IMPROVING THE ECOLOGICAL VALIDITY OF NON-MARKET VALUATION:
DEVELOPMENT AND APPLICATION OF BIOINDICATOR-BASED STATED PREFERENCE VALUATION FOR AQUATIC RESTORATION

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Improving the Ecological Validity of Non-Market Valuation: Development and Application of Bioindicator-Based Stated Preference Valuation for Aquatic Restoration

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Abstract: This paper introduces a variant of choice experiment valuation designed to ameliorate common limitations related to a lack of ecological clarity in stated preference survey scenarios. This approach, which we call Bioindicator-Based Stated Preference Valuation (BSPV), is distinguished by guidelines that inform the structure and use of ecological indicators to define policy scenarios in valuation surveys. The approach begins with a formal basis in ecological science and extends to relationships between attributes in respondents’ preference functions and those used to characterize policy outcomes. The resulting welfare measures are designed to be unambiguously linked to models and indicators of ecosystem function, are based on measurable and unambiguous ecological outcomes, and are more easily incorporated into benefit cost analysis. Performance of the developed methods is illustrated through an application to diadromous fish passage restoration in a Rhode Island watershed. Results suggest that less structured treatments of ecological change within stated preference surveys can omit information that is essential for the estimation of nonuse or indirect use values and relevant to respondents when determining WTP.
Introduction

Many recent studies apply stated preference (SP) techniques to assess willingness to pay (WTP) for policies that affect the ecology of aquatic systems (e.g., Bateman et al. 2006; Boyer and Polasky 2004; Flores and Shafran 2006; Hanley et al. 2006; Holmes et al. 2004; Johnston et al. 2002; Loomis et al. 2000; Morrison and Bennett 2004; Morrison et al. 2002). The validity of resulting welfare estimates depends on an appropriate integration of ecological and economic information. To date, however, the integration of ecological information within SP methods has been limited. A key concern is that SP surveys often communicate ecological changes using ad hoc, oversimplified representations of ecological systems (Johnston et al. 2007). Measures of change in aquatic living resources presented in SP surveys, for example, are rarely developed within the context of established models or formal indicators from the ecological literature, and are frequently based on arbitrary, vague or inadequately defined measurement units.

A case in point is the common use of ecologically indistinct terms such as “low,” “medium” or “high” biodiversity (e.g., Carlsson et al. 2003; Garrod and Willis 1997) or “unique” ecological quality (e.g., McGonagle and Swallow 2005; Opaluch et al. 1993). Similarly, Hanley et al. (2006, p. 186) characterize ecological condition using descriptors such as a “wide” versus “poor” “range of water plants, insects and birds,” noting that “none of these attributes are necessarily consistent with what an ecologist would choose in terms of either indicators of the ecological health of a waterbody, or underlying factors driving changes in ecological status.” Although representations of ecological change within SP surveys should reflect the cognitive capacity of survey respondents, weak correspondence between ecological and economic model components can render survey results of limited use for policy evaluation, bias associated welfare measures, and promote skepticism among natural scientists and policy analysts.

This paper introduces a variant of SP valuation designed to address these and other criticisms. This approach, which we call Bioindicator-Based Stated Preference Valuation (BSPV), is distinguished by guidelines that, among other things, inform the structure and use of ecological indicators to define policy scenarios in valuation surveys. Specifically designed for applications to ecological systems, the guidelines underlying BSPV promote ecological clarity and closer integration of ecological and economic information. The approach begins with a formal basis in ecological science and extends to relationships between attributes in respondents’ preference functions and those used to characterize policy outcomes. The resulting welfare measures can be unambiguously linked to models and indicators of ecosystem function, are based on measurable ecological outcomes, and are more easily incorporated into benefit cost analysis. BSPV also provides a means to estimate values for ecological outcomes that individuals might value, even though they may not fully understand all relevant ecological science.

Performance of the developed methods is illustrated through an application to migratory (diadromous) fish passage restoration in a Rhode Island watershed. Results provide sometimes unexpected insights into public preferences for aquatic ecosystem restoration, many of which might be obscured by more traditional approaches to welfare evaluation. These and other findings suggest that less structured treatments of ecological change within SP surveys can omit information that is both essential for the estimation of nonuse or indirect use values and highly relevant to respondents when determining WTP.
Valuation of ecological restoration: Conceptual and theoretical challenges

Stated preference methods, including choice experiments, use carefully designed surveys to estimate public, often non-market values for well-defined changes in the quantity or quality of a bundle of goods and services. Choice experiments present individuals with an opportunity to select one out of a set of available multiattribute options or to reject all of the presented options in favor of the status quo (Adamowicz et al. 1998; Louviere et al. 2000). Choice data over many sets of possible choice options, where each option is defined by component attribute levels, enables the probability of choice to be modeled as a function of attribute levels (Bennett and Blamey 2001). Model results reveal tradeoffs made by respondents when choosing over options; respondents’ willingness to tradeoff non-monetary attributes (e.g., ecological attributes) for changes in household cost provides the theoretical and empirical basis for WTP estimation.

To yield valid estimates of value, SP surveys and estimation methods must have certain features (cf. Bateman et al. 2002; Bennett and Blamey 2001; Louviere et al. 2000). For example, individuals can only make well-informed choices if provided with sufficient information (Hanemann 1994). In the context of ecological restoration, welfare estimates are contingent upon information that allows respondents to predict the expected influence of ecological changes on their welfare (Bingham et al. 1995). One of the associated challenges in the design of surveys for valuation of aquatic ecosystem restoration is characterization of the condition and change in ecosystem function and related impacts on valued goods and services (Flores and Shafran 2006).

Within the following discussion, we distinguish between two terms. The term “attribute” here refers either to an argument in a utility function or to a corresponding element characterizing a stated preference choice option. The term “ecological indicator,” in contrast, refers to a measure that characterizes, and may be used to communicate, the condition of an ecosystem or one of its components (Jackson et al. 2000).

While not made explicit in most SP studies, ecological change is most often communicated through indicators. That is, ecological indicators—either formal or ad hoc—are often used as attributes in choice experiments. Ecologists have developed indicators such as ecological diversity and integrity indices, among many others, to represent the status and function of natural systems (Bortone 2005; Davis and Simon 1995; Jorgensen et al. 2004). Within SP valuation, the role of such indicators is to communicate changes in welfare-relevant qualities or quantities, such that meaningful expressions of value can be elicited (Spash and Hanley 1995). This ecological information must not only be placed in a format that is readily understood by respondents, but that also provides an accurate representation of the change being valued. As stated by Schiller et al. (2001), “effective communication of ecological indicators involve[s] more than simply transforming scientific phrases into easily comprehensible words. [It requires] language that simultaneously fit[s] within both scientists’ and nonscientists’…frames of reference, such that resulting indicators [are] at once technically accurate and understandable.”

In contrast to the care devoted to indicator development and interpretation in the ecological literature, indicators used in SP surveys often lack documented reference to empirical findings regarding ways in which natural systems respond to changes or stresses, and have often
minimal grounding in prior ecological research. As a result, indicators of ecological change presented in SP surveys: (1) rarely correspond to formal indicators presented in the published ecological literature, (2) are often ambiguously linked to quantifiable policy impacts, (3) are often based on arbitrary or vague measurement units, and (4) have often incomplete or ambiguous links to ecosystem goods or services valued by respondents.

For example, SP studies often define policy changes solely in terms of convenient units such as the number of organisms or acres of habitat affected. While descriptions such as these may relate to welfare-relevant attributes in direct or indirect ways, is unlikely that such characterizations by themselves promote valid expressions of welfare. For example, it is not clear whether, when answering valuation questions of this type, respondents are directly valuing the commodity over which they have preferences. Do respondents care directly about the loss of a specific number of fish, birds or other organisms, or do they instead care about other environmental attributes or outcomes, such as ecosystem condition or biodiversity, and use the number of fish or birds as a proxy for the attributes or outcomes of real concern? In the latter case, respondents might assume ecological production function relationships that do not correspond with those quantified by ecologists. For example, respondents might use presented information on organism numbers or habitat area to make speculative inferences regarding other omitted but welfare relevant ecological attributes.

A related criticism of SP methods, and economic valuation is general, is that lay individuals might not understand or appreciate fully the ecological importance of certain species or processes, and hence estimated economic values will not reflect the true value of these ecological changes (e.g., U.S. EPA 2009). This is most likely to be a concern for nonuse values or use values for which the implications for use are very indirect, such as values stemming from contributions of microorganisms or insects to soil formation, primary productivity, and overall ecological integrity or condition (e.g., Nichols et al. 2008; Weslawski et al. 2004). Such contributions are often captured poorly if at all by the types of ecological variables often included in SP studies, such as changes in water quality, acreage of wetlands or habitat, or even species abundance. Unless such contributions can be expressed in terms that survey respondents can understand and value, and that are consistent with ecological science, welfare estimates will not reflect the full range of potential values derived from ecological systems.

Ambiguous correspondence between SP survey attributes and measurable policy impacts can also render welfare estimates of limited use for policy evaluations. Ecological outcomes of policy implementation are quantified, both in ex ante predictions and ex post evaluations, through effects on measurable outcomes, often called measurement endpoints (U.S. EPA 1998). An emerging literature discusses ways in which these endpoints correspond to well-defined ecosystem services and other valued outputs (Boyd and Banzhaf 2006). SP valuation, in contrast, often derives welfare estimates as a function of attributes that have little systematic relationship to measurable outcomes. As noted above, the use of ecologically ambiguous descriptors such as “unique wildlife habitat,” “high quality groundwater,” or “high biodiversity” is widespread among SP surveys in the literature. A consequence is that associated welfare estimates cannot be integrated into benefit cost analyses without ad hoc and often questionable assumptions required to reconcile ecologically ambiguous SP attributes with measurable policy outcomes.
The theoretical issue

As with all economic values, WTP estimated using SP methods is contingent upon the information available to respondents—either as provided by the survey or derived from other sources (Bergstrom and Stoll 1989; Bergstrom et al. 1989; Cameron and Englin 1997; Hoehn and Randall 2002). In the view of many ecologists and economists, SP survey instruments often fail to provide sufficient information to enable respondents to understand potential effects of ecological change on welfare (Spash and Hanley 1995).

In formal terms, the question regards the content and structure of the utility function. For example, assume that respondent $i$’s utility is of the form $U_i(E(X))$, where $E(\cdot)$ is ecosystem integrity, biodiversity or some other valued ecological outcome and $X$ is the number of migratory fish passing upstream, or some other quantifiable measure. If the respondent cares about $X$ solely because of its impact on $E(\cdot)$, then a valuation question framed solely in terms of $X$ effectively asks the respondent to directly value changes in the input rather than changes in the output that directly influences utility. 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 good or service that results from the input change (Boyd and Banzhaf 2006).

The first implication of this is a potential for bias. A preference function of the form $U_i(E(X))$ can be mathematically collapsed to $U_i(X)$. However, if individuals value $E(\cdot)$ directly, the specification of valuation scenarios solely in terms of $X$ will likely lead to biased WTP estimates, because this requires respondents to assume their own (almost certainly incorrect) ecological production function (i.e., to assume a relationship between $X$ and $E$). As noted by Carson (1998, p. 23), “[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 often have little baseline information (e.g., Carson 1998; Spash and Hanley 1995).

The potential for bias is compounded if utility takes a more complex form such as $U_i(E(X),H(X),X)$, where $H(X)$ represents a second valued ecological outcome influenced by $X$. For example, $H(\cdot)$ might represent expected quality of recreational fishing for non-diadromous species (e.g., largemouth bass), where diadromous fish could have a positive ($\frac{\partial H(\cdot)}{\partial X} > 0$) or negative ($\frac{\partial H(\cdot)}{\partial X} < 0$) marginal effect on recreational fisheries, depending on ecological relationships. Here, $X$ influences utility directly, as well as indirectly through its influence on both $E(X)$ and $H(X)$. Moreover, the indirect effects may not be universally positive. In this context, appropriate modeling of utility is critical to obtaining understanding and unbiased estimates of values for policies that influence $X$, $E(X)$ and $H(X)$.

The situation is further complicated if one cannot observe ecological outcomes directly. Assume, for example, that ecologists have developed multimetric indicators of overall ecosystem condition that are designed to proxy for true underlying ecosystem condition, $E(\cdot)$, that we assume influences individuals’ well being. As a simple illustration, assume that the multimetric indicator $W$ is a formal indicator or measure of $E(\cdot)$ developed by ecologists. However, the
survey designer, concerned that respondents will not be able to understand \( W \) (or unaware of the appropriate ecological literature), instead uses a simplified, \textit{ad hoc} indicator (e.g., high, medium, low), which we denote \( 	ilde{W} \). In this case, respondents must infer \( E(\cdot) \) based on \( 	ilde{W} \), with no provided function or information to make this inference; there is no quantitative relationship between \( 	ilde{W} \) and either \( W \) or \( E(\cdot) \).\textsuperscript{x} Hence, the model estimates WTP for a derived indicator, \( \tilde{W} \), which has no formal ecological interpretation or link to measurable policy outcomes.

The remainder of this paper presents and illustrates the use of a variant of the SP method designed to ameliorate these limitations. In particular, it allows for the incorporation of indicators designed to capture nonuse or indirect use values in a manner that is both grounded in the relevant ecological science and able to distinguish among the various ways in which ecological changes influence respondents’ welfare.\textsuperscript{xi}

\textbf{Structured use of ecological indicators within stated preference valuation}

Outcomes of ecological restoration may be measured and represented using ecological indicators developed by natural scientists. Some of these indicators may be simple and direct, such as the use of laser/optical fish counter data to measure the number of fish passing upstream through a fish ladder. Others are more complex, less direct, and can require additional interpretation. For example, the ecological literature offers numerous integrative, multimetric indicators that formally assimilate multiple ecosystem components and are widely used as indicators of ecosystem health or condition (e.g., Bortone 2005; Davis and Simon 1995; Jorgensen et al. 2004). These indicators (e.g., Index of Biotic Integrity (IBI) and the Estuarine Biotic Integrity Index (EBI)) are calculated from ecological data and provide a quantitative if controversial means to characterize changes in ecosystem condition (Bortone 2005; Karr 1981; 1991).

In contrast to traditional valuation, BSPV employs a more structured use of ecological indicators to characterize and communicate welfare-relevant changes. Specific guidelines ensure that survey scenarios and resulting welfare estimates are characterized by (1) a formal basis in established and measurable ecological indicators, (2) a clear structure linking these indicators to attributes influencing individuals’ well-being, (3) consistent and meaningful interpretation of ecological information, and (4) a consequent ability to link welfare measures to measurable and unambiguous policy outcomes. Specific guidelines to ensure (1) through (4) include:

1) \textit{Indicators used within BSPV survey scenarios must be associated with specific and well-defined ecological changes over which respondents express preferences.} Results of preliminary research such as focus groups and cognitive interviews (Johnston et al. 1995; Kaplowitz et al. 2004; Powe 2007) should be combined with findings from the relevant ecological literature to first determine a comprehensive set of ecological outputs with potential influences on respondents’ utility, then to identify ecological indicators best able to characterize those outputs. This requires differentiation between intermediate ecological changes valued solely as inputs into other final ecological outcomes versus those valued at least in part due to direct influences on utility (Boyd and Banzhaf 2006). Resulting information is then used to provide a one-to-one match between ecological indicators (serving as attributes) in SP scenarios and final welfare-relevant ecological outputs. This guideline requires mapping of relationships between underlying ecological
changes, formal indicators of those changes, and attributes in respondents’ utility functions.

2) **Indicators used within BSPV survey scenarios must have a systematic relationship to ecological field data or quantitative model results.** All indicators within survey scenarios should reflect a measure that is empirically quantifiable, e.g., as the output of a recognized ecological model or standard field measurement. If ordinal categories (e.g., high, medium, low) are used to represent ecological change, the empirical basis for these categories must be defined quantitatively and presented to survey respondents. In addition, there must be justification for the applied categorization within the ecological literature. That is, survey attributes must be traceable to unambiguous ecological measurements.

3) **The quantitative basis for survey attributes must be understandable to respondents and meaningful to scientists.** Survey pretesting must ensure that respondents understand the units and definition of ecological indicators included in survey scenarios, and that these correspond to meaningful outcomes from the perspective of individual utility. Respondents’ understandings’ of indicator units, definitions and interpretations must coincide, at least broadly, with those of natural scientists.

4) **Indicators used within BSPV survey scenarios should be specified such that respondents can identify baselines (i.e., status quo), reference conditions (i.e., the best possible outcome or level of the indicator in an undisturbed system), and changes in both relative and cardinal units where applicable.** Findings from the valuation literature demonstrate that WTP often depends on available (or visible) choice sets, reflecting the full set of policy options possible within a given choice experiment or policy context (Bateman et al. 2004). Similarly, ecological reference conditions may be thought of as characterizing the range of possible substitute goods or policy outcomes that might be available, regardless of whether these are available in presented survey scenarios. Reference conditions also provide a means to interpret quantities such as “200,000 birds” that might be otherwise largely meaningless to lay respondents (e.g., Desvousges et al. 1993; see also comments by Hanemann 1994). The communication of ecological information in this more comprehensive manner provides respondents’ with an increased capacity to comprehend policy scope and to anticipate effects on welfare.

5) **Indicators used within BSPV survey scenarios must be described so that respondents can understand potential linkages between ecological attributes and how these are, or are not, captured in illustrated scenarios.** To prevent respondent speculation regarding ecological production functions and implications for valued ecological outputs, surveys should provide sufficient information regarding relationships between ecological functions, e.g., as inputs and outputs according to ecological production functions. This, together with other BSPV guidelines, can help prevent problems in which, for example, respondents’ choices reflect a speculated (likely incorrect) value for an omitted ecological outcome \( E(X) \) caused by a presented change in \( X \).

6) **Indicators used within BSPV survey scenarios must provide a comprehensive perspective**
on welfare-relevant policy outcomes, including both direct and indirect effects. As noted above, incomplete coverage of welfare-relevant outcomes in survey scenarios can encourage respondents to speculate regarding omitted outcomes. For the case of complex ecological systems, such speculation will almost certainly lead to incorrect assumptions regarding the value of these outcomes. For many restoration policies this guideline requires specification of outcomes (using appropriate ecological indicators) at three levels: (1) the direct effect of proposed policies on targeted species or habitats (e.g., an increase in the number of fish able to travel upstream to spawning areas due to fish run restoration); (2) resulting indirect effects on other specific ecosystem commodities or human uses (e.g., indirect effects of fish passage on other wildlife species); (3) broader indirect effects on ecosystem condition (e.g., as might be measured using a multimetric index of biotic integrity). The inclusion of ecological changes at these three levels also provides a basis for investigating the structure of preferences. For example, it allows a test of whether utility takes the form \( U(E(X), X) \) or simply \( U(E(X)) \). That is, it allows for a distinction between utility gained from specific elements of the ecological system and utility associated with overall system condition, the latter of which may be a source of perhaps substantial nonuse values.

As noted above, there are many examples of SP survey instruments addressing aquatic restoration that incorporate ecological indicators—either simple or complex—in a less structured manner. Some of these surveys may have generated valid welfare estimates. However, the lack of a clear framework specifying linkages among different types of ecological impacts, well-defined ecological indicators and the structure of respondents’ utility can lead to ambiguity regarding the interpretation of model results and potential bias in welfare estimates. BSPV is designed to at least partially ameliorate these concerns and thereby promote more valid estimation of non-market values.

**Random utility model**

The theoretical model for the present application is adapted from a standard random utility specification (Hanemann 1984). We assume that the utility of household \( h \) from ecological restoration program \( k \) \( (U_{hk}) \) is given by

\[
U_{hk}(X_k, W_k(X_k), Y_h, C_{hk}) = v_{hk}(X_k, W_k(X_k), Y_h, C_{hk}) + \varepsilon_{hk}
\]

(1)

where:

- \( X_k \) = vector of indicators characterizing **direct** ecological outcomes of program \( k \);
- \( W_k(\cdot) \) = vector of indicators characterizing **indirect** ecological outcomes of program \( k \) (outcomes related to direct outcomes \( X_k \));
- \( Y_h \) = disposable income of household \( h \);
- \( C_{hk} \) = mandatory cost to the respondent of preservation plan \( k \);
- \( v_{hk}(\cdot) \) = function representing the empirically measurable component of utility;
- \( \varepsilon_{hk} \) = unobservable component of utility, modeled as econometric error.
Given the above specification, household $h$ chooses among three policy plans, ($j=A,B,N$). The household may choose option $A$, option $B$, or may reject both options and choose the status quo (neither plan, $j=N$). A choice of neither plan would result in no restoration and zero household cost, $C_{hk}=0$. The model assumes that household $h$ assesses the utility that would result from choice options ($j=A,B,N$) and chooses that which offers the greatest utility. That is, given (1), household $h$ will choose plan $A$ if

$$U_{ha}(X_A, W_A(X_A), Y_A - C_{ha}) \geq U_{hz}(X_z, W_z(X_z), Y_z - C_{hz}) \quad \text{for } z=B,N,$$

so that

$$v_{ha}(X_A, W_A(X_A), Y_A - C_{ha}) + \epsilon_{ha} \geq v_{hz}(X_z, W_z(X_z), Y_z - C_{hz}) + \epsilon_{hz}.$$  (3)

If the $\epsilon_{hk}$ are assumed independently and identically drawn from a type I extreme value distribution, the model may be estimated as a conditional logit (CL) model (Maddala 1983). The mixed logit (ML) model applied here is a generalization allowing alternative error structures, a relaxation of certain assumptions imposed by the CL model, and expanded possibilities to model response heterogeneity (Greene 2003).

**Empirical application and data**

The SP methods outlined above were developed for a case study addressing public preferences for the restoration of migratory fish passage in the Pawtuxet and Wood-Pawcatuck watersheds of Rhode Island. The present illustration is drawn from a subset of surveys addressing restoration of the Pawtuxet Watershed. The Pawtuxet Watershed currently provides no spawning habitat for migratory fish; access to all 4,347 acres of potential habitat is blocked by 22 dams and other obstructions (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 over dams and access to between 225 and 900 acres of historical, but currently inaccessible, habitat. Choice scenarios and restoration options were informed in part by data and restoration priorities in the *Strategic Plan for the Restoration of Anadromous Fishes to Rhode Island Coastal Streams* (Erkan 2002). Additional information was drawn from the ecological literature on fish passage restoration, interviews with ecologists and policy experts, and other sources described below. Consistent with the strategic plan, the choice experiment addressed restoration methods that neither require dam removal nor would cause appreciable changes in river flows; considered options included fish ladders, bypass channels and fish lifts.

The SP questionnaire was developed and tested over 2½ years through a collaborative process involving interactions of economists and ecologists; meetings with resource managers, natural scientists, and stakeholder groups; and 12 focus groups with 105 total participants. Economist and ecologist researchers were intimately involved in all aspects of survey design, and the ecologist researcher had “veto power” over any aspect of the survey that did not incorporate sufficient ecological clarity. 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 additional insight into respondents’ understanding and interpretation of the questionnaire. Careful attention to development and testing helped ensure that the survey language and format would be easily understood by respondents, that respondents would have similar interpretations of survey terminology and...
scenarios, and that the survey scenarios captured restoration outcomes viewed as relevant and realistic by both respondents and natural scientists. Survey development paid particular attention to the use and interpretation of ecological indicators and related information.

The choice experiment asked respondents to consider alternative options for the restoration of migratory fish passage in the Pawtuxet Watershed. Respondents were provided with two multiattribute restoration options, “Restoration Project A” and “Restoration Project B,” as well as a status quo option that would result in no policy change and zero household cost. Prior to administration of choice questions, the survey provided information (1) describing the current status of Rhode Island river ecology and migratory fish compared to historical baselines, (2) characterizing affected ecological systems and linkages, (3) describing methods and details of fish passage restoration, and (4) providing the definitions, derivations and interpretations of ecological indicators used in survey scenarios, including the reason for their inclusion. All survey language and graphics were pretested carefully to ensure respondent comprehension.

Using ecological indicators to characterize restoration outcomes

As noted above, the primary distinguishing feature of BSPV is the set of guidelines for the use and structure of ecological indicators within survey scenarios. Here, survey choice options are characterized by seven attributes (Table 1). These include five ecological indicators, one attribute characterizing public access, and one attribute characterizing unavoidable household cost. Table 2 illustrates associated attribute levels included in the choice experiment design.

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 numerical results that could be made available through population viability analysis (PVA)xiv;
3) the abundance of fish suitable for recreational harvest (catch), reflecting abundance measures available from ongoing statewide river sampling xv;
4) the abundance of fish-dependent wildlife (wildlife), reflecting the common 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., index of biotic integrity).

Following BSPV guidelines, all indicators have a systematic linkage to ecological field data or quantitative model results. Attribute levels (within the experimental design) are grounded in feasible restoration outcomes identified by ecological models, field studies or 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 informational materials. Scenarios also present the cardinal basis for relative scores where applicable. xvi Relative scores represent percent progress towards the upper reference condition (100%), starting from the lower reference condition (0%). This also implies bounds on
the potential attribute levels that might occur in choice questions, following guidance in the literature to provide visible choice sets (Bateman et al. 2004).

Consider, for example, the aquatic ecological condition score \((IBI)\), described in the survey as “0 - 100 score representing how close a river is to the most natural, undisturbed area found in Rhode Island.” This indicator was included following guidance from preliminary focus groups suggesting that respondents would be willing to pay to restore overall ecological condition in the Pawtuxet Watershed, above and beyond WTP related to all other identifiable restoration outcomes (guideline #1 above). Prior to the illustration of \(IBI\) scores in choice questions, the survey described the calculation and quantitative basis for this multimetric indicator, including characterization of the eight component uni-metric indicators included within the combined \(IBI\) score (guideline #2). Also included were descriptions of ecological linkages through which the restoration of fish passage influences ecological condition (guideline #5). Each choice question specified the status quo level for this indicator (65%) calculated to reflect current watershed conditions, as well as the maximum possible value in the watershed (100%) (guideline #4). Focus groups and pretests were used to ensure shared and accurate interpretation of this information among potential respondents, and that respondents’ interpretations of this indicator were broadly similar to those of policy experts and scientists (guideline #3). Survey pretests and verbal protocols (Schkade and Payne 1994) were also used to ensure correspondence between the presented \(IBI\) and the associated welfare-relevant policy outcome (i.e., the effect of restoration on overall ecological condition), and that this outcome was not confused with other restoration outcomes (guideline #6). Parallel guidelines were applied to all ecological indicators used within choice questions.

**Survey implementation**

From choice attributes (Table 1) and levels (Table 2), a fractional factorial experimental design was created using a D-efficiency criterion for main effects and selected two-way interactions, resulting in 180 profiles optimally blocked into 60 booklets (Lusk and Norwood 2005). Figure 1 illustrates a sample survey scenario and choice question. Each respondent was provided with three choice experiment questions and was instructed to consider each as an independent, non-additive choice. Detailed instructions were also provided, including reminders to consider budget constraints and statements highlighting survey consequentiality (Carson and Groves 2007).

Surveys were implemented using a dual wave phone-mail approach during June, July and August 2008. An initial random digit dial (RDD) 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 postal mail, with repeated follow-up mailings as suggested by Dillman (2000) to increase response rates. A total of 2,400 surveys were mailed to Rhode Island residents; 1,157 surveys were returned (48.2% response rate). Of the total surveys mailed, 600 addressed restoration in the Pawtuxet watershed. The following analysis is based on the 277 usable returns for the Pawtuxet questionnaire, providing 803 completed responses to choice questions.
Empirical model

Choice experiment research has increasingly applied mixed logit (ML) models of choice among discrete alternatives, replacing traditional conditional logit models (Hensher and Greene 2003). ML models allow for coefficients on attributes to be distributed across sampled individuals according to a set of estimated parameters and researcher-imposed restrictions (Hu et al. 2005). While such models require a greater number of choices regarding model specification, they have greater flexibility and can approximate any random utility model (Hensher and Greene 2003).

The random utility model is estimated using maximum likelihood ML with Halton draws in the likelihood simulation. Coefficients on acres, PVA, access, and IBI are specified as random with a normal distribution. The coefficient on annual household cost (cost) is specified as random with a lognormal distribution, ensuring a positive marginal utility of income. Sign-reversal is applied to the cost variable prior to estimation (Hensher and Greene 2003). Coefficients on other variables are specified as fixed. These include the coefficients on the alternative specific constant (neither), catch and wildlife. The final model was chosen after the estimation of preliminary models with varying specifications of fixed and random coefficients. Preliminary models show a high degree of robustness to variations in model specification.

Results

Model results are reported in Table 3. Estimated coefficients are jointly significant at p<0.0001 (-2 log likelihood $\chi^2 = 537.18; \text{df} = 13$), with a pseudo-$R^2$ of 0.31. Seven out of eight model coefficients are statistically significant at p<0.10; six of these are significant at p<0.01. Estimated standard deviations of all random parameter distributions are statistically significant, suggesting that a ML specification is warranted. Results of log-likelihood tests of conditional (suppressed for conciseness) versus mixed logit results yield the same conclusion at p<0.0001 ($\chi^2 = 118.08; \text{df} = 5$). Signs of estimated coefficients match prior expectations.

Even before welfare estimates such as WTP are calculated, the model specification provides a convenient means to interpret and contrast 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 (per percentage point change) for increases in restored acres (acres) and changes in overall ecological condition in the restored area (IBI). 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 and roughly half the size of those associated with acres and IBI. Marginal utility is lowest—and not statistically significant—for changes in the abundance of catchable fish (catch).

These results are mirrored by WTP estimates derived from model results. Because the ML model includes random coefficients, we estimate WTP using the welfare simulation approach of Johnston and Duke (2007; 2009) following the framework of Hensher and Greene (2003). 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.\textsuperscript{xvii} Welfare measures are calculated for each draw, resulting in a combined empirical distribution of $R \times S$ observations from which summary statistics are derived. The resulting empirical distributions accommodate both the sampling variance of parameter estimates and the estimated distribution of random parameters.

Using this procedure, we simulate WTP associated with each restoration policy attribute. While the lognormal distribution of the program cost coefficient in this case ensures a negative impact of cost on utility, and hence is the most commonly chosen distribution for SP payment vehicle parameters, it can also lead to unrealistic mean WTP estimates over simulated distributions, particularly given the long right-hand tail of the distribution (Hensher and Greene 2003). Accordingly, we follow Hu et al. (2005) and Johnston and Duke (2007; 2009) and simulate welfare estimates as the mean over the parameter simulation of median WTP calculated over the coefficient simulation (i.e., mean of median WTP).

Resulting WTP distributions are summarized in Table 4, along with p-values for the null hypothesis of zero (mean of median) WTP. For all attributes except access, 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 reference to the percentiles on the empirical distributions (Poe et al. 2005).

Welfare estimates may be interpreted in either relative (i.e., per percentage point increase in the indicator) or cardinal terms. For example, the reported welfare results in Table 4 imply that mean of median, per household annual WTP for a one percentage point increase in river acres accessible to migratory fish is $0.78. Given that each percentage increase in acres within the Pawtuxet watershed represents 43.47 cardinal acres restored,\textsuperscript{xx} marginal WTP per acre is $0.018, \textit{ceteris paribus}. In contrast, annual WTP for a one percentage point increase in the probability of 50-year fish run survival is $0.29, while that for a one percentage point increase in the aquatic ecological condition index (IBI) is $0.79. Based on simulated empirical welfare distributions (Poe et al. 2005), all WTP estimates except that on catch are statistically significant.

Estimated welfare measures demonstrate that respondents were able to distinguish between anticipated welfare effects of different ecological outcomes, with WTP higher for some outcomes (e.g., increases in ecosystem condition) than for others (e.g., increases in catchable fish abundance). Moreover, results suggest substantial WTP both for restoration outcomes that are straightforward and direct (e.g., the number of acres accessible to migratory fish), as well as more complex and indirect (e.g., effects on ecological condition). Moreover, while public access (access) to restored areas is valued highly, the relationship of ecological outcomes to human uses is not sufficient to guarantee non-zero WTP. Changes in catchable fish abundance (catch), for example, are not associated with statistically significant WTP. Taken together, results suggest that respondents are able to make systematic choices over complex restoration possibilities, and

WP 2009-06

13
Implications and discussion

Results presented in this paper suggest that ad hoc, oversimplified representation of ecological outcomes is not a compulsory aspect of survey-based welfare estimation; SP choice experiment scenarios can use indicators that are meaningful from both ecological and economic perspectives. More complete, structured specifications of ecological outcomes within SP questionnaires—while demanding greater attention to survey design and testing—can provide welfare results that are both more comprehensive from a utility perspective and more easily linked to measurable policy outcomes.

Results for the case study discussed here, that of migratory fish passage restoration in Rhode Island watersheds, provide insight into ways in which respondents value outcomes of aquatic ecosystem restoration. Survey responses suggest nuanced preferences for aquatic ecosystem change. Welfare estimates reveal differences in WTP across restoration attributes. Interestingly, some of the more direct effects of fish passage restoration, such as increases in the probability of fish run survival (PVA), are associated with modest WTP compared to some indirect restoration effects. This suggests that past surveys in which policy effects have been characterized solely in terms of the direct effects on individual species might have omitted some of the primary outcomes through which restoration influences utility.

For example, SP survey scenarios often omit effects on overall ecosystem condition,xxi as measured here using an index of biotic integrity. In the present study, this indicator has a larger associated marginal welfare estimate than any other ecological attribute (Table 4). These results suggest that respondents can gain significant utility from changes in overall ecosystem condition, even after one holds constant effects on other primary restoration outcomes. Welfare evaluations should therefore consider the possibility that ecosystem condition can yield significant non-market values that are distinct from the values of ecological attributes or outcomes that might influence that condition (such as species abundance). Model results also imply that some past SP surveys in the literature may have suppressed exactly the type of information that is most relevant to respondents when determining WTP for ecological restoration.

Model results also raise the prospect—following similar findings of Bulte et al. (2005) and Johnston and Duke (2007)—that respondents may have well-defined WTP for an attribute reflecting the status of an ecological process (the condition of a functioning ecological system), apart from the outcomes of that process. Some economists might argue that the condition of an ecosystem represents the status of an ecological process that should only be valued in as much as it contributes to other valued outputs. Others, however, might interpret ecological condition as an output that can be valued apart from other ecosystem services or outputs. While results here cannot resolve this debate, they do indicate clearly that respondents are willing to pay for improvements in ecological condition, holding other outputs constant. From an empirical perspective, respondents view ecological condition as directly welfare relevant.

Results also suggest the potential hazards in less comprehensive coverage of ecological outcomes in SP scenarios. For example, contingent valuation and choice experiment surveys often emphasize direct or indirect effects on recreational fishing, sometimes with relatively sparse descriptions of other ecological outcomes. Associated models frequently estimate
statistically significant WTP for improvements in recreational fishing. In contrast, results here suggest that if one accounts for other ecological impacts, respondents may not always express a significant WTP for recreational fishery improvements (i.e., the coefficient on catch is not statistically significant). This raises the possibility that the incorporation of recreational fishery effects in SP scenarios that omit other related ecosystem effects might cause respondents to use fishery impacts as a proxy for other welfare-relevant ecological outcomes.

At a minimum, results suggest that WTP for aquatic ecosystem restoration can reflect a wide range of direct and indirect effects. 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. Avoidance of speculation requires careful survey and model development to ensure a comprehensive perspective on welfare-relevant outcomes.

Conclusion

SP valuation methods may in some cases provide insufficient ecological information to enable meaningful, unbiased WTP estimation. While there are exceptions (e.g., Horne et al. 2005), the large number of valuation surveys implemented with ad hoc treatments of ecological outcomes suggests the need for more structured guidance on the use of ecological information within non-market valuation. The research presented above illustrates a method that addresses the need for a more systematic treatment of ecological indicators within SP valuation and related structural linkages between ecological and economic frameworks. Despite prior efforts to merge ecological and economic information for economic valuation, SP valuation still commonly retains often ad hoc treatments of ecological information, with little reference to the extensive work by ecologists to model and measure ecosystem condition and function.

The broader purpose of the research presented in this paper is to better ground SP welfare evaluation in the types of ecological indicators and models developed within the ecological sciences, as well as in measurable data available to policymakers. This paper formalizes an approach to SP valuation characterized by a set of guidelines and criteria for use of ecological indicators within models and surveys. These guidelines promote a more defensible structure for ecological valuation and a more comprehensive representation of ecological change within SP scenarios, yielding more valid welfare estimation.

We emphasize that this paper addresses only a few of the many challenges involved in the coordination of economics and ecology for welfare estimation. Reported findings are limited by the policy case study from which they are drawn, and additional verification in other valuation contexts will be required to assess the broader applicability of the proposed methods. These limitations aside, results presented here suggest the potential benefits of closer and more meaningful collaboration among natural and social scientists. Potential benefits for economists include enhanced validity and relevance of welfare estimates, increased ability to link non-market values to measurable policy outcomes, and improved validity of results.
References


Johnston, R.J. and J.M. Duke. 2009. Willingness to Pay for Land Preservation Across States and


### Table 1. Choice Experiment Variables and Descriptive Statistics

| Variable | Definition | Mean (Std. Dev.)
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>acres</td>
<td>The number of acres of river habitat accessible to migratory fish, presented as a percentage of the reference value for the watershed (Erkan 2002). Range 0-100.</td>
<td>8.1794 (8.1550)</td>
</tr>
<tr>
<td>PVA</td>
<td>Population viability analysis (PVA) score: Estimated probability, in percentage terms, that migratory species will migrate the river in 50 years. Reference condition is estimated based from surveys of experts in fish restoration. Range 0-100.</td>
<td>33.4413 (28.1265)</td>
</tr>
<tr>
<td>access</td>
<td>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.</td>
<td>0.3296 (0.4702)</td>
</tr>
<tr>
<td>IBI</td>
<td>Index of biotic integrity (IBI) score: A linear multimetric index of aquatic ecological condition, 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 species, number of sensitive species, number of feeding types, percentage of native individuals, percentage of migratory individuals, and percentage of fish that are tumor free. Presented as a percentage of the reference condition. Range 0-100.</td>
<td>71.6978 (6.0762)</td>
</tr>
<tr>
<td>cost</td>
<td>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.</td>
<td>11.9762 (14.1019)</td>
</tr>
<tr>
<td>neither</td>
<td>Alternative specific constant (ASC) associated with the status quo, or a choice of neither plan.</td>
<td>0.3333 (0.4715)</td>
</tr>
<tr>
<td>catch</td>
<td>The number of catchable-size fish in restored areas, measured as 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.</td>
<td>79.9087 (7.5807)</td>
</tr>
<tr>
<td>wildlife</td>
<td>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.</td>
<td>65.0125 (10.3920)</td>
</tr>
</tbody>
</table>

a Means and standard deviations include status quo option of no restoration.
Table 2. Attribute Levels in Choice Experiment Design

<table>
<thead>
<tr>
<th>Variable</th>
<th>Levels</th>
</tr>
</thead>
</table>
| acres    | 1. 0% (0 acres accessible to fish)
|          | 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)
|          | 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) |
| access   | 1. Public Cannot Walk and Fish in Area
|          | 2. Public Can Walk and Fish in Area |
| IBI      | 1. 65% (aquatic ecological condition score)
|          | 2. 70% (aquatic ecological condition score)
|          | 3. 75% (aquatic ecological condition score)
|          | 4. 80% (aquatic ecological condition score) |
| cost     | 1. $0 (cost to household per year)
|          | 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) |
| catch    | 1. 70% (102 fish/hour sampling abundance)
|          | 2. 80% (116 fish/hour sampling abundance)
|          | 3. 90% (130 fish/hour sampling abundance) |
| wildlife | 1. 55% (20 species common)
|          | 2. 60% (22 species common)
|          | 3. 70% (25 species common)
|          | 4. 80% (28 species common) |

\(^a\) Status quo value.
Table 3. Mixed Logit Results: Pawtuxet Restoration Choice Experiment

| Variable            | Coefficient | Std. Error | Pr > |t| |
|---------------------|-------------|------------|------|---|
| **Random Parameters** |             |            |      |   |
| acres               | 0.0520      | 0.0150     | 0.0005 |
| PVA                 | 0.0196      | 0.0067     | 0.0035 |
| access              | 1.2435      | 0.2354     | 0.0001 |
| IBI                 | 0.0525      | 0.0219     | 0.0163 |
| cost (lognormal)    | -2.9738     | 0.1724     | 0.0001 |
| **Fixed Parameters** |             |            |      |   |
| neither             | -1.7598     | 0.5038     | 0.0005 |
| catch               | 0.0027      | 0.0105     | 0.7957 |
| wildlife            | 0.0275      | 0.0099     | 0.0057 |
| **Standard Deviations of Random Parameter Distributions** | | | |
| std_acres           | 0.1089      | 0.0257     | 0.0001 |
| std_PVA             | 0.0452      | 0.0083     | 0.0001 |
| std_access          | 1.3564      | 0.4572     | 0.0030 |
| std_IBI             | 0.1531      | 0.0380     | 0.0001 |
| std_cost (lognormal)| 0.8351      | 0.2347     | 0.0004 |
| -2 Log Likelihood $\chi^2$ | 537.18 |  | 0.0001 |
| Pseudo-$R^2$        | 0.31        | | |
| Observations ($N$)  | 803         | | |
Table 4. Marginal Willingness to Pay Estimates (Implicit Prices): Empirical Distributions\textsuperscript{a}

| Variable | WTP   | Standard Deviation | Skewness | Kurtosis | Percentiles (1%, 99%) | Pr > |t|\textsuperscript{b} |
|----------|-------|--------------------|----------|----------|------------------------|------|----------------------|
| acres    | 0.7841| 0.2658             | 0.37     | 3.17     | (0.23, 1.44)           | <0.01|
| PVA      | 0.2881| 0.1143             | 0.39     | 3.34     | (0.05, 0.58)           | <0.01|
| access   | 19.9620| 4.5110            | 0.53     | 3.59     | (10.71, 33.46)         | <0.01|
| IBI      | 0.7916| 0.3749             | 0.28     | 3.52     | (-0.03, 1.70)          | 0.02 |
| catch    | 0.0497| 0.2109             | -0.32    | 3.61     | (-0.52, 0.53)          | 0.76 |
| wildlife | 0.5489| 0.2024             | 0.20     | 3.12     | (0.12, 1.04)           | <0.01|

\textsuperscript{a} Results reflect the mean over the parameter simulation of median 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.

\textsuperscript{b} P-values are two-tailed, for the null hypothesis of zero mean of median WTP, calculated directly from percentiles in simulated empirical WTP distributions (see text).
**Question 5.** Projects A and B are possible restoration projects for the Pawtuxet River, and the **Current Situation** is the status quo with no restoration. Given a choice between the three, how would you vote?

<table>
<thead>
<tr>
<th>Effect of Restoration</th>
<th>Current Situation (no restoration)</th>
<th>Restoration Project A</th>
<th>Restoration Project B</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fish Habitat</td>
<td>0% 0 of 4347 river acres accessible to fish</td>
<td>5% 225 of 4347 river acres accessible to fish</td>
<td>20% 900 of 4347 river acres accessible to fish</td>
</tr>
<tr>
<td>Population Survival Score</td>
<td>0% Chance of 50-year survival</td>
<td>30% Chance of 50-year survival</td>
<td>30% Chance of 50-year survival</td>
</tr>
<tr>
<td>Catchable Fish Abundance</td>
<td>80% 116 fish/hour found out of 145 possible</td>
<td>70% 102 fish/hour found out of 145 possible</td>
<td>70% 102 fish/hour found out of 145 possible</td>
</tr>
<tr>
<td>Fish-Dependent Wildlife</td>
<td>55% 20 of 36 species native to RI are common</td>
<td>80% 28 of 36 species native to RI are common</td>
<td>60% 22 of 36 species native to RI are common</td>
</tr>
<tr>
<td>Aquatic Ecological Condition Score</td>
<td>65% Natural condition out of 100% maximum</td>
<td>70% Natural condition out of 100% maximum</td>
<td>80% Natural condition out of 100% maximum</td>
</tr>
<tr>
<td>Public Access</td>
<td>Public CANNOT walk and fish in area</td>
<td>Public CANNOT walk and fish in area</td>
<td>Public CAN walk and fish in area</td>
</tr>
</tbody>
</table>

| $0 | $15 | $25 |
| Cost to your Household per Year | Increase in Annual Taxes and Fees | Increase in Annual Taxes and Fees |

**HOW WOULD YOU VOTE? (CHOOSE ONE ONLY)**

- [ ] I vote for **NO RESTORATION**
- [ ] I vote for **PROJECT A**
- [ ] I vote for **PROJECT B**
**Figure 1. Sample Choice Experiment Question**

**Endnotes**

i Similar limitations are found in other areas of economic analysis (Simpson 1998).

ii Affected species include river herring (blueback herring and alewife), shad and American eel.

iii For example, respondents might select among different options for coastal wetland restoration, each with distinct effects on wetland attributes, related ecosystem services and household costs (Johnston et al. 2002).

iv For an exception, see Holmes et al. (2004).

v For example, Population Viability Analysis (PVA) is an approach that yields benchmarks on the status of individual species in the system. PVA is a process wherein demographic features (abundance, population structure), genetic characteristics, and environmental variability are modeled to yield predictions of the probability that a population will persist for a specified period of time under different scenarios, bounded by 0 and 1 (Boyce 1992). Indices of Biotic Integrity provide comparable benchmarks on a system-wide level representing ecological condition; they are conventionally scaled relative to least-impacted, relatively pristine reference sites (Jackson et al. 2000).

vi Many of these fall under the category of “supporting services” in the typology of ecosystem services used in the Millennium Ecosystem Assessment (2005).

vii An alternative approach would be to educate the survey respondents about the relevant ecological science. However, the ability to do this within the confines of a survey is limited.

viii For example, in the absence of other defining information, WTP for attainment of “high” biodiversity cannot be linked to measurable policy outcomes without assumptions regarding the combinations of observable conditions, or measurement endpoints, that one might associate with this otherwise ambiguous descriptor.

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

x This lack of formalized relationship may be realized at either the researcher level (i.e., researchers have not specified any formal relationship) or at the respondent level (i.e., researchers may have specified a relationship, but this is unknown or poorly understood by respondents.

xi While not explored here, the proposed method also has the potential to expand possibilities for benefit transfer. Consider, for example, a utility function of the form $U(E(X,Y))$, where $E$ is an ecological outcome determined by two (or more) relevant variables $X$ and $Y$. Suppose that in a given context a change in $X$ leads to a change in $E$, and this change in $E$ is valued using the illustrated methods. Provided the affected population is sufficiently similar, this value could then be transferred and used in a context where $E$ changes as a result of a change in $Y$ rather than a change in $X$. This would be valid provided that individuals care only about the change in $E$ and not independently about the source of any change in $E$, as posited in a utility function of this form. Whether utility, in fact, takes this form is an empirical question, which the presents methods can help to answer.
xii This total includes one focus group conducted with respondents from an undergraduate economics course (11 participants); all other groups included respondents recruited from the general public.

xiii No formal count was made of individual interviews used in survey design, but we approximate that a minimum of 60 in-person and telephone interviews were conducted with expert and non-expert respondents between September 2005 and June 2008.

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

xv This reflects the effect of fish passage on the abundance of other recreationally harvested fish such as largemouth bass (Yako et al. 2000).

xvi For example, river acres available to fish migrations (acres) are presented both as a cardinal number of acres and as a percentage relative to historical habitat available in the watershed.

xvii Other versions of the survey included variations in the definition or set of included ecological indicators or addressed restoration in other Rhode Island watersheds.

xviii This excludes access, which is measured as a dummy variable and is hence not comparable in this manner.

xix Because the coefficient on cost is specified with a lognormal distribution, an exponential transformation is required subsequent to the coefficient simulation to obtain a simulated empirical density for the coefficient itself (Hensher and Greene 2003).

xx There are 4,347 total acres of potential river habitat in the watershed, of which zero are currently accessible to migratory fish.

xvi For an exception, see Jakus and Shaw (2003).