F(X + \Delta X, t + \Delta t)|X, u]\right\} \tag{5}$$

Equation (5) indicates that at the beginning of each time interval, society makes an optimal choice between (a) exercising the option to remove the dam prior to its full production life and receiving a payout of $\Omega(X,t)$ and (b) continuing to operate the dam and receiving an expected payout of $\pi(X, t) + (1 + \rho)^{-1} E[F(X + \Delta X, t + \Delta t)]$. The expected value of the dam is recursively determined assuming optimal decisions are made at each time point in the future.

The "payout" from removing the dam at time t, $\Omega(X,t)$, is the same as Equation (1): the discounted present value of dam

[1]Note that we treat benefits, such as hydropower production, deterministically so that our results are focused on ecosystem cost uncertainty.

[2]A Weiner process is also known as Brownian motion and $dz = \varepsilon(t)\sqrt{dt}$, where $\varepsilon(t) \sim N(0, 1)$.

removal costs and the value generated by avoiding ecosystem damages after removal. If, $\Omega(X,t)$, is less than the value of leaving the dam in place then the optimal decision is to wait until the next period and reevaluate. This case is shown in Equation (6).

$$\Omega(X, t) < \left(\pi(X, t) + (1 + \rho)^{-1} E[F(X + \Delta X, t + \Delta t)]\right) \quad (6)$$

If the less than sign in Equation (6) is reversed (i.e., if the discounted present value of removing the dam is greater than the value of leaving the dam in place), the optimal decision is to tear the dam down immediately. When Equation (6) is an equality, the value of the dam satisfies the return equilibrium condition (see Dixit and Pindyck, 1994, p. 109, Equation 13). This condition is represented by a stochastic differential equation that captures the effects of the system dynamics parameters on the "free boundary" dividing the state space into regions over which removal is optimal from regions where maintaining the dam is optimal. This free boundary is given by Equation (7).

$$\pi(X, t) + F_t(x, t) + \mu F_x(x, t) + (\sigma^2/2) F_{xx}(x, t) - \rho F(X, t) = 0 \quad (7)$$

Our goal is to determine the ecosystem cost threshold at each time period where the decision to tear down the dam occurs. At ecosystem cost levels above this threshold it is optimal to remove the dam. At ecosystem cost levels below this threshold it is optimal to delay the decision. Our model therefore solves for values of ecosystem cost and time (X,t) such that the value of removing the dam is exactly equal to the value of leaving the dam in place, i.e., Equation (8) is satisfied.

$$\Omega(X, t) = F(X, t) \quad (8)$$

Equation (8) specifies a curve in state (ecosystem cost), time space (X,t) where the value associated with continuing to operate the dam is the same as the value of the dam removed. This curve is known as the free boundary curve and defines a set of critical (ecosystem cost, time) pairs $(X^*(t), t)$. When $X(t) < X^*(t)$ it is optimal to continue operating the dam; when $X(t) > X^*(t)$ it is optimal to remove the dam. The free boundary curve provides a decision rule for the society as it observes ecosystem costs caused by the dam. When the ecological cost of the dam exceeds the free boundary curve, it is optimal to remove the dam; when the cost is less than the free boundary curve it is optimal to delay removal.

We use a binomial tree algorithm that is modified to account for the possibility of reaching an extreme cost state. The algorithm is presented in the Appendix of Supplementary Material.

Our model specification captures the option value of delaying action on a large (costly) and irreversible investment in species recovery in order to wait for more information. This is a well-known feature of option pricing problems and is often excluded in benefit-cost analyses of protected resource recovery actions. In addition, the specification of the ecosystem costs time path with a jump process captures the possibility that the delay may have very bad consequences. In the protected resource context, delaying costly action may result in extinction.

To show the effects of uncertainty in ecosystem costs and properties of the ecosystem, we solve the model for multiple values of ecosystem cost variability (σ), the expected rate at which ecosystem costs increase in the absence of any restoration investment (i.e., with the dam in place) (μ), and the speed at which the ecosystem can recover after the investment is made (α). We also solve the model for multiple parameter values in the extinction probability function (the jump process, Equation 4) to illustrate the effect of adding probabilistic extinction risk to the cost-benefit analysis. Our results show that the value of waiting for more information and the desire to avoid a severe cost outcome work against each other in determining the optimal decision.

We choose representative values for the constants used to solve the model numerically. The length of time remaining in the dam's useful life (T) is set at 20 years, a reasonable number given that in the United States hydropower operating licenses may be issued for up to 50 years. The lump sum cost to remove the dam (K) is set to \$30 million. Dam removal costs vary widely based on many factors, but many recent dam removal projects designed to improve habitat for anadromous fish on the Pacific coast of the United States fall within this range. For example, detailed, site-specific studies estimated removal costs in several cases:

- Four dams on the Klamath River in California—between \$19.3 and \$83.9 million (in 2012 dollars) per dam (US Bureau of Reclamation, 2012a).
- Condit Dam on the White Salmon River—\$20.5 million (2005 dollars; Washington State Department of Ecology, 2007).
- Marmot Dam on the Sandy River in Oregon—\$17 million (Portland General Electric, 2002).
- Savage Rapids Dam, Rogue River, Oregon—\$28 million (American Rivers, 2006).

We set the annualized benefit flow from preserving the dam at \$3 million per year. This is a reasonable number based on recent estimates from dam removal projects. For example, the annualized reduction in hydropower benefits from removal of four dams on the Klamath River was estimated to be about \$26.4 million per year (or \$6.6 million per dam; US Bureau of Reclamation, 2012b). The annual flow of hydropower benefits from Condit Dam before its removal was estimated to be approximately \$2.5 million (Federal Energy Regulatory Commission, 2002).

RESULTS

In this section we demonstrate the results of our model for a specific example and show how varying parameter values changes the optimal dam removal decision rule. In Sections Sensitivity Analysis to Ecosystem Cost Variability, σ, Sensitivity Analysis of the Drift Rate (μ), and Sensitivity to the Speed of Ecosystem Recovery (α), we solve the model for the free boundary curve in the case where there is uncertainty in ecosystem costs, but with no potential of moving to an extreme cost state. The free boundary curve represents a decision threshold that triggers dam removal at a given point in time. If ecosystem costs exceed the value specified in the free boundary curve at a point in time, the

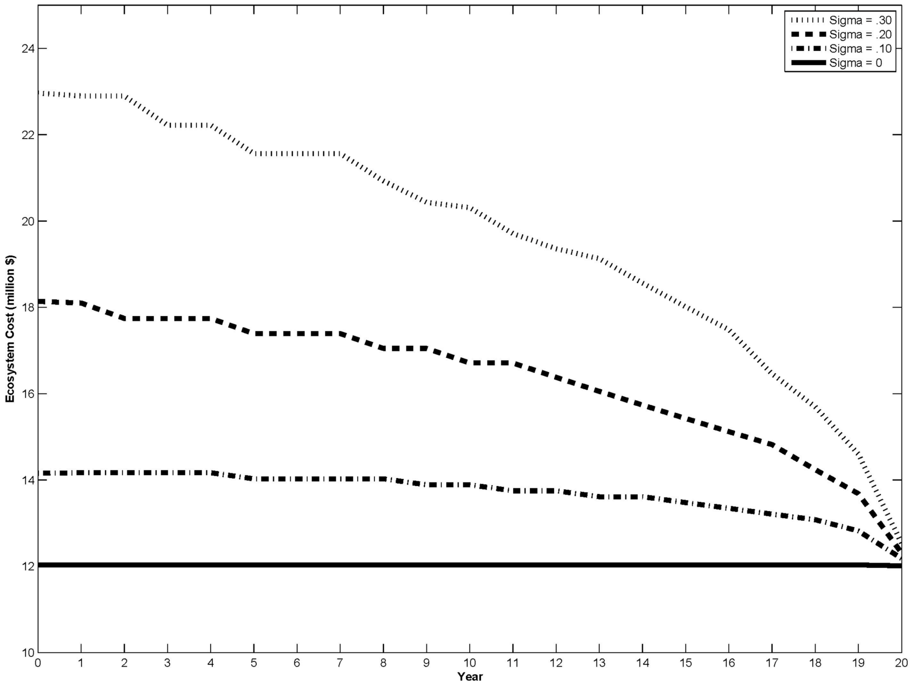

FIGURE 1 | Critical cost (free boundary) curves for different levels of ecosystem cost uncertainty, σ. The curves represent the threshold value of ecosystem costs at which the dam should be removed.

optimal decision is to remove the dam at that point. In these sections, we will show how the optimal decision rule is affected by the amount of ecosystem cost variability (σ), the expected rate at which ecosystem costs increase (μ), and the speed of recovery parameter (α) at which the ecosystem recovers following dam removal. In Section Positive Probability of Moving to an Extreme Cost State, we introduce the possibility of an event that leads to the extreme cost state, such as species extinction. The stair step nature of the resulting free boundary curves (**Figures 1–4**) is due to the discrete numerical algorithm used. Reducing the step size in our solution procedure (see the Appendix in Supplementary Material) would make the curves smoother, but would greatly increase the time required to solve the model. The results are based on the difference between curves generated under alternative assumptions not the slope of individual curves. The chosen time step in our solutions (0.01 years) generates curves that are sufficiently well-defined to draw insights regarding the effects of changing uncertainty parameters while requiring a practical amount of time to generate a solution for each set of assumptions.

Sensitivity Analysis to Ecosystem Cost Variability, Σ

Figure 1 shows critical cost curves obtained from solving the model using the parameter values in **Table 1**, but with varying levels of ecosystem cost uncertainty (σ) from 0 to 30 percent. In **Figure 1**, we observe that variability in ecosystem costs generated by the dam (i.e., $\sigma > 0$) *increases* the critical value at which the dam should be removed. The difference between the curves where ecosystem cost variability exists and the $\sigma = 0$ curve is the value of the option associated with delaying dam removal.

In **Figure 1** all of the critical costs curves, regardless of the magnitude of the ecosystem cost variability, converge to the deterministic critical removal value as the end of the dam's production life is reached. Options decrease in value as the time to maturity approaches, i.e., as the end of the dam's production life is reached. There are two reasons for this. First, the discounted present value of the future benefits (e.g., hydropower, flood control) declines as fewer years of the benefits can be realized. Second, and more important to our discussion of the effects of uncertainty on the dam removal decision, there is less uncertainty regarding the magnitude of the ecosystem costs associated with the dam because the stochastic process describing these costs has less time to drift.

It might seem counter-intuitive that greater ecosystem cost variability prior to dam removal creates an incentive to delay a project intended to restore ecosystem health. However, this is explained by the fact that option values increase with the level of uncertainty, σ. In our model, this means that there is value associated with waiting for more information when the uncertainty is high. The dam removal is irreversible, so there are sunk costs if the dam is removed. Consider that even if ecological costs are high, it is possible that they may decrease as the stochastic ecosystem damages evolve over time -making paying for dam removal avoidable. Meanwhile, if ecological damages move to higher levels, decision-makers can observe this and remove the dam. Hence, additional uncertainty, with the ability to take action, increases the value of waiting. This

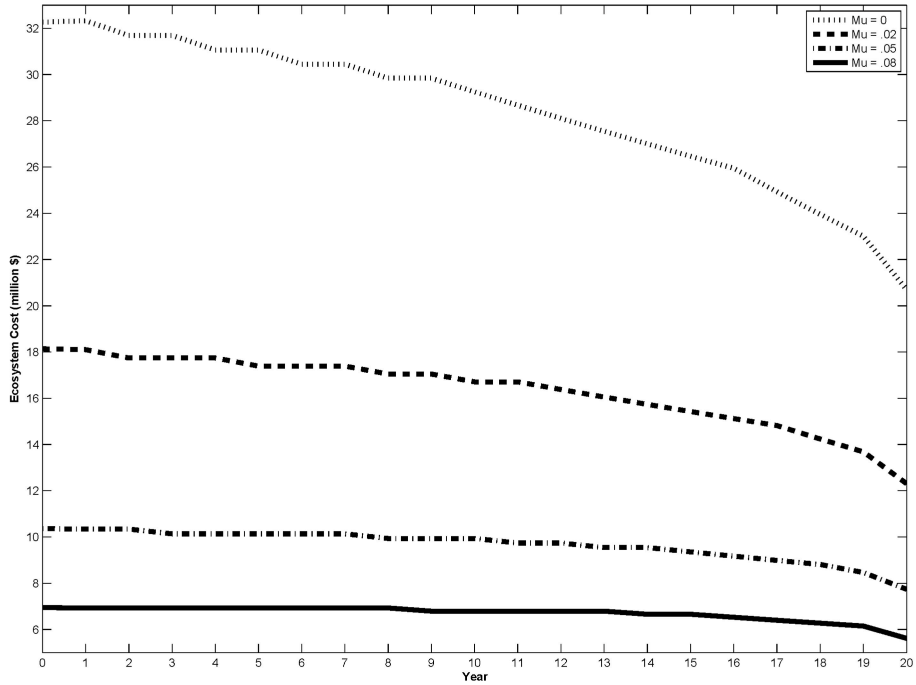

FIGURE 2 | Critical cost curves for different values of the drift rate in ecosystem costs, μ. The curves represent the threshold value of ecosystem costs at which the dam should be removed.

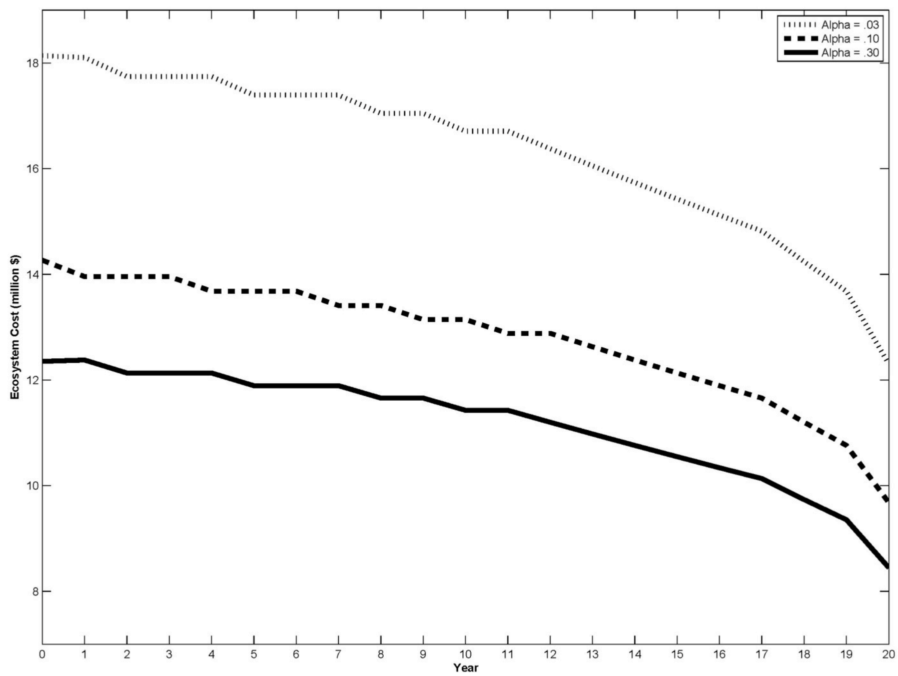

FIGURE 3 | Critical cost curves for different values of the speed of recovery parameter, α. The curves represent the threshold value of ecosystem costs at which the dam should be removed.

raises the level of ecosystem cost that society is willing to tolerate before removing the dam relative to the case where ecosystem costs are certain. Of course, though, at some level of cost waiting must cease, and the dam should be removed. This is what the solution to the optimal stopping problem provides—an illustration of the value of delaying a costly ecosystem

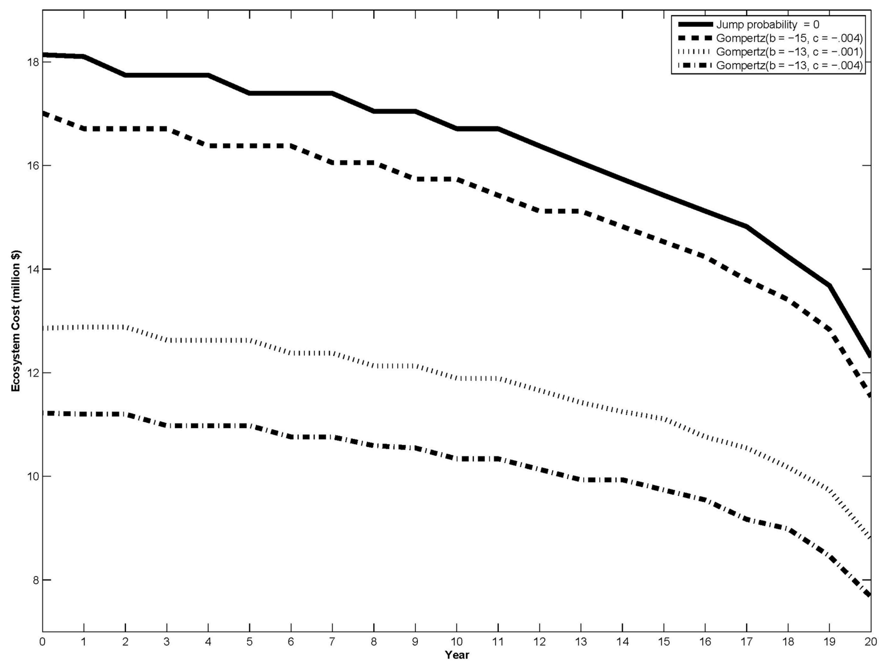

FIGURE 4 | Critical cost curves for different values of the probability of jumping to an extreme cost state. The curves represent the threshold value of ecosystem costs at which the dam should be removed.

TABLE 1 | Example model parameters.

Parameter	Description	Value
T	Number of years remaining in the dams useful life. Dam must be removed as the end of its useful life	20 years
K	Cost of removing the dam	$30 million
$B(t)$	Expected annual benefit derived from the dam the dam, e.g., value of flood control, value of water supply services, net income from hydropower	$ 3 million per year
ρ	Discount rate	5%
α	Rate at which ecosystem cost decays after the dam is removed	3%[a]
μ	Expected increase in ecosystem costs per year if the dam remains in place	2%[a]
σ	Standard deviation of the annual change in ecosystem costs	20%[b]
Δt	Time step	0.01 years

The example model parameters are consistent with fixed values chosen to run numerical examples in previous work on uncertainty and irreverisibility.

[a]*Leroux et al. (2009): $\alpha = 0.05$, rate of increase in species value; Kassar and Lasserre (2004): $\alpha = 0.04$, rate of change in species value; Conrad (2000): $\gamma = 0.03$, drift rate in wilderness amenity values; Pindyck (2000, 2002): $\delta = 0.02$, rate at which the stock of pollutant decays.*

[b]*Leroux et al. (2009): $\sigma = 0.1$ SD of change in species value; Kassar and Lasserre (2004): $\sigma^2 = 0.02$,*
$\sigma = 0.14$, SD of change in species value; Conrad (2000): $\sigma_E = 0.3$, uncertainty wilderness amenity values.

restoration action. Note that when we incorporate the possibility of jumping to an extreme cost state (e.g., species extinction) in Section Positive Probability of Moving to an Extreme Cost State below, we will observe a countervailing incentive to hasten the action.

Sensitivity Analysis of the Drift Rate (μ)

We specify our model so that ecosystem costs increase stochastically over time as long as the dam remains in place. This is implied by a drift rate parameter, μ, greater than zero (see Equation 3). This drift rate parameter represents the rate at which ecosystem costs increase when the dam is left in place. **Figure 2** shows the critical cost curves for several values of μ including the base value in **Table 1**; all other parameters are held constant at **Table 1** values. A higher value of μ shifts the free boundary curve down, i.e., decreases the critical cost at which the dam should be removed. Increasing the drift rate increases the expected rate at which ecosystem costs from the dam accrue. This creates an incentive to remove the dam at an earlier time period. Recall from Section Sensitivity Analysis to Ecosystem Cost Variability, σ and **Figure 1** that the critical cost for positive σ converges to the deterministic critical cost at the end of the dam's production life. **Figure 2** shows that changing μ causes the deterministic critical cost to shift. For higher μ, it is more valuable to remove the dam at a lower ecosystem cost for any fixed σ. The expected flow of ecosystem cost paid over the next interval of time increases with μ, as does the expected residual costs of the dam. So, higher rates of growth in ecosystem costs imply a lower critical cost that would trigger dam removal.

Sensitivity to the Speed of Ecosystem Recovery (α)

The parameter α describes how quickly the ecosystem will recover once the dam is removed and thus determines the value of residual costs. In our model the ecosystem recovers (i.e., ecosystem cost decays) exponentially at a rate of α after the dam is removed. When α is low the ecosystem recovers slowly; when α is high the ecosystem recovers quickly. **Figure 3** shows the critical cost curves for different values of α including the **Table 1** base value with all other parameters set at **Table 1** values.

Changes in the ecosystem cost decay parameter (α) shift the deterministic critical cost and therefore the point to which positive σ cases converge. The discounted residual costs can be thought of as a lump sum cost that is paid when the dam is removed. Decreasing α increases this lump sum payment because it lowers the rate at which costs decay. The higher the value of this lump sum payment, the greater the benefit to delaying it into the future. According to this model, society is willing to absorb a higher ongoing ecosystem cost when the ecosystem is slow to recover after the dam removal investment (i.e., when residual costs are high) because the present value of the benefit of removing the dam is not as great.

Positive Probability of Moving to an Extreme Cost State

Thus, far we've analyzed the decision of removing the dam under a model of costs changing stochastically according to GBM. While GBM provides a stochastic model for costs, it does not represent the potential outcome of jumping to an extreme cost state—a state that represents extinction or another type of severe event. Under GBM cost uncertainty we found that more uncertainty led to further value in delaying the decision to act. Now we consider that waiting may result in with unacceptably high ecosystem costs.

This risk of irreversible ecological cost is modeled by incorporating a positive probability of jumping to an extreme cost state, C_E (see the jump process in Equation 3). In the example cases below let the extreme cost value $C_E = \$700$ million. To get a sense of the relative size of this number, recall from **Figure 1**, that the costs where it is optimal to remove the dam are in the region of \$10 million. In one of the curves we use the following parameterization of the Gompertz equation, which gives the probability of jumping to an undesirable state (e.g., species extinction):

$$p_E(X, b = -13, c = -0.004) = exp\left\{-13e^{-0.004X}\right\} \tag{9}$$

These values and others were chosen to provide an interesting example; in practice they would need to be estimated.

Figure 4 shows the effect of including extinction risk in the model. Incorporating the possibility of jumping to a high cost state causes the free boundary curve to shift down, as shown by the dotted lines in **Figure 4**. This means that the dam removal is triggered at a lower level of ecosystem costs due to the dam. The additional parameterizations of the jump probability illustrate that any positive probability of jumping to an extreme cost state results in a lower threshold cost. The amount of the shift depends on the parameterization of the jump probability function[3]. The possibility of jumping to an extreme cost state produces a countervailing incentive to the delay option shown in isolation in Sections Sensitivity Analysis to Ecosystem Cost Variability, σ, Sensitivity Analysis of the Drift Rate (μ), and Sensitivity to the Speed of Ecosystem Recovery (α). While the delay option increases the critical cost to trigger restoration, the possibility of jumping to an extreme cost state decreases the critical cost to trigger restoration. With both of these incentives (delay and hasten) present in the same model, a balance is reached where both the value of waiting for more information and the potential risk of waiting too long are considered.

DISCUSSION

In this article, we presented a model that incorporates two types of uncertainty that affect the timing of large, irreversible investments in ecosystem restoration: uncertainty regarding stochastically evolving ecosystem costs (geometric Brownian uncertainty) and uncertainty regarding a jump to an unacceptably high level of ecosystem costs (e.g., species extinction). These two uncertainties connect to different irreversible outcomes, sunk costs and permanent ecosystem damage. We use our model to illustrate two main results. First, variability in ecosystem costs creates an incentive to delay a project intended to restore ecosystem health. The uncertainty regarding ecosystem costs creates an option value to waiting to invest in restoration at a later date. Second, the possibility jumping to an irreversible and unacceptably bad ecosystem state creates an incentive to hasten restoration. Many large investments in ecosystem restoration have both characteristics: uncertain ecosystem costs going forward and a risk of outcomes such as species extinction. Therefore, policy makers are faced with countervailing incentives when deciding when to make these investments.

Uncertainty represented by geometric Brownian motion in the evolution of ecosystem costs creates an incentive to delay restoration investments. In our example of dam removal, varying or declining fish populations due to the dam create ecosystem costs that evolve stochastically over time. Policymakers considering dam removal may believe that costs due to poor fish habitat will increase, but this is not a certain outcome. Because dam removal is an irreversible decision, this uncertainty creates an option value which encourages delaying the decision beyond what a deterministic model would show. By contrast, when there is a chance that ecosystem costs can jump to an unacceptably high cost state, an incentive to hasten the restoration investment follows. Policymakers will want to

[3]In general, the b parameter in the Gompertz equation (see Equation 9) shifts the Gompertz curve left and right. The c parameter determines the growth rate of the probability. Using b to shift the Gompertz curve to the right (for example b going from −13 to −15) decreases the chance of jumping to the extreme state for the same value of X. This parameter perturbation moves the critical cost curve up toward the curve where there is a zero probability of jumping to the extreme cost state. Increasing the probability growth rate (parameter c) moves the critical cost curve down. This can be seen in **Figure 4** from the example where the growth rate is changed from 0.001 to 0.004 for the same value of $b = -13$. A full sensitivity analysis involving the jump probability function parameters would be an interesting exercise, but is beyond the scope of this paper. Such an exercise would involve linking these parameters to specific ecological characteristics of the system.

consider each of these countervailing incentives as they make restoration decisions.

Our work applies the result of an option value modeling framework to the important issue of planning for protected species recovery. The results of our work are consistent with previous work that incorporates option value into environmental decision-making, such as pollution prevention investments. Any large, irreversible investment in the presence of uncertain future benefits creates an option value which delays the optimal time of the investment. Similarly, when there is a possibility of moving to an extremely undesirable state, the optimal time of the investment is hastened.

Previous work has applied such models to species extinction problems. That work generally describes the irreversibilty associated with destroying habitat or otherwise reducing the viability of a protected species. Our model, however, specifies investment in an expensive recovery action as the sunk cost. Therefore, we apply the option value framework to a different decision that often faces decision makers: i.e., the best time to spend a large amount of money on recovering a species at risk.

Our model provides a systematic way of understanding the risks and irreversibilities involved in ecosystem management. Clarifying how uncertainty and irreversibility affect the optimal timing of restoration expenditures can be useful as policymakers decide which projects to pursue and in helping to prioritize expenditures and research. Moreover, uncertainty can be reduced by better insight gained through research. Understanding how different uncertainties influence the value of ecosystem restoration decisions can help scientists direct their expertise to research that optimizes the use of funding aimed at ecosystem restoration.

Reaching a potentially extreme cost state is currently an issue in some rivers where Pacific salmon species are listed under the United States Endangered Species Act. Policymakers are currently evaluating large dam removal projects. Our hope is that the dynamic model illustrated here provides insight into making some of these difficult decisions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://journal.frontiersin.org/article/10.3389/fmars.2015.00101

Figure S1 | Illustration of the the cost diffusion in the binomial tree. C_0 is the initial ecosystem cost. C_E is the cost associated with moving to an ecosystem state where restoration is prohibitively costly. C_u and C_d are the potential states that the cost can move to in the next time step under a binomial process.

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