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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.

Marine species	Source study	Country
Martín-López et al. (2008, CONSERVATION BIOLOGY)		
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.

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,

[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.

[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).

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

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 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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REFERENCES

Adamowicz, W. L., Bhardwaj, V., and McNab, B. (1993). Experiments on the difference between willingness to pay and willingness to accept. *Land Econ.* 69, 416–427. doi: 10.2307/3146458

Aldrich, G. A., Grimsrud, K. M., Thacher, J. A., and Kotchen, M. J. (2007). Relating environmental attitudes and contingent values: how robust are methods for identifying preference heterogeneity? *Environ. Resour. Econ.* 37, 757–775. doi: 10.1007/s10640-006-9054-7

Alpizar, F., Carlsson, F., and Martinsson, P. (2003). Using choice experiments for non-market valuation. *Econ. Issues* 8, 83–110. Available online at: http://www.economicissues.org.uk/Files/2003/103fUsing%20Choice%20Experiments%20for%20Non%20Market%20Valuation.pdf

Alvarez-Farizo, B., Hanley, N., Barberan, R., and Lazaro, A. (2007). Choice modelling at the 'Market Stall': individual versus collective interest in environmental valuation. *Ecol. Econ.* 60, 743–751. doi: 10.1016/j.ecolecon.2006.01.009

Arrow, K., Solow, R., Portney, P. R., Leamer, E. E., Radner, R., and Schuman, H. (1993). Report of the NOAA panel on contingent valuation. *Federal Regist.* 58, 4601–4614.

Balbontin, C., de Ortuzar, J. D., and Swait, J. D. (2015). A joint best-worst scaling and stated choice model considering observed and unobserved heterogeneity: an application to residential location choice. *J. Choice Model.* 16, 1–14. doi: 10.1016/j.jocm.2015.09.002

Bateman, I. J., Carson, R. T., Day, B., Hanemann, M., Hanley, N., Hett, T., et al. (2002). *Economic Valuation with Stated Preference Technique: A Manual.* Cheltenham: Edward Elgar. doi: 10.4337/9781781009727

Bateman, I. J., Cole, M. A., Georgiou, S., and Hadley, D. J. (2006). Comparing contingent valuation and contingent rating: a case study considering the benefits of urban river water quality improvements. *J. Environ. Manage.* 79, 221–231. doi: 10.1016/j.jenvman.2005.06.010

Bell, K. P., Huppert, D., and Johnson, R. L. (2003). Willingness to pay for local Coho Salmon enhancement in coastal communities. *Mar. Resour. Econ.* 18, 15–31. Available online at: http://www.jstor.org/stable/42629381

Bennett, J., and Birol, E. (2010). *Choice Experiments in Developing Countries.* Cheltenham: Edward Elgar. doi: 10.4337/9781781000649

Bergstrom, J. C., and Taylor, L. O. (2006). Using meta-analysis for benefits transfer: theory and practice. *Ecol. Econ.* 60, 351–360. doi: 10.1016/j.ecolecon.2006.06.015

Berrens, R. P., Bohara, A. K., Silva, C. L., Brookshire, D., and McKee, M. (2000). Contingent values for New Mexico instream flows: with tests of scope, group-size reminder and temporal reliability. *J. Environ. Manag.* 58, 73–90. doi: 10.1006/jema.1999.0308

Blamey, R. K., Bennett, J. W., and Morrison, M. D. (1999). Yea-saying in contingent valuation surveys. *Land Econ.* 75, 126–141. doi: 10.2307/3146997

Bliem, M., Getzner, M., and Rodiga-Laßnig, P. (2012). Temporal stability of individual preferences for river restoration in Austria using a choice experiment. *J. Environ. Manag.* 103, 65–73. doi: 10.1016/j.jenvman.2012.02.029

Bockstael, N. E., and McConnell, K. E. (2007). *Environmental and Resource Valuation with Revealed Preferences: A Theoretical Guide to Empirical Models.* Dordrecht: Springer.

Bosetti, V., and Pearce, D. (2003). A study of environmental conflict: the economic value of Grey Seals in southwest England. *Biodivers. Conserv.* 12, 2361–2392. doi: 10.1023/A:1025809800242

Boxall, P. C., Adamowicz, W. L., Olar, M., West, G. E., and Cantin, G. (2012). Analysis of the economic benefits associated with the recovery of threatened marine mammal species in the Canadian St. Lawrence Estuary. *Mar. Policy* 36, 189–197. doi: 10.1016/j.marpol.2011.05.003

Boxall, P. C., Adamowicz, W. L., Swait, J., Williams, M., and Louviere, J. (1996). A comparison of stated preference methods for environmental valuation. *Ecol. Econ.* 18, 243–253. doi: 10.1016/0921-8009(96)00039-0

Boyle, K. J. (2003). "Contingent valuation in practice," in *A Primer on Nonmarket Valuation,* eds A. Patricia Champ, J. Kevin Boyle, and C. Thomas Brown (Dordrecht: Kluwer Academic Publishers). doi: 10.1007/978-94-007-0826-6_5

Boyle, K. J., and Bergstrom, J. (1992). Benefit transfer: myths, pragmatism, and idealism. *Water Resourc. Res.* 28, 657–663. doi: 10.1029/91WR02591

Boyle, K. J., and Bishop, R. C. (1987). "Valuing wildlife in benefit-cost analyses: a case study involving endangered species. *Water Resour. Res.* 23, 943–950. doi: 10.1029/WR023i005p00943

Brouwer, R. (2000). Environmental value transfer: state of the art and future prospects. *Ecol. Econ.* 32, 137–152. doi: 10.1016/S0921-8009(99)00070-1

Brouwer, R. (2006). Do stated preference methods stand the test of time? A test of the stability of contingent values and models for health risks when facing an extreme event. *Ecol. Econ.* 60, 399–406. doi: 10.1016/j.ecolecon.2006.04.001

Brouwer, R., and Bateman, I. J. (2005). Temporal stability and transferability of models of willingness to pay for flood control and wetland conservation. *Water Resour. Res.* 41, W03017. doi: 10.1029/2004WR003466

Brouwer, R., Martin-Ortega, J., Dekker, T., Sardonini, L., Andreu, J., Kontogianni, A., et al. (2015). Improving value transfer through socio-economic adjustments in a multicountry choice experiment of water conservation alternatives. *Aust. J. Agric. Resour.* Econ. 59, 1–21. doi: 10.1111/1467-8489.12099

Brown, G., Layton, D., and Lazo, J. (1994). *Valuing Habitat and Endangered Species*. Discussion paper 94-1. Seattle, WA: Institute for Economic Research, University of Washington.

Brown, T. C. (2003). "Introduction to stated preference methods," in *A Primer on Nonmarket Valuation,* eds A. Patricia Champ, J. Kevin Boyle, and C. Thomas Brown (Dordrecht: Kluwer Academic Publishers), 99–110. doi: 10.1007/978-94-007-0826-6_4

Bulte, E. H., and van Kooten, G. C. (1999). Marginal valuation of charismatic species: implications for conservation. *Environ. Resour. Econ.* 14, 119–130. doi: 10.1023/A:1008309816658

Burton, A. C., Carson, K. S., Chilton, S. M., and Hutchinson, W. G. (2007). Resolving questions about bias in real and hypothetical referenda. *Environ. Resour. Econ.* 38, 513–525. doi: 10.1007/s10640-007-9095-6

Carlsson, F., and Martinsson, P. (2001). Do hypothetical and actual marginal willingness to pay differ in choice experiments? *J. Environ. Econ. Manag.* 41, 179–192. doi: 10.1006/jeem.2000.1138

Carson, R. T., Flores, N. E., Martin, K. M., and Wright, J. L. (1996). Contingent valuation and revealed preference methodologies: comparing the estimates for quasi-public goods. *Land Econ.* 72, 80–99. doi: 10.2307/3147159

Carson, R. T., Flores, N. E., and Meade, N. F. (2001). Contingent valuation: controversies and evidence. *Environ. Resour. Econ.* 19, 173–210. doi: 10.1023/A:1011128332243

Carson, R. T., Flores, N. E., and Mitchell, R. C. (1999). "The theory and measurement of passive-use values," in *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries*, Chapter 4, eds I. J. Bateman and K. G. Willis (Oxford: Oxford University Press), 97–130.

Carson, R. T., Hanemann, W. M., Kopp, R. J., Krosnick, J. A., Mitchell, R. C., Presser, S., et al. (1994). *Prospective Interim Lost Use Value Due to DDT and PCB Contamination in the Southern California Bight*. Report to the National Oceanic and Atmospheric Administration, Contract No. 50-DGNC-1-00007.

Carson, R. T., Hanemann, W. M., Kopp, R. J., Krosnick, J. A., Mitchell, R. C., Presser, S., et al. (1997). Temporal reliability of estimates from contingent valuation. *Land Econ.* 73, 151–163. doi: 10.2307/3147279

Champ, P. A., Bishop, R. C., Brown, T. C., and McCollum, D. W. (1997). Using donation mechanisms to value nonuse benefits from public goods. *J. Environ. Econ. Manag.* 33, 151–162. doi: 10.1006/jeem.1997.0988

Champ, P. A., Moore, R., and Bishop, R. C. (2009). A comparison of approaches to mitigate hypothetical bias. *Agric. Resour. Econ. Rev.* 38, 166–180. Available online at: http://purl.umn.edu/55867

Christie, M., and Azevedo, C. D. (2009). Testing the consistency between standard contingent valuation, repeated contingent valuation, and choice experiments. *J. Agric. Econ.* 60, 154–170. doi: 10.1111/j.1477-9552.2008.00178.x

Colombo, S., Christie, M., and Hanley, N. (2013). What are the consequences of ignoring attributes in choice experiments? Implications for ecosystem service valuation. *Ecol. Econ.* 96, 25–35. doi: 10.1016/j.ecolecon.2013.08.016

Colombo, S., Hanley, N., and Louviere, J. (2009). Modeling preference heterogeneity in stated choice data: an analysis for public goods generated by agriculture. *Agric. Econ.* 40, 307–322.

Cummings, R. G., and Harrison, G. W. (1995). The measurement and decomposition of nonuse values: a critical review. *Environ. Resour. Econ.* 5, 225–247. doi: 10.1007/BF00691518

Cummings, R. G., and Taylor, L. O. (1999). Unbiased value estimates for environmental goods: a cheap talk design for the contingent valuation method *Am. Econ. Rev.* 89, 649–665. doi: 10.1257/aer.89.3.649

Desvousges, W. H., Naughton, M. H., and Parsons, G. R. (1992), Benefit transfer: conceptual problems in estimating water quality benefits using existing studies. *Water Resour. Res.* 28, 675–683. doi: 10.1029/91WR02592

Diamond, P. A., and Hausman, J. A. (1994). Contingent valuation: is some number better than no number? *J. Econ. Perspect.* 8, 45–64. doi: 10.1257/jep.8.4.45

Duffield, J., and Patterson, D. (1992). *Field Testing Existence Values: Comparison of Hypothetical and Cash Transaction Values*. Benefits and Costs in Natural Resource Planning, 5th Report. W-133 Western Regional Research Publication. B. Rettig, compiler, Department of Agricultural and Resource Economics, Oregon State University, Corvallis, OR.

Dunford, R. W., Ginn, T. C., and Desvousges, W. H. (2004). The use of habitat equivalency analysis in natural resource damage assessments. *Ecol. Econ.* 48, 49–70. doi: 10.1016/j.ecolecon.2003.07.011

Ehmke, M. D., Lusk, J. L., and List, J. A. (2008). Is hypothetical bias a universal phenomenon? A multinational investigation. *Land Econ.* 84, 489–500. doi: 10.3368/le.84.3.489

Endter-Wada, J., Blahna, D., Krannich, R., and Brunson, M. (1998). A framework for understanding social science contributions in ecosystem management. *Ecol. Appl.* 8, 891–904. doi: 10.1890/1051-0761(1998)008[0891:AFFUSS]2.0.CO;2

Farber, S., Costanza, R., Childers, D. L., Erickson, J., Gross, K., Grove, M., et al. (2006). Linking ecology and economics in ecosystem management. *BioScience* 56, 121–133. doi: 10.1641/0006-3568(2006)056[0121:LEAEFE]2.0.CO;2

Farr, M., Stoeckl, N., and Beg, R. A. (2014). The non-consumptive (Tourism) 'Value' of marine species in the northern section of the Great Barrier Reef. *Mar. Policy* 43, 89–103. doi: 10.1016/j.marpol.2013.05.002

Fiebig, D. G., Keane, M. P., Louviere, J., and Wasi, N. (2010). The generalized multinomial logit model: accounting for scale and coefficient heterogeneity. *Mark. Sci.* 29, 393–421. doi: 10.1287/mksc.1090.0508

Fredman, P. (1995). The existence of existence value—a study of the economic benefits of an endangered species. *J. Forest Econ.* 1, 307–328.

Freeman, A. M., Herriges, J. A., and Kling, C. L. (2014). *The Measurement of Environmental and Resource Values: Theory and Methods, 3rd Edn*. Washington, DC: Resources for the Future.

Giraud, K., Loomis, J. B., and Johnson, R. L. (1999). Internal and external scope in willingness-to-pay estimates for threatened and endangered wildlife. *J. Environ. Manag.* 56, 221–229. doi: 10.1006/jema.1999.0277

Giraud, K., Turcin, B., Loomis, J., and Cooper, J. (2002). Economic benefits of the protection program for the Steller sea lion. *Mar. Policy*. 26, 451–458. doi: 10.1016/S0308-597X(02)00025-8

Giraud, K, and Valcic, B. (2004). Willingness-to-pay estimates and geographic embedded samples: case study of Alaskan Steller sea lion. *J. Int. Wildl. Law Policy* 7, 57–72. doi: 10.1080/13880290490480167

Glenk, K., Martin-Ortega, J., Pulido-Velazquez, M., and Potts, J. (2015). Inferring attribute non-attendance from discrete choice experiments: implications for benefit transfer. *Environ. Resour. Econ.* 60, 497–520. doi: 10.1007/s10640-014-9777-9

Hageman, R. K. (1985). *Valuing Marine Mammal Populations: Benefit Valuations in a Multi-Species Ecosystem*. Southwest Fisheries Center, National Marine Fisheries Service, Administrative Report, LJ-85–J-22.

Hageman, R. K. (1986). "Economic valuation of marine wildlife: does existence value exist?" in *Paper presented at the Association of Environmental and Resource Economists Workshop on Marine Pollution and Environmental Damage Assessment* (Narragansett, RI).

Hanemann, M., Loomis, J., and Kanninen, B. (1991). Statistical efficiency of double-bounded dichotomous choice contingent valuation. *Am. J. Agric. Econ.* 73, 1255–1263. doi: 10.2307/1242453. Available online at: http://www.jstor.org/stable/1242453

Hanley, N., Mourato, S., and Wright, R. E. (2001). Choice modelling approaches: a superior alternative for environmental valuation? *J. Econ. Surveys* 15, 435–462. doi: 10.1111/1467-6419.00145

Hanley, N., Wright, R. E., and Adamowicz, V. (1998). Using choice experiments to value the environment: design issues, current experience, and future prospects. *Environ. Resour. Econ.* 11, 413–428. doi: 10.1023/A:1008287310583

Hausman, J. A. (1993). *Contingent Valuation: A Critical Assessment*. Amsterdam: North Holland Press. doi: 10.1108/S0573-8555(1993)220

Hausman, J. A. (2012). Contingent valuation: from dubious to hopeless. *J. Econ. Perspect.* 26, 43–56. doi: 10.1257/jep.26.4.43

Herriges, J. A., and Kling, C. L. (1999). *Valuing Recreation and the Environment: Revealed Preference Methods in Theory and Practice*. Northampton, MA: Edward Elgar.

Horowitz, J. K., and McConnell, K. E. (2002). A review of WTA/WTP studies. *J. Environ. Econ. Manag.* 44, 426–447. doi: 10.1006/jeem.2001.1215

Jin, J., Indab, A., Nabangchang, O., Thuy, T. D., Harder, D., and Subade, R. F. (2010). Valuing marine turtle conservation: a cross-country study in Asian cities. *Ecol. Econ.* 69, 2020–2026. doi: 10.1016/j.ecolecon.2010.05.018

Johnston, R. J., Besedin, E. Y., and Ranson, M. H. (2006). Characterizing the effects of valuation methodology in function-based benefits transfer. *Ecol. Econ.* 60, 407–419. doi: 10.1016/j.ecolecon.2006.03.020

Johnston, R. J., and Rosenberger, R. S. (2010). Methods, trends, and controversies in contemporary benefit transfer. *J. Econ. Surveys* 24, 479–510. doi: 10.1111/j.1467-6419.2009.00592.x

Johnston, R. J., Schultz, E. T., Segerson, K., Besedin, E. Y., and Ramachandran, M. (2012). Enhancing the content validity of stated preference valuation: the structure and function of ecological indicators. *Land Econ.* 88, 102–120. doi: 10.3368/le.88.1.102

Jones, C. A. (2000). Economic valuation of resource injuries in natural resource liability suits. *J. Water Resour. Plan. Manag.* 126, 358–365. doi: 10.1061/(ASCE)0733-9496(2000)126:6(358)

Kanninen, B. J. (1993). Design of sequential experiments for contingent valuation studies. *J. Environ. Econ. Manag.* 25, S1–S11. doi: 10.1006/jeem.1993.1029

Kaul, S., Boyle, K. J., Kuminoff, N. V., Parmeter, C. F., and Pope, J. C. (2013). What can we learn from benefit transfer errors? Evidence from 20 years of research on convergent validity. *J. Environ. Econ. Manag.* 66, 90–104. doi: 10.1016/j.jeem.2013.03.001

Kling, C. L., Phaneuf, D. J., and Zhao, J. (2012). From Exxon to BP: has some number become better than no number? *J. Econ. Perspect.* 26, 3–26. doi: 10.1257/jep.26.4.3

Kontogianni, A., Luck, G. W., and Skourtos, M. (2010). Valuing ecosystem services on the basis of service-providing units: a potential approach to address the 'endpoint problem' and improve stated preference methods. *Ecol. Econ.* 69, 1479–1487. doi: 10.1016/j.ecolecon.2010.02.019

Kontogianni, A., Tourkolias, C., Machleras, A., and Skourtos, M. (2012). Service providing units, existence values, and the valuation of endangered species: a methodological test. *Ecol. Econ.* 79, 97–104. doi: 10.1016/j.ecolecon.2012.04.023

Kotchen, M. J., and Reiling, S. D. (1998). Estimating and questioning economic values for endangered species: an application and discussion. *Endangered Species Update* 15, 77–83.

Kotchen, M. J., and Reiling, S. D. (2000). Environmental attitudes, motivations, and contingent valuation of nonuse values: a case study involving endangered species. *Ecol. Econ.* 32, 93–107. doi: 10.1016/S0921-8009(99)00069-5

Ladenburg, J., and Olsen, S. B. (2014). Augmenting short cheap talk scripts with a repeated Opt-Out reminder in choice experiment surveys. *Res. Energy Econ.* 37, 39–63. doi: 10.1016/j.reseneeco.2014.05.002

Langford, I. H., Kontogianni, A., Skourtos, M. S., Georgiou, S., and Bateman, I. J. (1998). Multivariate mixed models for open-ended contingent valuation data. *Environ. Resour. Econ.* 12, 443–456. doi: 10.1023/A:1008286001085

Langford, I. H., Skourtos, M. S., Kontogianni, A., Day, R. J., Georgiou, S., and Bateman, I. J. (2001). Use and nonuse values for conserving endangered species: the case of the Mediterranean monk seal. *Environ. Plan. A* 33, 2219–2233. doi: 10.1068/a348

Larson, D. M., and Shaikh, S. L. (2004). Recreation demand choices and revealed values of leisure time. *Econ. Inquiry* 42, 264–278. doi: 10.1093/ei/cbh059

Larson, D. M., Shaikh, S. L., and Layton, D. F. (2004). Revealing preferences for leisure time from stated preference data. *Am. J. Agric. Econ.* 86, 307–320. doi: 10.1111/j.0092-5853.2004.00580.x

Layton, D., Brown, G., and Plummer, M. (2001). *Valuing Multiple Programs to Improve Fish Populations*. Working Paper, University ofWashington. Seattle, WA.

Layton, D. F., and Levine, R. A. (2005). "Bayesian approaches to modeling stated preference data," in *Applications of Simulation Methods in Environmental and Resource Economics*, eds R. Scarpa and A. Alberini (Dordrecht: Springer), 187–208. doi: 10.1007/1-4020-3684-1_10

Leslie, H. M., and McLeod, K. L. (2007). Confronting the challenges of implementing marine ecosystem-based management. *Front. Ecol. Environ.* 5, 540–548. doi: 10.1890/060093

Lew, D. K., Layton, D. F., and Rowe, R. D. (2010). Valuing enhancements to endangered species protection under alternative baseline futures: the case of the Steller Sea Lion. *Mar. Resour. Econ.* 25, 133–154. doi: 10.5950/0738-1360-25.2.133

Lew, D. K., and Wallmo, K. (2011). External tests of embedding and scope in stated preference choice experiments: an application to endangered species valuation. *Environ. Resour. Econ.* 48, 1–23. doi: 10.1007/s10640-010-9394-1

Liebe, U., Meyerhoff, J., and Hartje, V. (2012). Test-retest reliability of choice experiments in environmental valuation. *Environ. Resour. Econ.* 53, 389–407. doi: 10.1007/s10640-012-9567-1

Lindhjem, H., and Navrud, S. (2008). How reliable are meta-analyses for international benefit transfers? *Ecol. Econ.* 66, 425–435. doi: 10.1016/j.ecolecon.2007.10.005

Lindhjem, H., and Tuan, T. H. (2012). Valuation of species and nature conservation in Asia and Oceania: a meta-analysis. *Environ. Econ. Policy Stud.* 14, 1–22. doi: 10.1007/s10018-011-0019-x

List, J. A., and Gallet, C. A. (2001). What experimental protocol influence disparities between actual and hypothetical stated values? *Environ. Resour. Econ.* 20, 241–254. doi: 10.1023/A:1012791822804

List, J. A., Sinha, P., and Taylor, M. H. (2006). Using choice experiments to value non-market goods and services: evidence from field experiments. *Adv. Econ. Anal. Policy* 6, 1–37. doi: 10.2202/1538-0637.1132

Loomis, J. (1996). Measuring the economic benefits of removing dams and restoring the Elwha river: results of a contingent valuation survey. *Water Resour. Res.* 32, 441–447. doi: 10.1029/95WR03243

Loomis, J. B. (1992). The evolution of a more rigorous approach to benefit transfer: benefit function transfer. *Water Resour. Res.* 28, 701–705. doi: 10.1029/91WR02596

Loomis, J. B., and Larson, D. M. (1994). Total economic values of increasing gray whale populations: results from a contingent valuation survey of visitors and households. *Mar. Resour. Econ.* 9, 275–286.

Loomis, J. B., and White, D. S. (1996). Economic benefits of rare and endangered species: summary and meta-analysis. *Ecol. Econ.* 18, 197–206. doi: 10.1016/0921-8009(96)00029-8

Loomis, J. B., Yorizane, S., and Larson, D. M. (2000). Testing significance of multi-destination and multi-purpose trip effects in a travel cost method demand model for whale watching trips. *Agric. Resour. Econ. Rev.* 29, 183–191. Available online at: http://purl.umn.edu/31308

Lusk, J. L., and Norwood, F. B. (2005). The effect of experimental design on choice-based conjoint valuation estimates. *Am. J. Agric. Econ.* 87, 771–785. doi: 10.1111/j.1467-8276.2005.00761.x

Lusk, J. L., and Schroeder, T. C. (2004). Are choice experiments incentive compatible? A test with quality differentiated beef steaks. *Am. J. Agric. Econ.* 86, 467–482. doi: 10.1111/j.0092-5853.2004.00592.x

Lyssenko, N., and Martinez-Espineira, R. (2006). Respondent uncertainty in contingent valuation: the case of whale conservation in Newfoundland and Labrador. *Appl. Econ.* 44, 1911–1930. doi: 10.1080/00036846.2011.556590

Mansfield, C. (1999). Despairing over disparities: explaining the difference between willingness to pay and willingness to accept. *Environ. Resour. Econ.* 13, 219–234. doi: 10.1023/A:1008246228773

Martín-López, B., Montes, C., and Benayas, J. (2008). Economic valuation of biodiversity conservation: the meaning of numbers. *Conserv. Biol.* 22, 624–635. doi: 10.1111/j.1523-1739.2008.00921.x

McConnell, K. E. (1983). "Existence and bequest value," in *Managing Air Quality and Scenic Resources at National Parks and Wilderness Areas*, eds R. D. Rowe and L. G. Chestnut (Boulder, CO: Westview Press), 254–264.

McConnell, K. E. (1992). Model building and judgment: implications for benefit transfers with travel cost models. *Water Resour. Res.* 28, 695–700. doi: 10.1029/91WR02595

McConnell, K. E., Strand, I. E., and Valdes, S. (1998). Testing temporal reliability and carry-over effect: the role of correlated responses in test-retest reliability studies. *Environ. Resour. Econ.* 12, 357–374. doi: 10.1023/A:1008264922331

McVittie, A., and Hussain, S. S. (2013). *The Economics of Ecosystems and Biodiversity - Valuation Database Manual.* The Economics of Ecosystems and Biodiversity (TEEB). Available online at: http://doc.teebweb.org/wp-content/uploads/2014/03/TEEB-Database-and-Valuation-Manual_2013.pdf

Medina, C., Aravena, C., and Vasquez, F. (2012). *Valoracion Economica de la Conservacion de Tiburones en la Reserva Marina de Galapagos*. Latin American and Caribbean Environmental Economics Program Working Paper Series No. WP34.

Meyerhoff, J., Oehlmann, M., and Weller, P. (2015). The influence of design dimensions on stated choices in an environmental context. *Environ. Resour. Econ.* 61, 385–407. doi: 10.1007/s10640-014-9797-5

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-Being: Synthesis.* Washington, DC: Island Press.

Moeltner, K., Boyle, K. J., and Paterson, R. W. (2007). Meta-analysis and benefit transfer for resource valuation – addressing classical challenges with Bayesian modeling. *J. Environ. Econ. Manag.* 53, 250–269. doi: 10.1016/j.jeem.2006.08.004

Murphy, J. J., Allen, P. G., Stevens, T. H., and Weatherhead, D. (2005). A meta-analysis of hypothetical bias in stated preference valuation. *Environ. Resour. Econ.* 30, 313–325. doi: 10.1007/s10640-004-3332-z

National Research Council (2005). *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. New York, NY: The National Academies Press.

Navrud, S., and Ready, R. (eds.). (2007). *Environmental Value Transfer: Issues and Methods*. Dordrecht: Springer. doi: 10.1007/1-4020-5405-X

NOAA (1996). *Guidance Document for Natural Resource Damage Assessment Under the Oil Pollution Act of 1990.* Report for the National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program.

Ojea, E., and Loureiro, M. L. (2010). Valuing the recovery of overexploited fish stocks in the context of existence and option values. *Mar. Policy* 34, 514–521. doi: 10.1016/j.marpol.2009.10.007

Olsen, D., Richards, J., and Scott, R. D. (1991). Existence and sport values for doubling the size of Columbia River Basin Salmon and Steelhead Runs. *Rivers* 2, 44–56.

Ready, R. C., Champ, P. A., and Lawton, J. L. (2010). Using respondent uncertainty to mitigate hypothetical bias in a stated choice experiment. *Land Econ.* 86, 363–381. doi: 10.3368/le.86.2.363

Ressurreicao, A., Gibbons, J., Dentinho, T. P., Kaiser, M., Santos, R. S., and Edwards-Jones, G. (2011). Economic valuation of species loss in the open sea. *Ecol. Econ.* 70, 729–739. doi: 10.1016/j.ecolecon.2010.11.009

Ressurreicao, A., Gibbons, J., Kaiser, M., Dentinho, T. P., Zarzycki, T., Bentley, C., et al. (2012). Different cultures, different values: the role of cultural variation in public's WTP for marine species conservation. *Biol. Conserv.* 145, 148–159. doi: 10.1016/j.biocon.2011.10.026

Richardson, L., and Loomis, J. (2009). The total economic value of threatened, endangered, and rare species: an updated meta-analysis. *Ecol. Econ.* 68, 1535–1548. doi: 10.1016/j.ecolecon.2008.10.016

Roach, B., and Wade, W. W. (2006). Policy evaluation of natural resource injuries using habitat equivalency analysis. *Ecol. Econ.* 58, 421–433. doi: 10.1016/j.ecolecon.2005.07.019

Rosenberger, R., and Loomis, J. B. (2003). "Benefits transfer," in *A Primer on Nonmarket Valuation,* eds A. Patricia Champ, J. Kevin Boyle, and C. Thomas Brown (Dordrecht: Kluwer Academic Publishers), 445–482. doi: 10.1007/978-94-007-0826-6_12

Rosenberger, R., and Phipps, T. (2007). "Correspondence and convergence in benefit transfer accuracy: meta-analytic review of the literature," in *Environmental Value Transfer: Issues and Methods*, eds S. Navrud and R. Ready (Dordrecht: Springer), 23–44.

Rudd, M. A. (2009). National values for regional aquatic species at risk in Canada. *Endangered Species Res.* 6, 239–249. doi: 10.3354/esr00160

Ryan, M., Gerard, K., and Amaya-Amaya, M. (2010). *Using Discrete Choice Experiments to Value Health and Health Care*. Dordrecht: Springer.

Samples, K., Dixon, J., and Gowen, M. (1986). Information disclosure and endangered species valuation. *Land Econ.* 62, 306–312.

Samples, K. C., and Hollyer, J. R. (1990). "Contingent valuation of wildlife resources in the presence of substitutes and complements," in *Economic Valuation of Natural Resources: Issues, Theory, and Applications*, Chapter 11, eds R. Johnson and G. Johnson (Boulder, CO: Westview Press), 177–192.

Sanchirico, J., Lew, D. K., Haynie, A., Kling, D., and Layton, D. F. (2013). Conservation values in marine ecosystem-based management. *Mar. Policy* 38, 523–530. doi: 10.1016/j.marpol.2012.08.008

Shaikh, S. L., and Larson, D. M. (2003). A two-constraint almost ideal demand model of recreation and donations. *Rev. Econ. Stat.* 85, 953–961. doi: 10.1162/003465303772815853

Shrestha, R., Rosenberger, R., and Loomis, J. B. (2007). "Benefit transfer using meta-analysis in recreation economic valuation," in *Environmental Value Transfer: Issues and Methods*, eds S. Navrud and R. Ready (Dordrecht: Springer), 161–178.

Siikamaki, J., and Layton, D. F. (2007). Discrete choice survey experiments: a comparison using flexible methods. *J. Environ. Econ. Manag.* 53, 122–139. doi: 10.1016/j.jeem.2006.04.003

Skourtos, M., Kontogianni, A., and Harrison, P. A. (2010). Reviewing the dynamics of economic values and preferences for ecosystem goods and services. *Biol. Conserv.* 19, 2855–2872. doi: 10.1007/s10531-009-9722-3

Smith, V. K., van Houtven, G., and Pattanayak, S. K. (2002). Benefit transfer via preference calibration: 'Prudential Algebra' for policy. *Land Econ.* 78, 132–152. doi: 10.2307/3146928

Solomon, B. D., Corey-Luse, C. M., and Halvorsen, K. E. (2004). The Florida Manatee and eco-tourism: toward a safe minimum standard. *Ecol. Econ.* 50, 101–115. doi: 10.1016/j.ecolecon.2004.03.025

Stanley, D. L. (2005). Local perception of public goods: recent assessments of willingness-to-pay for endangered species. *Contemp. Econ. Policy* 23, 165–179. doi: 10.1093/cep/byi013

Stevens, T. H., Echeverria, J., Glass, R. J., Hager, T., and More, T. A. (1991). Measuring the existence value of wildlife: what do CVM estimates really show? *Land Econ.* 67, 390–400. doi: 10.2307/3146546

Stithou, M., and Scarpa, R. (2012). Collective versus voluntary payment in contingent valuation for the conservation of marine biodiversity: an exploratory study from Zakynthos, Greece. *Ocean Coastal Manag.* 56, 1–9. doi: 10.1016/j.ocecoaman.2011.10.005

The Economics of Ecosystems and Biodiversity (2011). *The Economics of Ecosystems and Biodiversity in National and International Policy Making,* ed P. Ten Brink (London; Washington, DC: Earthscan), 429. Available online at: http://doc.teebweb.org/wp-content/uploads/2014/04/TEEB-in-national-and-international-Policy-Making2011.pdf

Tisdell, C., and Wilson, C. (2002). Ecotourism for the survival of sea turtles and other wildlife. *Biodivers. Conserv.* 11, 1521–1538. doi: 10.1023/A:1016833300425

Tkac, J. (1998). The effects of information on willingness-to-pay values of endangered species. *Am. J. Agric. Econ.* 80, 1214–1220. doi: 10.2307/1244227. Available online at: http://www.jstor.org/stable/1244227

Tuncel, T., and Hammitt, J. K. (2014). A new meta-analysis on the WTP/WTA disparity. *J. Environ. Econ. Manag.* 68, 175–187. doi: 10.1016/j.jeem.2014.06.001

Unsworth, R. E., and Petersen, T. B. (1995). *A Manual for Conducting Natural Resource Damage Assessments: The Role of Economics.* Division of Economics, U.S. Fish and Wildlife Service, U.S. Department of Interior: Washington, DC.

U.S. General Accounting Office (1994). *Ecosystem Management: Additional Actions Needed to Adequately Test a Promising Approach*. Report GAO/RCED-94–111.

Vianna, G. M. S., Meekan, M. G., Pannell, D. J., Marsh, S. P., and Meeuwig, J. J. (2012). Socio-economic value and community benefits from shark-diving tourism in Palau: a sustainable use of reef shark population. *Biol. Conserv.* 145, 267–277. doi: 10.1016/j.biocon.2011.11.022

Vossler, C. A., and Kerkvliet, J. (2003). A criterion validity test of the contingent valuation method: comparing hypothetical and actual voting behavior for a public referendum. *J. Environ. Econ. Manag.* 45, 631–649. doi: 10.1016/S0095-0696(02)00017-7

Wallmo, K., and Lew, D. K. (2011). Valuing improvements to threatened and endangered marine species: an application of stated preference choice experiments. *J. Environ. Manag.* 92, 1793–1801. doi: 10.1016/j.jenvman.2011.02.012

Wallmo, K., and Lew, D. K. (2012). Public values for recovering and downlisting threatened and endangered marine species. *Conserv. Biol.* 26, 830–839. doi: 10.1111/j.1523-1739.2012.01899.x

Wallmo, K., and Lew, D. K. (2015). Public preferences for endangered species recovery: an examination of geospatial scale and non-market values. *Front. Mar. Sci.* 2:55. doi: 10.3389/fmars.2015.00055

Whitehead, J. C. (1991). "Economic values of threatened and endangered wildlife: a case study of coastal nongame wildlife," in *Transactions of the 57th North American Wildlife and Natural Resources Conference* (Washington, DC: Wildlife Management Institute).

Whitehead, J. C. (1992). Ex ante willingness to pay with supply and demand uncertainty: implications for valuing a sea turtle protection programme. *Appl. Econ.* 24, 981–988. doi: 10.1080/00036849200000075

Whitehead, J. C., Pattanayak, S. K., van Houtven, G. L., and Gelso, B. R. (2008). Combining revealed and stated preference data to estimate the nonmarket value of ecological services: an assessment of the state of the science. *J. Econ. Surveys* 22, 872–908. doi: 10.1111/j.1467-6419.2008.00552.x

Wilson, C., and Tisdell, C. (2003). Conservation and economic benefits of wildlife-based marine tourism: sea turtles and whales as case studies. *Hum. Dimens. Wild. Int. J.* 8, 49–58. doi: 10.1080/10871200390180145

Yaffee, S. L. (1996). Ecosystem management in practice: the importance of human institutions. *Ecol. Appl.* 6, 724–727. doi: 10.2307/2269472

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