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Methods

Indices of biotic integrity in stated preference valuation of aquatic ecosystem services

Robert J. Johnston [a,*], Kathleen Segerson [b], Eric T. Schultz [c], Elena Y. Besedin [d], Mahesh Ramachandran [e]

[a] George Perkins Marsh Institute and Department of Economics, Clark University, 950 Main St. Worcester, MA 01610, United States
[b] Department of Economics, University of Connecticut, Storrs, CT 06269, United States
[c] Department of Ecology and Evolutionary Biology, University of Connecticut, Storrs, CT 06269, United States
[d] Abt Associates Inc., 55 Wheeler Street, Cambridge, MA 02138, United States
[e] George Perkins Marsh Institute, Clark University, 950 Main St. Worcester, MA 01610, United States

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ABSTRACT

Stated preference surveys often give minimal attention to distinctions between intermediate and final ecosystem services, leading to the potential for welfare estimates that overlook, misrepresent or double count associated values. This paper illustrates potential mechanisms through which multimetric indexes of the type developed in the ecological literature, here an index of biotic integrity, can be used within stated preference survey scenarios to both improve the validity of survey responses and provide otherwise unavailable information on willingness to pay for intermediate and final ecosystem services. We illustrate the approach using a choice experiment application to the restoration of migratory fish in a Rhode Island watershed. Where assumptions of the model hold, results can allow transparent disentanglement and estimation of marginal values for both intermediate and final ecosystem services.

1. Introduction

Incorporating benefits or costs associated with changes in ecosystem goods and services (henceforth, "services") into policy analysis requires some way of estimating the value of these changes, or quantifying implications for human welfare. Economics provides a range of market and nonmarket methods that, when coordinated appropriately with ecological data, may be used to estimate these values (Bateman et al., 2011; Freeman, 2003; Hanley and Barbier, 2009; US EPA, 2009). However, unlike other commodities that may be evaluated using these methods, valuation of ecosystem service flows requires attention to often complex relationships which link ecological outcomes. Among the related issues that must be addressed, consistent estimates of ecosystem service benefits require one to distinguish intermediate ecosystem functions or outcomes from final ecosystem services (Boyd and Banzhaf, 2007; Brown et al., 2007; Fisher et al., 2009; Johnston and Russell, in press; Kontogianni et al., 2010; Turner and Daily, 2008; Wallace, 2007). Such issues are particularly germane for regulating or supporting services, which are often of primary concern to ecologists and policy-makers, but do not provide benefits directly.[1]

In standard economic parlance, regulating, supporting and other indirect services are akin to intermediate services; these can be viewed as inputs into the biophysical production of final services (Boyd and Krupnick, 2009; Brown et al., 2007; Fisher et al., 2009; Turner and Daily, 2008). Final ecosystem goods and services, in contrast, are defined as ecosystem outputs that directly enhance the utility of humans by providing either use or nonuse benefits. Intermediate services can predominate in particular ecosystem components; failure to recognize their contributions and value can lead to misguided policy. On the other hand, summation of willingness to pay (WTP) for both intermediate and associated final services is also misleading, because it double counts the contribution of the intermediate services to welfare (de Groot et al., 2002; Fisher et al., 2009; Wallace, 2007). Fisher et al. (2009), for example, point out that aggregation of values associated with the different ecosystem service categories in the Millennium Ecosystem Assessment (MEA, 2005) would double count the value of many services.

As a purely conceptual matter, valuation of intermediate services is straightforward. Within economic theory one can value either changes in inputs or the corresponding change in the final output; the value of the change in output reflects the sum of the value of changes in all inputs used in production. However, in the context of ecosystem services, valuing changes in intermediate services can present empirical challenges. For example, the revealed preference literature rarely accounts for the influence of intermediate ecological inputs on final ecosystem goods and services that (directly) influence observed behavior. Hence, these methods generally provide values for final

* Corresponding author. Tel.: +1 508 751 4619; fax: +1 508 751 4600.
E-mail addresses: rjohnston@clarku.edu (R.J. Johnston), kathleen.segerson@uconn.edu (K. Segerson), eric.schultz@uconn.edu (E.T. Schultz), elena_besedin@abtassoc.com (E.Y. Besedin), mramachandran@clarku.edu (M. Ramachandran).

[1] A common example is water purification, a regulating service which can result from nutrient removal in riparian buffers or water filtration in bivalve shellfish (MEA, 2005; Turner and Daily, 2008; US EPA, 2009).

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ecosystem services only.[2] Additional biophysical information is required to estimate values for associated intermediate services; this is often unavailable.

Stated preference (SP) research has also given little attention to distinctions between intermediate and final services, or more broadly to the definition of ecosystem services, leading to the potential for welfare biases (Boyd and Krupnick, 2009; Fisher et al., 2009). Welfare measures may be biased for at least two reasons. First, respondents asked to value changes in an intermediate input may not be aware of resulting impacts on valued final ecosystem services, leading to statements of WTP that do not reflect welfare contributions of the intermediate service. Second, even if respondents recognize a relationship between an intermediate and final service, they may have an incorrect understanding of the ecological production functions that link the two.[3] Resulting WTP estimates will be based at least in part on these incorrect assumptions (Johnston et al., in press).

Such challenges have led some to argue that economic valuation methods, including SP methods, cannot be used reliably to value changes in intermediate ecosystem services. For example, Turner et al. (1997) argue that estimating the value of life support systems in monetary terms is difficult because “the general public seldom has information about the support functions of ecosystems” (p. 135). Likewise, Limburg and Folke, (1999) state that

> In an ecologically illiterate world, humans do not always perceive their indirect dependence on critical ecosystem services and support. Even if they do, they may not value it… Therefore, economic valuation based on an aggregation of preferences may only capture part of the ecological prerequisites for social and economic development (Costanza and Folke, 1997) (p. 180).

These challenges can be particularly acute when respondents have little experience with the goods or services in question (Bateman et al., 2011). A primary concern is that lack of understanding of the role of intermediate services will lead individuals to state a WTP for these services that does not appropriately reflect the role these services play in providing final ecosystem services.

The goal of this paper is to explore practical approaches for the treatment of intermediate ecosystem services within SP valuation, directly addressing skepticism regarding the potential role of SP methods in valuing these services. Specifically, we explore potential ways in which multimetric ecological indexes – here an index of biotic integrity (IBI) – can be used within economic frameworks to (1) communicate the role of intermediate services within SP surveys, (2) formally distinguish between intermediate and final services, and (3) help quantify associated WTP. We begin with theoretical frameworks that formally distinguish attributes of the ecosystem that are valued as final services versus those valued as inputs into the production of final services. These frameworks inform SP designs using ecological indicators to provide welfare estimates for final and intermediate services. The developed approaches also help avert problems in which SP scenarios require respondents to speculate regarding ecological production functions (Johnston et al., in press). We illustrate the approach and its usefulness, as well as its potential limitations, using an application to migratory fish passage restoration in a Rhode Island watershed.

2. Conceptual Framework

The distinction between intermediate and final services can be illustrated using a simple utility framework. Suppose an individual has a utility function of the general form $U(X,Y(X,Z))$, where X and Y are measures of ecosystem functions or conditions that influence welfare; we refer to these here as ecosystem attributes. This specification allows ecological attribute X to affect utility directly (and hence be a final service) as well as indirectly through its contribution to the production of a final service Y through biophysical production function $Y(X,Z)$. For example, X could represent the population of a species or organism which might be valued directly (e.g., have existence value) and also play an ecological role in sustaining other valued ecosystem services. In contrast, Z only affects utility though its influence on Y; it is solely an intermediate service.[4]

The marginal utility of a change in X is $\mathrm{d}U/\mathrm{d}X = U_X + U_Y \cdot \partial Y/\partial X$, reflecting both the direct and indirect effect of the change. From a theoretical perspective, WTP estimated using SP methods provides a money metric of this change. There are, however, empirical considerations that can render $\mathrm{d}U/\mathrm{d}X$ and related WTP difficult to quantify. Challenges for valuation are particularly severe if researchers do not clarify relationships between X and Y in survey instruments. For example, an individual without ecological expertise might not be aware of the relationship $Y(X,Z)$ through which changes in X influence Y. In this case, if asked to directly value a change in X, WTP would omit $U_Y \cdot \partial Y/\partial X$, because the individual incorrectly assumes $\partial Y/\partial X = 0$. Estimated WTP hence provides a biased measure of welfare change. If X were to provide no direct utility ($U_X = 0$), for example, stated WTP would be zero, even though the change in X affects the individual's utility through Y.

Perhaps a more likely outcome is that non-expert respondents presented with information on X will, in the absence of information on Y, condition responses on erroneous speculations regarding the true ecological production function $Y(X, Z)$. That is, they will assume an incorrect relationship between X and Y. The result is again bias in estimated WTP for X. As noted by Carson (1998, p. 23) for the case of stated preference survey elicitation in general, “[r]espondents will tend to fill in whatever details are missing in the … survey with default assumptions. These may differ considerably from what the researcher perceives.” This is particularly true for ecological resources and functions, for which respondents may have little baseline information (Bateman et al., 2011; Johnston et al., in press; Spash and Hanley, 1995).

As suggested by the simple model above, biophysical relationships between ecological attributes, combined with the role of these attributes in individuals' utility, can have important implications for the design of valuation methods and validity of associated welfare measures. Despite this, it is not uncommon for surveys in the SP literature to combine final and intermediate services within scenarios in a seemingly ad hoc manner, without information that could assist respondents in disentangling associated biophysical relationships.

One solution is to structure survey scenarios to provide information only on final services (Boyd and Krupnick, 2009). Once WTP estimates are calculated for Y, one could apply biophysical production relationships *ex post* to calculate WTP for the associated intermediate service X. This approach is only valid, however, when the direct effect of X on utility is zero. Moreover, lack of information on X may cause some respondents to speculate as to whether there are additional effects of presented policy scenarios related to changes in X that have been omitted in SP scenarios—leading to potential confusion. Finally, if the primary purpose of the valuation effort is to estimate WTP for changes in X, but survey scenarios make no mention of X (i.e., only

[2] An exception is ecological productivity methods, in which empirical welfare estimates are grounded in an explicit model of ecological production (e.g., Johnston et al., 2002a,b).

[3] This is akin to asking an individual to value a change in the labor or capital that is used in the production of a good or service, rather than a change in the output of the valued good or service that results from the input change.

[4] More complex models are required when utility is derived through market goods produced through a combination of ecosystem services with other inputs such as manufactured and human capital (Bateman et al., 2011; Brown et al., 2007). As SP surveys rarely address such market goods, we retain a simpler utility function for illustration purposes in which ecosystem services enter the utility function directly.

present information on Y), researchers could be accused of the same lack of transparency in SP research decried by Morrison (2003). For such reasons, researchers may wish to design survey scenarios that – at least in some cases – clarify relationships between intermediate and final services and allow values for the two to be disentangled in a more transparent manner.

3. Using Ecological Indices to Characterize Linkages Among Services

One possibility that addresses the above concerns is to present both X and Y in an SP scenario, but provide respondents with information on biophysical relationships between the two. This approach can be informationally burdensome, especially when X contributes to multiple final services or the link between X and Y is complex. There are, however, a variety of multimetric indices that have been developed in the ecological literature to summarize changes in ecological systems (Barbour et al., 1995), including relationships between overall index values (that can be used to quantify final services) and ecological attributes that serve as index components (that can be used to quantify intermediate services). When used appropriately within SP survey scenarios, such indices may in some instances promote survey responses that better reflect relationships between intermediate and final services. While doing so involves challenges, the judicious use of these indices may provide a heretofore unexplored approach to enhance the information on intermediate and final service values provided by SP methods.

Indicators combined into multimetric ecological indices (or bioindices) are typically drawn from multiple levels of biological organization and communicate the status of a wide range of attributes. The most widely-used multimetric index for aquatic systems is the Index of Biotic Integrity (IBI), designed to represent "the ability to support and maintain [a] community of organisms having a species composition, diversity and functional organization comparable to that of natural habitat of the region" (Jordan and Smith, 2005, p. 468, citing Karr et al., 1987; cf. Karr 1981; 1991). In lay terms, it characterizes the overall condition or naturalness of an ecosystem relative to an undisturbed referent. An IBI typically includes numerous indicators of species composition, trophic role, reproductive strategy and the abundance and/or condition of individual organisms. In the context of ecosystem services, these lower level indicators are akin to intermediate services or outcomes (X or Z) that may in some instances be related to a final service (Y) communicated by the multimetric bioindex. The relationship is determined by the structure of the bioindex. IBIs and similar bioindices are tailored to specific geographic regions and ecological systems (e.g., Bilkovic et al., 2005; Deegan et al., 1997; Mebane et al., 2003; Morgan and Cushman, 2005).[5]

While such indicators are not designed to capture the full complexity of ecological interactions through which intermediate and final services are related, they provide a conceptually simple means to communicate changes in a selected set of ecosystem services and relationships between intermediate and final services. Assume that respondents hold nonuse values for improvements to the ecological condition of an aquatic system, apart from values for other services delivered by that system. Ecological condition thus represents a final service. An appropriately structured IBI can provide an ecologically grounded means to quantify this service within survey scenarios. It can also provide information on the sub-indices that comprise the index, thereby quantifying the intermediate ecological attributes related to ecological condition. Some of these attributes might provide utility both directly and indirectly, whereas others might provide utility only through changes in ecological condition. The use of clearly specified multimetric indices in SP scenarios provides a means to make these relationships transparent, enabling estimation of more informed and defensible stated preferences. While the use of ecosystem condition indexes in economics is not new (Jakus and Shaw, 2003), the authors are aware of no applications to clarify relationships between intermediate and final ecosystem services in SP valuation.

Returning to the utility framework above, the IBI would be equivalent to Y, whereas IBI sub-indices would be equivalent to X and Z. A corresponding SP scenario would include information on final services Y and X, generating associated WTP estimates. The function $Y(X,Z)$ would represent the formal structure of the IBI. It would be described in survey background information using language to encourage respondent comprehension of the index and its use as a measure of ecosystem condition (Johnston et al., in press). This function also includes the information needed for researchers to calculate the marginal impact on Y of a change in any IBI sub-component. For example, although the respondent does not directly value the intermediate service Z (she only values the IBI as a whole), the researcher can calculate $\partial Y/\partial Z$. Combined with WTP for Y estimated directly from SP results, this provides the information necessary to calculate indirect WTP for Z, related solely to the impact of Z on Y.

4. Application: Restoring Fish Passage in Rhode Island

4.1. Empirical Model

We investigate the use of multimetric IBIs for ecosystem service valuation in the context of a choice experiment addressing public preferences for the restoration of migratory fish passage in the Pawtuxet watershed of Rhode Island. This watershed currently provides no spawning habitat for migratory fish; access to all 4347 acres of potential habitat is blocked by 22 dams (Erkan, 2002). The choice experiment questionnaire (*Rhode Island Rivers: Migratory Fishes and Dams*) estimated willingness to pay of Rhode Island residents for options that would provide fish passage to between 225 and 900 acres of historical, but currently inaccessible, habitat. Within this context, restoration of fish passage would not only affect fish populations but also other ecosystem services that rely on the presence or abundance of migratory fish.

The theoretical model is grounded in a random utility framework adapted to reflect the conceptual model above. Here, the utility of household h from option k is

$$U_{hk}(\mathbf{X_k},\mathbf{Y_k}(\mathbf{X_k},\mathbf{Z_k}),I_h-C_{hk}) = v_{hk}(\mathbf{X_k},\mathbf{Y_k}(\mathbf{X_k},\mathbf{Z_k}),I_h-C_{hk}) + \varepsilon_{hk} \qquad (1)$$

where:

$\mathbf{X_k}$ vector characterizing ecosystem outcomes that influence utility directly, as well as through their influence on $\mathbf{Y_k}$;
$\mathbf{Z_k}$ vector characterizing ecosystem outcomes that influence utility only through their influence on $\mathbf{Y_k}$;
$\mathbf{Y_k}(\cdot)$ vector of indicators characterizing utility-relevant ecological outcomes affected by $\mathbf{X_k}$ and $\mathbf{Z_k}$ through function $\mathbf{Y_k}(\mathbf{X_k}, \mathbf{Z_k})$.
I_h disposable income of household h;
C_{hk} mandatory cost to the household of option k;
$v_{hk}(\cdot)$ function representing the empirically measurable component of utility;
ε_{hk} unobservable component of utility, modeled as econometric error.

Mirroring the conceptual framework above, attributes in $\mathbf{X_k}$ influence utility directly and also through their influence on $\mathbf{Y_k}$. $\mathbf{Z_k}$ influences utility only through its effect on $\mathbf{Y_k}$. Within this model,

[5] The development of bioindices in the ecological literature has been accompanied by scrutiny of their efficacy. Guidelines for evaluation have been developed and applied in various monitoring efforts (e.g., Jackson et al., 2000, Naweedi, 2005). These guidelines emphasize: 1) relevance with respect to the ecological endpoints and stressors of concern; 2) feasibility with respect to cost-effective routine data collection; 3) accuracy with respect to sources of measurement and process uncertainty; and, 4) interpretability with respect to the ability to discern changes and to facilitate management decisions. While some have advocated bioindices such as the IBI (e.g., Jordan and Smith, 2005), critiques have pointed out the limitations of such metrics (Suter, 1993, 2001).

the role of an appropriately specified multimetric index would be to communicate changes in selected elements of $\mathbf{Y_k}$ and how these changes are related to $\mathbf{X_k}$ and $\mathbf{Z_k}$.

Given this specification, household h chooses among three policy plans, ($k = A,B,N$), including two restoration options (A, B) and a status quo (N) that includes no restoration and zero household cost (i.e., $C_{hk} = 0$). We assume that household h chooses the option which offers the greatest utility. That is, given Eq. (1), household h will choose plan A if

$$U_{hA}(\mathbf{X_A}, \mathbf{Y_A}(\mathbf{X_A}, \mathbf{Z_A}), I_h - C_{hA}) \geq U_{hk}(\mathbf{X_k}, \mathbf{Y_k}(\mathbf{X_k}, \mathbf{Z_k}), I_h - C_{hk}) \text{for } k = B, N, \quad (2)$$

so that

$$v_{hA}(\mathbf{X_A}, \mathbf{Y_A}(\mathbf{X_A}, \mathbf{Z_A}), I_A - C_{hA}) + \varepsilon_{hA} \geq v_{hk}(\mathbf{X_k}, \mathbf{Y_k}(\mathbf{X_k}, \mathbf{Z_k}), I_h - C_{hk}) + \varepsilon_{hk}. \quad (3)$$

A model of this structure is estimated using methods for discrete outcomes since the analyst does not observe $v_{hk}(\cdot)$ directly, but rather observes households' choices among policy options $k = A,B,N$. Assumptions regarding the unobservable component of utility ε_{hk}, preference heterogeneity, and other model elements determine the most appropriate econometric model, with mixed logit models increasingly common (Hensher and Greene, 2003; McFadden and Train, 2000; Train, 2009).

4.2. Survey Development and Testing

The choice experiment asked respondents to consider alternative options for the restoration of migratory fish passage. Choice scenarios and restoration options were informed by data and restoration priorities in the *Strategic Plan for the Restoration of Anadromous Fishes to Rhode Island Coastal Streams* (Erkan, 2002). Consistent with the strategic plan, the choice experiment addressed restoration methods that neither require dam removal nor would cause appreciable changes in river flows. Fish ladders and fish lifts are the most widespread examples (Schilt, 2007). They are usually designed to permit migratory movements of diadromous fishes between ocean and fresh waters. Fish species that directly benefit from fish passage restoration in this area are alewife (*Alosa pseudoharengus*), blueback herring (*A. aestivalis*), shad (*A. sapidissima*), and American eel (*Anguilla rostrata*). The ecological roles of these species are well understood (Loesch, 1987) and formed the basis for conceptual models linking restoration to valued commodities identified in focus groups. Prior to presenting choice questions, the survey provided information (1) describing the status of Rhode Island river ecology and migratory fish compared to historical baselines, (2) characterizing affected ecological systems and linkages, (3) describing fish passage restoration, and (4) providing definitions, derivations and interpretations of ecological indicators used in survey scenarios.

The questionnaire was developed and tested over 2½ years in a collaborative process involving economists and ecologists (Johnston et al., 2011; in press). This included meetings with resource managers, natural scientists, and stakeholder groups, and 12 focus groups. In addition to survey development and testing in focus groups, individual interviews were conducted with both ecological experts and non-experts. These included cognitive interviews (Kaplowitz et al., 2004), verbal protocols (Schkade and Payne, 1994) and other pretests conducted to gain insight into respondents' interpretation of the questionnaire. Development and testing helped ensure that the survey language and format were easily understood by respondents, that respondents and scientists shared interpretations of terminology and scenarios, and that the survey scenarios captured restoration outcomes viewed as relevant and realistic by both respondents and scientists. Survey language and graphics were pretested carefully to ensure respondent comprehension. Particular attention was given to the definition and interpretation of ecological indicators. Additional details on the general development, description and testing of survey indicators are provided by Johnston et al. (2011; in press).

4.3. Characterizing Restoration Outcomes Using Bioindicators

Choice options are characterized by seven attributes: five ecological indicators, one attribute characterizing public access, and one attribute

Table 1
Choice experiment variables and descriptive statistics.

Variable	Definition	Mean (Std. dev.)[a]
acres	The number of acres of river habitat accessible to migratory fish, presented as a percentage of the established reference value for the Pawtuxet watershed (Erkan, 2002). Range 0–100%.	8.1794 (8.1550)
PVA	Population viability analysis (PVA) score: estimated probability, in percentage terms, that migratory species will continue to appear in the river in 50 years. Reference condition is estimated based from surveys of experts in fish restoration, and interpreted following standard mechanisms for PVA models. Range 0–100%.	33.4413 (28.1265)
catch	The number of catchable-size fish in restored areas, estimated from the number of fish per hour caught by scientific sampling crews. Presented as a percentage of the reference value for the watershed, defined as the highest average level sampled in any Rhode Island river (from Rhode Island Department of Environmental Management sampling data). Range 0–100%.	79.9087 (7.5807)
wildlife	Number of fish-eating species that are common in restored areas, such as egrets, osprey, otters, eagles, turtles and mink. Presented as a percentage of the reference value for the watershed, quantified from surveys of regional experts in wildlife biology. Range 0–100%.	65.0125 (10.3920)
IBI	Index of biotic integrity (IBI) score: A linear multimetric index of aquatic ecological condition following Karr (1981), reflecting the similarity of the restored area to the most undisturbed watershed area in Rhode Island. Index components include overall fish abundance, number of mussel species, number of native fish species, number of sensitive fish species, number of feeding types in fishes, percentage of individual fish that are native, percentage of individual fish that are migratory, and percentage of individual fish that are tumor free. Presented as a percentage of the reference condition. Range 0–100%.	71.6978 (6.0762)
access	Binary (dummy) variable indicating whether the restored area is accessible to the public for walking and fishing; a value of 1 indicates that the public can access the area. Range 0–1.	0.3296 (0.4702)
cost	Household annual cost, described as the mandatory increase in annual taxes and fees required to implement the restoration plan. Household cost for the status quo is zero. Range 0–25.	11.9762 (14.1019)
neither	Alternative specific constant (ASC) associated with the status quo, or a choice of neither plan.	0.3333 (0.4715)

[a] Means and standard deviations include status quo option of no restoration.

characterizing unavoidable household cost (Table 1). Ecological indicators included in each choice option characterize: (1) the quantity of river habitat accessible to migratory fishes (*acres*), based upon restorable Pawtuxet habitat acreage in Erkan (2002); (2) the probability that the restored fish run will exist in 50 years, reflecting results calculable through applications of population viability analysis (*PVA*)[6]; (3) the abundance of fish suitable for recreational harvest (*catch*), reflecting abundance measures from statewide sampling; (4) the abundance of fish-dependent wildlife (*wildlife*), reflecting the appearance of identifiable species within restored areas; and (5) overall ecological condition (*IBI*), reflecting the output of a multimetric aquatic ecological condition score (i.e., an index of biotic integrity). Table 2 illustrates attribute levels included in the choice experiment design.

Each of these attributes was selected based on an underlying conceptual model that coordinated ecological science with findings from focus groups and interviews. The primary direct ecological effect of restoration is to provide migratory fish with access to additional river areas; these effects are captured by the attributes *acres* and *PVA*. Related indirect outcomes include effects on the abundance of other recreationally harvested fish such as largemouth bass (Yako et al., 2000), captured by the attribute *catch*. This attribute is closely linked to use benefits of recreational fishing. These fish are of different species than the migratory fishes that directly benefit from fish passage restoration, but are indirectly affected through species interactions.[7] Fish runs can also support other wildlife populations. Thus, fish passage restoration would likely also increase populations of fish-dependent wildlife, as reflected in the attribute *wildlife*.

Restoration of fish passage can also affect the provision of regulating and supporting ecosystem services that, while not valued directly, contribute to other services that are valued. Among these supporting services are various functions provided by freshwater mussels, including biofiltration (Strayer et al., 1999). Freshwater mussels can also serve as an indicator species for ecosystem condition. Mussels are sensitive to water quality and can serve as biomonitor organisms (McMahon and Bogan, 2001), and many species are sensitive to the presence of dams and migratory fish (Nedeau, 2003; Raithel and Hartenstine, 2006). For example, the mussel species *Anodonta implicata* relies on migratory shad, alewives and blue-back herring to carry its larvae and complete its life cycle. In the absence of these fish species, *A. implicata* disappears from aquatic ecosystems (Nedeau, 2003; Smith, 1985). Hence, restoration of fish passage can both enhance the abundance of freshwater mussels and promote re-population of species that have been extirpated (Nedeau, 2003; Raithel and Hartenstine, 2006).

Although the capacity of freshwater mussels to provide regulating and supporting services is well established, focus groups used in survey development indicated that typical respondents were *not* willing to pay for improvements to these species alone (i.e., in the absence of related changes to other final ecosystem services). Pretests with preliminary versions of the survey including a mussel population attribute revealed similar preference patterns—mussels were valued only as an intermediate service that contributed to other, directly valued outcomes. Moreover, focus groups strongly suggested that respondents were willing to pay for improvements in ecosystem condition or naturalness, even holding other final services constant.

To reflect changes in ecosystem condition related to the presence of freshwater mussels and other measurable sub-indices, we included in survey scenarios the multimetric aquatic ecological condition score *IBI*, reflecting a simple index of biotic integrity. The ecological structure of the score is made explicit within the survey, using common language. The variable was calculated as an unweighted linear combination of eight unimetric sub-indices on a 0–100 relative scale. The reference condition of 100 was calculated based on reference values for sub-indices found in the Wood–Pawcatuck watershed, a comparable watershed considered to be the most pristine in Rhode Island. This provides an empirically grounded means to quantify ecosystem condition, based on methods developed in the ecological literature. Table 3 illustrates the structure of the index, including values for component sub-indices.

Table 2
Attribute levels in choice experiment design.

Variable	Levels
acres	1. 0% (0 acres accessible to fish)[a] 2. 5% (225 acres accessible to fish) 3. 10% (450 acres accessible to fish) 4. 20% (900 acres accessible to fish)
PVA	1. 0% (probability of 50 year fish run survival)[a] 2. 30% (probability of 50 year fish run survival) 3. 50% (probability of 50 year fish run survival) 4. 70% (probability of 50 year fish run survival)
catch	1. 70% (102 fish/hour sampling abundance) 2. 80% (116 fish/hour sampling abundance)[a] 3. 90% (130 fish/hour sampling abundance)
wildlife	1. 55% (20 species common)[a] 2. 60% (22 species common) 3. 70% (25 species common) 4. 80% (28 species common)
IBI	1. 65% (aquatic ecological condition score)[a] 2. 70% (aquatic ecological condition score) 3. 75% (aquatic ecological condition score) 4. 80% (aquatic ecological condition score)
access	1. Public Cannot Walk and Fish in Area[a] 2. Public Can Walk and Fish in Area
cost	1. \$0 (cost to household per year)[a] 2. \$5 (cost to household per year) 3. \$10 (cost to household per year) 4. \$15 (cost to household per year) 5. \$20 (cost to household per year) 6. \$25 (cost to household per year)

[a] Status quo value.

A separate page of the survey was devoted to a description of the index, together with its components, structure and interpretation (Fig. 1). This material was subject to extensive focus group testing to ensure that respondents' interpretations were similar to those of natural scientists. The inclusion of this multimetric index – representing a final ecosystem service in our case study – renders transparent the role of the component sub-indices. The structure of the index, for example, allows one to calculate WTP for freshwater mussels related to their contributions towards the overall index value. This reflects the intermediate contribution of improvements in these species to household welfare.

In summary, choice experiment survey responses allow WTP to be estimated for changes in *IBI* as a multimetric indicator of overall ecosystem condition. Combined with the mathematical structure of the index, this estimate enables researchers to calculate the indirect welfare influence of a change in any index sub-component, realized through its consequent influence on *IBI*. The formal structure and components of *IBI* are included in the survey so that respondents can understand the index (and hence give more informed responses). However, the model does not assume that respondents calculate or possess direct values for *IBI* sub-components, only for *IBI* as a whole reflecting overall ecosystem condition.[8]

[6] For an illustration of population viability analysis applied to diadromous fish, see Lee and Rieman (1997).

[7] For example, some species of diadromous fish compete with juveniles of other valued recreational species and also are prey for the adult stages of the same species. The balance of such positive and negative effects depends on other factors (Yako et al., 2000).

[8] Scenarios neither asked respondents to value the sub-components of *IBI* directly, nor to mathematically link a change in *IBI* to underlying changes in index components. Because any given change in *IBI* could be generated by an infinite number of combinations of sub-index changes, the latter does not lead to a unique solution.

Table 3
Structure of the aquatic ecological condition index (*IBI*).

	Total fish abundance (fish/hour sampled)	Number of mussel species	Number of native fish species	Number of sensitive fish species	Number feeding types	Percent native individuals (fish)	Percent migratory individuals (fish)	Percent tumor free fish	IBI[a]
Reference condition (Pawcatuck River downstream)	372	8	23	6	6	99%	9%	94 to 100%	–
Pawtuxet River cardinal values	320	5	12	5	4	91%	0%	80%	–
Pawtuxet relative values (proportion of reference)	0.86	0.63	0.52	0.83	0.67	0.92	0.00	0.80	0.65

[a] Calculated as the unweighted mean of component relative values.

4.4. Survey Implementation

Attribute levels within the experimental design (Table 2) are grounded in feasible restoration outcomes identified by ecological models, field studies and expert consultations. Choice scenarios represent each ecological attribute in relative terms with regard to upper and lower reference conditions (i.e., best and worst possible in the Pawtuxet) as defined in survey materials. Scenarios also present the cardinal basis

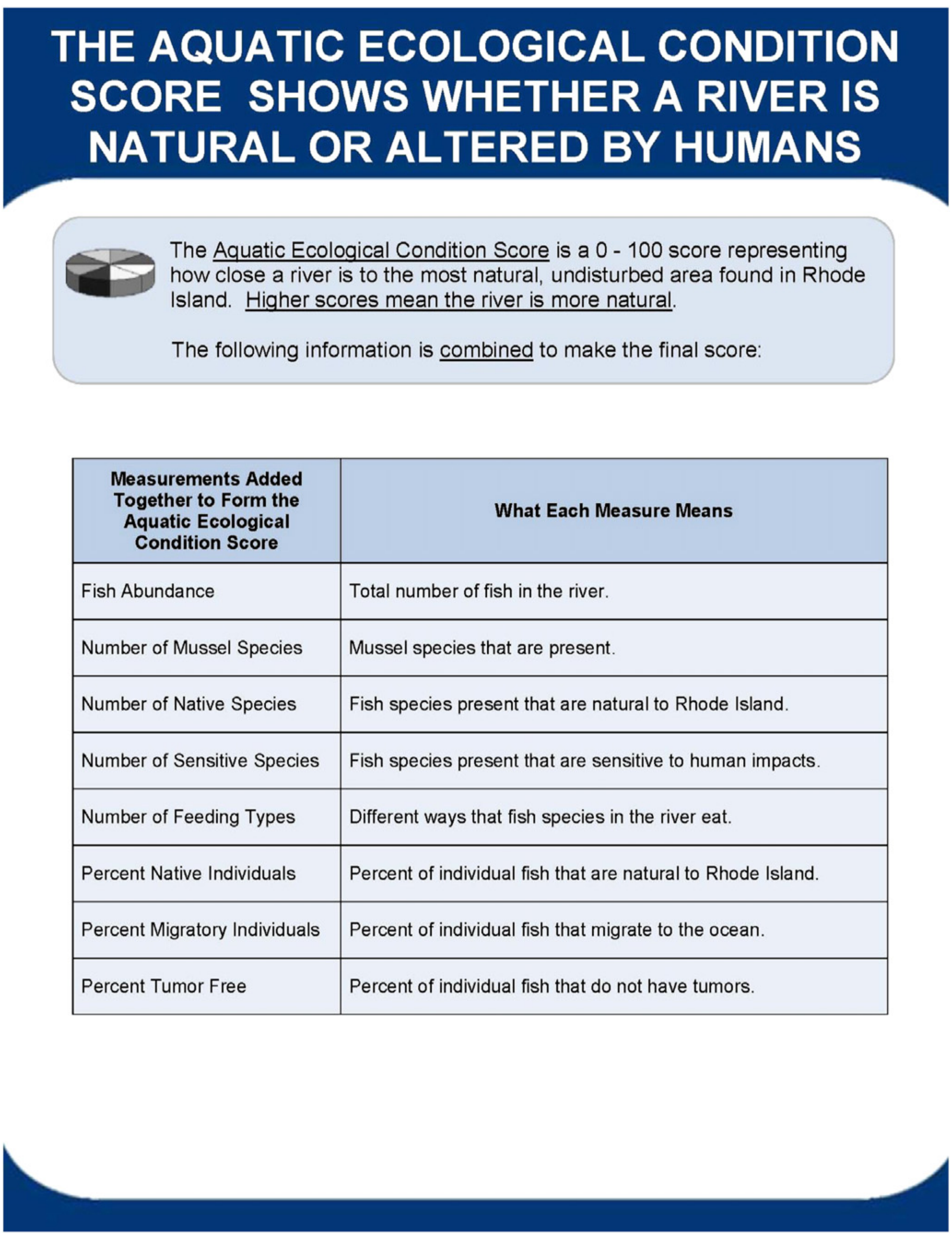

Fig. 1. Survey description of the aquatic ecological condition score.

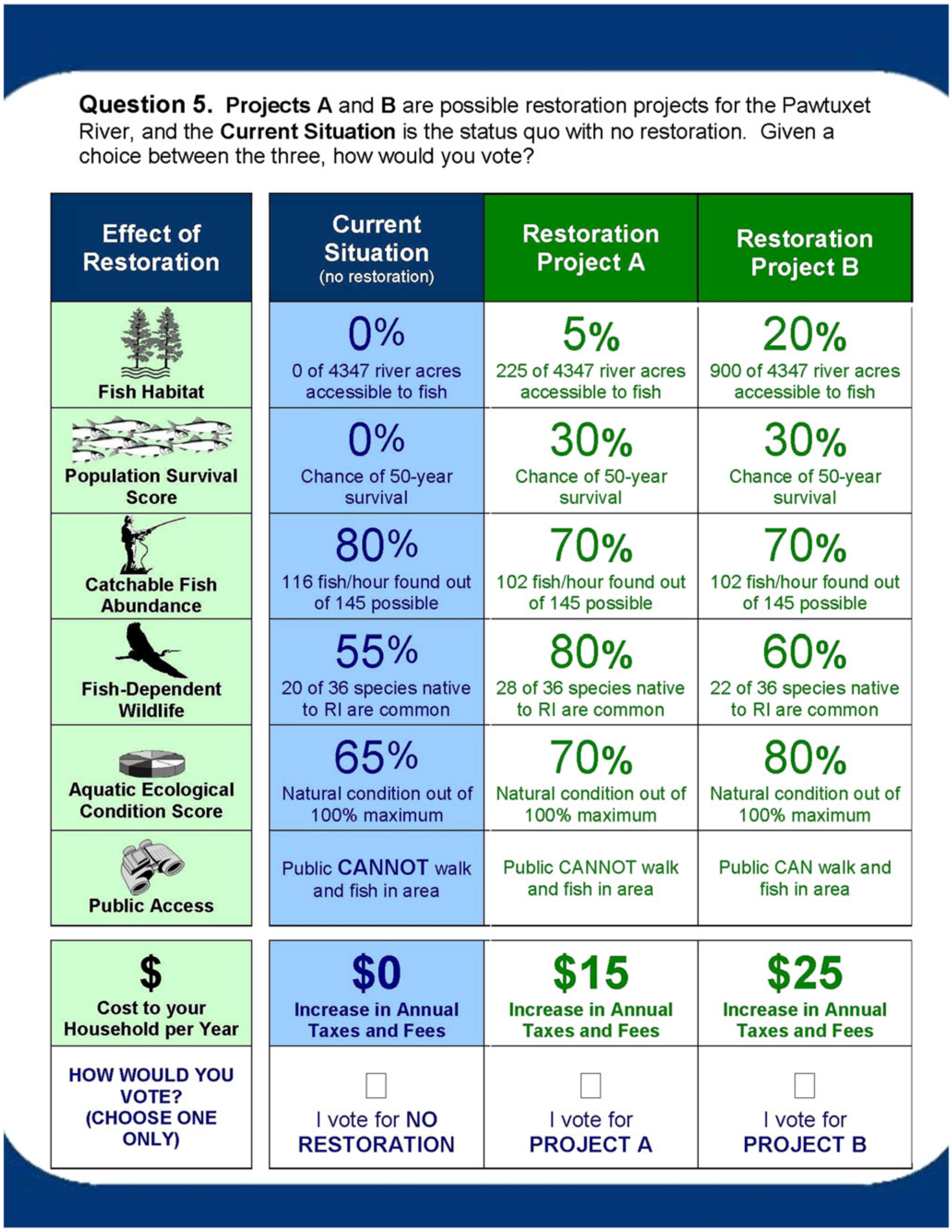

Fig. 2. Sample choice experiment question.

for relative scores where applicable. Relative scores represent percent progress toward the upper reference condition (100%), starting from the lower reference condition (0%). A sample choice question is illustrated in Fig. 2.

Four variants of the survey were designed, of which data from one is analyzed here—the version of the Pawtuxet Watershed survey that includes all ecological attributes.[9] A fractional factorial experimental design was generated using a criterion that minimized D-error for a choice model covariance matrix, assuming a model with both main effects and selected two-way interactions (Kuhfeld, 2010; Kuhfeld and Tobias, 2005). The result was a design of 180 profiles blocked into 60 booklets. Each respondent was provided with three choice experiment questions and instructed to consider each as an independent, non-additive choice. Surveys were implemented using a dual wave phone-mail approach during June–August, 2008. An initial random digit dial sample of Rhode Island households was contacted via telephone and asked to participate in a survey addressing Rhode Island "environmental issues and government programs." Those agreeing to participate were sent the questionnaire via mail, with follow-up mailings to increase response rates (Dillman, 2000). For the version of the Pawtuxet survey analyzed here, a total of 600 questionnaires were sent to Rhode Island residents. The analysis is based on 277 usable returns. These provide 803 completed responses to choice questions.[10]

[9] An alternative version was also implemented that excludes *IBI* but is otherwise identical (Johnston et al., in press).

[10] Although sampling was designed to reduce the potential for self-selection biases, such biases are still possible.

Table 4
Mixed logit results: Pawtuxet restoration choice experiment.

Choice attribute	Coefficient (Std. error)
Random parameters	
acres	0.0487 (0.0138)***
PVA	0.0183 (0.0056)***
IBI	0.0539 (0.0209)***
access	1.2208 (0.2247)***
cost (*bounded triangular*, *sign-reversed*)	0.0623 (0.0085)***
Non-random parameters	
catch	0.0035 (0.0092)
wildlife	0.0280 (0.0095)***
neither	−1.6367 (0.4522)***
Distributions of random parameters[a]	
std. dev. acres	0.0896 (0.0254)***
std. dev. PVA	0.0448 (0.0079)***
std. dev. access	1.5585 (0.3702)***
std. dev. IBI	0.1492 (0.0329)***
spread cost (*bounded triangular*)	0.0623 (0.0085)***
−2 Log Likelihood χ^2	533.62***
Pseudo-R^2	0.30
Observations (*N*)	803

[a] Normal distributions are characterized by standard deviations. Triangular distributions are characterized by the spread.
* $p \leq 0.10$.
** $p \leq 0.05$.
*** $p \leq 0.01$.

Table 5
Marginal implicit prices: empirical distributions.[a]

Variable	WTP	Standard deviation	Percentiles (1%, 99%)	Pr>|t|[b]
acres	1.0910	0.3523	(0.39, 2.03)	<0.01
PVA	0 .4136	0.1462	(0.11, 0.86)	<0.01
catch	0.0688	0.2073	(−0.38, 0.56)	0.72
wildlife	0.6369	0.2088	(0.15, 1.17)	<0.01
IBI	1.1879	0.5017	(0.00, 2.42)	<0.01
access	27.3285	6.0602	(15.87, 43.70)	<0.01

[a] Results reflect the mean over the parameter simulation of mean WTP over the coefficient simulation (see text). Estimates are per household, per year. For all variables except *access*, estimates represent WTP for a one percentage point increase.
[b] P-values are two-tailed, for the null hypothesis of zero WTP.

5. Results

5.1. Overall Results

The random utility model is estimated using simulated likelihood mixed logit (ML) with one hundred Halton draws,[11] with the model specified to account for correlations among the three survey responses from each individual respondent (panel data). The final model specification was chosen after the estimation of preliminary models with varying specifications of fixed and random coefficients.[12] Within the final model, coefficients on *acres*, *PVA*, *access*, and *IBI* are specified as random with a normal distribution. The coefficient on *cost* is specified as random with a bounded triangular distribution, ensuring positive marginal utility of income, with sign-reversal applied prior to estimation (Campbell et al., 2009; Hensher and Greene, 2003). Coefficients on *neither* (the alternative specific constant, or ASC), *catch* and *wildlife* are specified as fixed. Results are reported in Table 4. Coefficients are jointly significant at $p<0.0001$ for both models, with pseudo-$R^2>0.30$. All coefficients except for that on *catch* are statistically significant, as are all estimated standard deviations of random parameter distributions. Signs of coefficients match prior expectations in all instances.

Even before welfare estimates are calculated, the model specification provides a convenient means to interpret results. As detailed in Table 1, all model variables except *access* and *cost* represent percent progress towards the upper reference condition (100%). Hence, model coefficients may be directly compared as the relative weight (or marginal utility) given to a one percentage point change in each attribute. Viewed from this perspective, marginal utility is greatest for changes in overall ecological condition (*IBI*), followed closely by restored acres (*acres*).[13] Changes in marginal utility associated with the probability of fish run survival (*PVA*) and increases in the number of fish-dependent wildlife species (*wildlife*) are statistically significant but less than that associated with *acres* and *IBI*. Marginal utility is lowest, and not statistically significant, for changes in catchable fish abundance (*catch*).

These results are mirrored by WTP estimates. Here, we illustrate implicit prices, or marginal WTP for changes in each attribute. Because the model includes random coefficients with both normal and bounded triangular distributions (cf. Campbell et al., 2009), we estimate these using the welfare simulation described by Johnston and Duke (2007; 2009) following the general framework of Hensher and Greene (2003).[14] We simulate implicit price estimates as the mean over the parameter simulation of mean WTP calculated over the coefficient simulation.

Resulting implicit price distributions are summarized in Table 5, along with p-values for the null hypothesis of zero WTP. For all attributes except *access*, implicit price results are interpreted as WTP for a marginal, one percentage point increase in the indicator, holding all else constant. For *access*, results indicate WTP for the provision of public access in the restored area, relative to the default of no access. Statistical significance levels (p-values) in Table 4 are determined through percentiles on the empirical distributions (Poe et al., 2005). For all attributes except *catch* implicit prices are significant at $p<0.01$.

5.2. Willingness to Pay for Ecosystem Change and Implications for Intermediate Services

Implicit prices demonstrate that respondents were able to distinguish between anticipated welfare effects of different ecological outcomes, with WTP significantly higher for some outcomes (e.g., increases in ecosystem condition) than for others (e.g., increases in catchable fish abundance). We find non-trivial WTP for both direct restoration outcomes (e.g., acres accessible to migratory fish), as well as indirect outcomes (e.g., effects on fish-eating wildlife). Some of the most direct effects of restoration, such as increases in the probability of fish run survival (*PVA*), are associated with modest WTP compared to some

[11] The stability of estimates was tested at both higher and lower numbers of draws; results are stable at 100 draws.
[12] McFadden and Train (2000) describe a formal method that may be used to choose random coefficients.
[13] This excludes *access*, which is measured as a dummy variable and is hence not comparable in this manner.
[14] The procedure begins with a parameter simulation following the parametric bootstrap of Krinsky and Robb (1986), with $R=1000$ draws taken from the mean parameter vector and associated covariance matrix. For each draw, the resulting parameters are used to characterize asymptotically normal empirical densities for fixed and random coefficients. For each of these R draws, a coefficient simulation is then conducted for each random coefficient, with $S=1000$ draws taken from simulated empirical densities. Welfare measures are calculated for each draw, resulting in a combined empirical distribution of $R \times S$ observations from which summary statistics are derived.

indirect restoration effects. This suggests the relevance of indirect ecological outcomes for welfare evaluation, and implies that surveys characterizing policy effects solely in terms of direct effects may omit some of the primary avenues through which restoration influences utility. In addition, model results demonstrate that a population sample of respondents may not always express significant WTP for ecosystem changes related to recreational use (i.e., the coefficient on *catch* is not statistically significant). In this model as in many others involving ecological changes, nonuse values appear to be among the primary motivations for WTP (cf. Brown, 1993; Johnston et al., 2003).

Model results also provide insight into the potential use of a multimetric bioindex (here, *IBI*) to communicate changes in final ecosystem services and relationships to intermediate services. Despite the relative complexity of this bioindex, respondents are willing to pay more per percentage point for enhancements to aquatic ecological condition than for any other attribute; this mirrors focus group results which suggested high values for improvements in overall ecological condition, ceteris paribus. This welfare measure is statistically significant at $p<0.01$ (Table 5). Information in the survey questionnaire also details structural relationships between *IBI* and its component sub-indices. This provides a transparent mechanism whereby WTP for these sub-indices may be calculated from respondents' estimated WTP for *IBI* as a whole. As noted above, for example, focus groups suggested that typical Rhode Island residents do not directly value changes in the population of freshwater mussels in the Pawtuxet watershed. Mussels do not provide a final ecosystem service in our case study and are not included as choice attributes. However, choice experiment results demonstrate that respondents do value associated changes in aquatic ecological condition as reflected in the variable *IBI*. These results, together with the formal structure linking changes in mussels to changes in *IBI* (Table 3), provide a means to quantify welfare gains associated with policy-driven changes in freshwater mussels.

Table 6 illustrates this quantification. The Pawtuxet watershed currently supports a status quo of five freshwater mussel species (Raithel and Hartenstine, 2006). At best, fish restoration could increase the number of resident mussel species to eight; this is the reference condition representing the greatest number of mussel species found in any Rhode Island watershed. The structure of the *IBI* summarized in the survey (Table 3; Fig. 1) provides a monotonic, linear relationship between the number of mussel species and the *IBI* score. Holding all else constant, a single species increase in freshwater mussels is associated with a 1.56 percentage point change in *IBI*. Given implicit prices in table 5, per household WTP for this change is \$1.86. This welfare estimate reflects the contribution of mussel changes to valued *IBI* improvements (Table 6). Similar estimates could be calculated for any of the sub-indices used to calculate the final ecological condition score. For example, an increase in the number of native fish species in the Pawtuxet River from 12 to 15 causes an *IBI* increase of 1.63 and an associated WTP of \$1.93. This again reflects the intermediate contribution of native fish species to the welfare gain associated with ecosystem condition improvement.

Aside from facilitating the calculation of WTP for ecological outcomes that affect welfare only indirectly, the information provided on the structure of the multimetric index within questionnaires provides transparency to these calculations. Although respondents only value *IBI* as a composite whole, they are informed that outcomes for the *IBI* are related to changes in the underlying sub-indices. Hence, associated welfare calculations for these intermediate outcomes are grounded in information available to respondents; respondents are aware that their choices imply some value for intermediate outcomes such as changes in freshwater mussel populations, even though these changes are not quantified, valued directly, or included as choice attributes.

Table 6
Willingness to pay for changes in intermediate ecosystem services of freshwater mussels (realized through ecosystem condition, *IBI*).

Change in mussel species	Associated change in *IBI*	WTP per household
0 (5 species baseline)	0	0
+1 (from 5 to 6 species)	1.56	1.86
+2 (from 5 to 7 species)	3.13	3.71
+3 (from 5 to 8 species)	4.69	5.57

Provision of information on the components and structure of the multimetric index also discourages speculation that might otherwise occur with ambiguously defined attributes (Johnston et al., in press). In the absence of details on such effects and related ecosystem services, respondents might "fill in whatever details are missing … with default assumptions" (Carson, 1998, p. 23). Some of these default assumptions might involve erroneous speculations regarding ecological production functions, with a concomitant likelihood of bias in survey responses and welfare estimates. That is, the potential use of appropriately specified multimetric bioindices may not only provide mechanisms to investigate WTP for intermediate ecological outcomes that might otherwise be obscured, it also can serve to enhance the validity of survey responses and associated welfare estimates for final ecosystem services.

There are, however, non-trivial challenges associated with the use of multimetric indices such as IBIs in SP valuation. While such indices can provide a more defensible means to communicate ecological change than the ad hoc indicators typically found in the SP literature (Johnston et al., in press), there is controversy in the ecological literature over the structure and interpretation of multimetric indices, including IBIs. Criticisms include the potential sensitivity of index scores to the choices involving the included sub-indices and mathematical index structure, as well as skepticism over the core concept of ecological integrity (Niemi and McDonald, 2007; Suter, 1993, Suter, 2001). Accordingly, researchers considering the use of such methods in SP survey design should consider the implied ecological assumptions as well as the potential shortcomings of multimetric indices. It is also crucial to conduct appropriate pretesting to ensure that respondents interpret indices as intended, particularly given the complexity of most multimetric indices found in the ecological literature (Davis and Simon, 1995).

The validity of the presented approach also depends on assumptions regarding correspondence between specification of the multimetric index, the structure of respondents' utility, and respondents' perception of choice attributes. These include the assumption that the index as specified – including its components and functional form – accurately represents a final service valued by respondents. In the present case, this also implies constant marginal WTP for index changes that is invariant to the intermediate cause (i.e., which sub-indices caused a change to occur). There is no way to directly validate these assumptions here, beyond qualitative results of focus groups. Researchers considering applications of such methods should be aware of the potential sensitivity of results to these and other underlying assumptions, and ideally integrate elements within surveys through which they might be validated. In the absence of evidence that such assumptions hold, results should be treated with caution.

6. Conclusion

Despite prior efforts to merge ecological and economic information, SP valuation still commonly retains imprecise treatments of ecological information, with little reference to the extensive work by ecologists to quantify ecosystem condition and function. This paper illustrates a potential means through which multimetric ecological indexes of the type developed in the ecological literature can be used to improve the validity of SP survey responses and provide otherwise unavailable information on the value of intermediate ecosystem services. These services, while not valued directly, can have substantial implications for public welfare related to effects on other ecosystem services or outcomes that are valued. Results suggest non-trivial values for these services that might otherwise be overlooked.

This paper addresses only a few of the many challenges involved in comprehensive coordination of economics and ecology for welfare estimation, including ways in which intermediate ecosystem services can (or cannot) be defensibly linked to a potentially large number of associated final services. In addition, there is controversy in the ecological literature regarding the use and interpretation of multimetric indices such as indices of biotic integrity—the validity of any associated welfare measures will of course depend on the properties of these underlying indices. Reported findings are also limited by the policy case study from which they are drawn. Additional verification in other valuation contexts will be required to assess the broader applicability and empirical validity of the proposed methods.

These and other caveats aside, results presented here highlight the benefits of more meaningful collaboration among natural and social scientists, and more specifically the targeted use of multimetric indices to assist in more comprehensive and potentially useful valuation of final and intermediate ecosystem services. Within a carefully designed and tested SP questionnaire, the use of such ecological tools can both enhance the validity of responses by promoting improved respondent understanding of ecological changes, as well as provide mechanisms to estimate values for ecosystem services that might otherwise remain obscured.

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