Valuation for Environmental Policy: Ecological Benefits

A Workshop sponsored by U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Crowne Plaza Washington National Airport Hotel
1480 Crystal Drive
Arlington, VA 22202

April 23-24, 2007

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U.S. Environmental Protection Agency (EPA)
National Center for Environmental Economics (NCEE) and
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Agenda

April 23, 2007: Valuation for Environmental Policy

8:00 a.m. – 8:30 a.m.	**Registration**	
8:30 a.m. – 8:45 a.m.	**Introductory Remarks** Rick Linthurst, National Program Director for Ecology, EPA, Office of Research and Development	
8:45 a.m. – 11:30 a.m.	**Session I: Benefits Transfer** Session Moderator: Steve Newbold, EPA, NCEE	
	8:45 a.m. – 9:15 a.m.	Benefits Transfer of a Third Kind: An Examination of Structural Benefits Transfer George Van Houtven, Subhrendu Pattanayak, Sumeet Patil, and Brooks Depro, Research Triangle Institute
	9:15 a.m. – 9:45 a.m.	The Stability of Values for Ecosystem Services: Tools for Evaluating the Potential for Benefits Transfers John Hoehn, Michael Kaplowitz, and Frank Lupi, Michigan State University
9:45 a.m. – 10:00 a.m.	**Break**	
	10:00 a.m. – 10:30 a.m.	Meta-Regression and Benefit Transfer: Data Space, Model Space and the Quest for 'Optimal Scope' Klaus Moeltner, University of Nevada, Reno, and Randall Rosenberger, Oregon State University
	10:30 a.m. – 10:45 a.m.	Discussant: Matt Massey, EPA, NCEE
	10:45 a.m. – 11:00 a.m.	Discussant: Kevin Boyle, Virginia Tech University
	11:00 a.m. – 11:30 a.m.	Questions and Discussion
11:30 a.m. – 12:45 p.m.	**Lunch**	

April 23, 2007 (continued)

12:45 p.m. – 3:30 p.m.	**Session II: Wetlands and Coastal Resources** Session Moderator: Cynthia Morgan, EPA, NCEE	
	12:45 p.m. – 1:15 p.m.	A Combined Conjoint-Travel Cost Demand Model for Measuring the Impact of Erosion and Erosion Control Programs on Beach Recreation Ju-Chin Huang, University of New Hampshire; George Parsons, University of Delaware; Min Qiang Zhao, The Ohio State University; and P. Joan Poor, St. Mary's College of Maryland
	1:15 p.m. – 1:45 p.m.	A Consistent Framework for Valuation of Wetland Ecosystem Services Using Discrete Choice Methods David Scrogin, Walter Milon, and John Weishampel, University of Central Florida
1:45 p.m. – 2:00 p.m.	**Break**	
	2:00 p.m. – 2:30 p.m.	Linking Recreation Demand and Willingness To Pay With the Inclusive Value: Valuation of Saginaw Bay Coastal Marsh John Whitehead and Pete Groothuis, Appalachian State University
	2:30 p.m. – 2:45 p.m.	Discussant: Jamal Kadri, EPA, Office of Wetlands, Oceans, and Watersheds
	2:45 p.m. – 3:00 p.m.	Discussant: John Horowitz, University of Maryland
	3:00 p.m. – 3:30 p.m.	Questions and Discussion
3:30 p.m. – 3:45 p.m.	**Break**	
3:45 p.m. – 5:45 p.m.	**Session III: Invasive Species** Session Moderator: Maggie Miller, EPA, NCEE	
	3:45 p.m. – 4:15 p.m.	Models of Spatial and Intertemporal Invasive Species Management Brooks Kaiser, Gettysburg College, and Kimberly Burnett, University of Hawaii at Manoa
	4:15 p.m. – 4:45 p.m.	Policies for the Game of Global Marine Invasive Species Pollution Linda Fernandez, University of California at Riverside
	4:45 p.m. – 5:00 p.m.	Discussant: Marilyn Katz, EPA, Office of Wetlands, Oceans, and Watersheds
	5:00 p.m. – 5:15 p.m.	Discussant: Lars Olsen, University of Maryland
	5:15 p.m. – 5:45 p.m.	Questions and Discussion
5:45 p.m.	**Adjournment**	

April 24, 2007: Valuation for Environmental Policy

8:30 a.m. – 9:00 a.m.	**Registration**	
9:00 a.m. – 11:45 a.m.	**Session IV: Valuation of Ecological Effects** Session Moderator: William Wheeler, EPA, NCER	
	9:00 a.m. – 9:30 a.m.	Integrated Modeling and Ecological Valuation: Applications in the Semi Arid Southwest David Brookshire, University of New Mexico, Arriana Brand, Jennifer Thacher, Mark Dixon,Julie Stromberg, Kevin Lansey, David Goodrich, Molly McIntosh, Jake Gradny, Steve Stewart, Craig Broadbent and German Izon
	9:30 a.m. – 10:00 a.m.	Contingent Valuation Surveys to Monetize the Benefits of Risk Reductions Across Ecological and Developmental Endpoints Katherine von Stackelberg and James Hammitt, Harvard School of Public Health
10:00 a.m. – 10:15 a.m.	**Break**	
	10:15 a.m. – 10:45 a.m.	Valuing the Ecological Effects of Acidification: Mapping the Extent of Market and Extent of Resource in the Southern Appalachians Shalini Vajjhala, Anne Mische John, and David Evans, Resources for the Future
	10:45 a.m. – 11:00 a.m.	Discussant: Joel Corona, EPA, Office of Water
	11:00 a.m. – 11:15 a.m.	Discussant: David Simpson, Johns Hopkins University
	11:15 a.m. – 11:45 a.m.	Questions and Discussion
11:45 a.m. – 1:00 p.m.	**Lunch**	
1:00 p.m. – 4:15 p.m.	**Session V: Water Resources** Session Moderator: Adam Daigneault, EPA, NCEE	
	1:00 p.m. – 1:30 p.m.	Valuing Water Quality as a Function of Physical Measures Kevin Egan, Joe Herriges, John Downing, and Katherine Cling, Iowa State University
	1:30 p.m. – 2:00 p.m.	Cost-Effective Provision of Ecosystem Services from Riparian Buffer Zones Jo Albers, Oregon State University; David Simpson, Johns Hopkins University; and Steve Newbold, NCEE
2:00 p.m. – 2:15 p.m.	**Break**	
	2:15 p.m. – 2:45 p.m.	Development of Bioindicator-Based Stated Preference Valuation for Aquatic Resources Robert Johnston, Eric Shultz, Kathleen Segerson, Jessica Kukielka, Deepak Joglekar, University of Connecticut; and Elena Y. Besedin, Abt Associates

April 24, 2007 (continued)

	2:45 p.m. – 3:05 p.m.	Comparing Management Options and Valuing Environmental Improvements in a Recreational Fishery Steve Newbold and Matt Massey, NCEE
	3:05 p.m. – 3:20 p.m.	Discussant: Julie Hewitt, EPA, Office of Water
	3:20 p.m. – 3:35 p.m.	Discussant: George Parsons, University of Delaware
	3:35 p.m. – 4:05 p.m.	Questions and Discussions
4:05 p.m. – 4:15 p.m.	**Final Remarks**	
4:15 p.m.	**Adjournment**	

Valuing Water Quality as a Function of Physical Measures[1]

Kevin J. Egan, Joseph A. Herriges, and Catherine L. Kling

Department of Economics, Iowa State University

John A. Downing

Department of Ecology, Evolution and Organismal Biology, Iowa State University

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Abstract

Employing a unique and rich data set of physical water quality attributes in conjunction with detailed household characteristics and trip information, we develop a mixed logit model of recreational lake usage. Our empirical analysis shows that individuals are responsive to the full set of physical water quality measures used by biologists to identify the impaired status of lakes. WTP estimates are reported based on improvements in these physical measures. This implies that cost benefit analysis based on physical water quality measures can be used as a direct policy tool.

1 Introduction

More than three decades have elapsed since the passage of the 1972 Clean Water Act (CWA), yet progress toward meeting the standards set forth in the CWA has been slow in the area of nonpoint source pollution. The most recent National Water Quality Inventory (USEPA,[21]) categorizes forty-five percent of assessed lake acres in the U.S. as impaired, with the leading causes of these impairments being nutrients and siltation. Moreover, few states have developed the priority ranking of their impaired waters or determined the Total Maximum Daily Loads (TMDLs) as required under Section 303(d) of the CWA.[1] Legal actions by citizen groups have prompted renewed efforts toward developing both the priority listing and associated TMDL standards.[2] However, the task facing both the EPA and state regulatory agencies remains a daunting one. The prioritization process alone, which is all the more important given current tight budgets, requires information on the cost of remediation and the potential benefits that will flow from water quality improvements. Both types of information are in short supply. The purpose of this paper is to help fill this gap by providing information on the recreational value of water quality improvements as a function of detailed physical attributes of the water bodies involved. The water quality values are obtained from a recreation demand model of lake usage in the state of Iowa, combining trip and socio-demographic data from the Iowa Lakes Valuation Project and an extensive list of physical water quality measures collected by Iowa State University's Limnology Laboratory.

Recreation demand models have long been used to value water quality improvements, but studies typically rely on limited measures of water quality. The most commonly used indicators are fish catch rates (e.g., [4], [15]). However, catch rates are themselves endogenous,

[1] TMDLs specify the amount of a pollutant that a water body can receive and still meet existing water quality standards.

[2] As of March 2003, there have been approximately 40 legal actions taken against the USEPA in 38 states concerning the implementation of Section 303(d) of the CWA.

depending on both fishing pressure and the abilities of the anglers, and provide only indirect measures of the underlying water quality. Physical water quality measures, such as Secchi depth and bacteria counts, are used only sparingly, in large part because of limitations in available data. Phaneuf, Kling, and Herriges [18] use fish toxin levels in their model of Great Lakes fishing, but the toxin levels were available only for a limited number of aggregate sites in the region. Parsons and Kealy [17] use dummy variables based on dissolved oxygen levels and average Secchi depth readings to capture the impact of water quality on Wisconsin lake recreation. Similarly, Parsons, Helm, and Bondelid [16] construct dummy variables indicating *high* and *medium* water quality levels for use in their analysis of recreational demand in six northeastern states. These dummy variables are based on pollution loading data and water quality models, rather than on direct measurements of the local water quality. In all of these studies, the physical water quality indicators are found to significantly impact recreation demand, but, because of the limited nature of the measures themselves, provide only a partial picture of value associated with possible water quality improvements.

Bockstael, Hanemann, and Strand's [3] analysis of beach usage in the Boston-Cape Cod area has perhaps one of the most extensive lists of objective physical water quality attributes included in a model of recreation: oil, fecal coliform, temperature, chemical oxygen demand (COD), and turbidity. However, the study also points out one of the frequently encountered problems in isolating the impact of individual water quality attributes - multicollinearity. Seven additional water quality measures were available to the analysts: color, pH, alkalinity, phosphorus, nitrogen, ammonia, and total coliform. These latter variables were excluded from the analysis because of correlations among the various groups of water quality measures. The five water quality variables used were chosen because they were either directly observable by recreationists or highly publicized. While these choices are certainly reasonable given limitations in the available data, the lack of direct information on how nutrient

levels (phosphorus and nitrogen) impact recreational usage is unfortunate in the context of setting standards in Midwestern states, where nutrient loadings are of particular concern.

The contribution of the current paper lies in our ability to incorporate a rich set of physical water quality attributes, as well as site and household characteristics, into a model of recreational lake usage. Importantly, the full set of water quality variables used by biologists to classify lakes as impaired by the EPA, and therefore potentially in need of policy action, are included. Trip data for the study are drawn from the 2002 Iowa Lakes Survey, the first in a four-year project aimed at valuing recreational lake usage in Iowa. The survey was sent to a random sample of 8,000 Iowa households, eliciting information on their recreational visits to Iowa's 129 principal lakes, along with socio-demographic data and attitudes toward water quality issues. The unique feature of the project, however, is that a parallel inventory of the physical attributes of these lakes is being conducted by Iowa State University's Limnology Laboratory.[3] Three times a year, over the course of a five-year project, thirteen distinct water quality measurements are being taken at each of the lakes, providing a clear physical characterization of the conditions in each lake. Moreover, because of the wide range of lake conditions in the state, Iowa is particularly well suited to identifying the impact of these physical characteristics on recreation demand. Iowa's lakes vary from a few clean lakes with up to fifteen feet of visibility to other lakes having some of the highest concentrations of nutrients in the world, and roughly half of the 129 lakes included in the study are on the EPA's list of impaired lakes.

A second unique contribution of this study is the application of careful model specification and fitting procedures to identify the best set of explanatory variables, and their functional form, for the estimated model. Since economic theory does not provide guidance to the analyst on these issues, ex ante selection of model variables and structure will often fail to

[3]The limnological study is funded by the Iowa Department of Natural Resources.

achieve the best model fit. On the other hand, specification searching on a given data set leads to the well known problem of incorrect standard error estimates (Leamer, [13]). Thus, we exploit our large sample size by splitting the sample into three equal parts. With the first sample, we extensively explore various specifications of the model, including a variety of variables and their functional form. The second sample was reserved for clean model estimation, allowing us to generate unbiased estimates of precision for all of the parameter estimates. The third split of the sample was used to perform out-of-sample prediction to provide an overall assessment of the model fit.[4]

The remainder of the paper is divided into five sections. Section 2 provides an overview of the two data sources. A repeated mixed logit model of recreational lake usage is then specified in Section 3. The mixed logit model allows for a wide variety of substitution patterns among the recreational sites and for heterogeneity among households in terms of their reaction to individual site characteristics. (See, e.g., [9],[14], and [20].) The specification search procedure and parameter estimates are reported in Section 4.[5] In Section 5, we illustrate not only the implications of the model in terms of recreational value of meeting the objectives of the CWA (i.e., removing all of the lakes in the state from the impaired water quality list) but also how the model can be used to prioritize the remediation task. Conclusions of the paper are provided in Section 6.

2 Data

Two principal data sources are used in developing our model of recreational lake usage in Iowa: the 2002 Iowa Lakes Survey and the physical water quality measures collected by Iowa State University's Limnology Laboratory. As noted earlier, the 2002 Iowa Lakes Survey is

[4] We are aware of only one other recreation demand study that has adopted this procedure. Creel and Loomis ([5], [6]) use this procedure to identify the key explanatory variables for deer hunting in California.

[5] The current paper highlights the results of the specification and estimation stages. A more detailed analysis, including out-of-sample predictions findings, is provided in a companion paper [8].

the first survey in a four-year study of lake usage in the state. The focus of the survey was on gathering baseline information on the visitation patterns to Iowa's 129 principal lakes, as well as socio-demographic data and attitudes towards water quality issues. After initial focus groups and pre-testing of the survey instrument, the final survey was administered by mail in November 2002 to 8,000 randomly selected households in the state. Standard Dillman procedures ([7]) were used to ensure a high response rate.[6] Of the 8,000 surveys mailed, 4,423 were returned. Allowing for the 882 undeliverable surveys, this corresponds to an overall response rate of sixty-two percent.

The survey sample was initially paired down to 3,859 households as follows. Those individuals who returned the survey from out of state were excluded (thirty-eight observations). It is not feasible to ascertain whether these respondents have permanently left the state or simply reside elsewhere for part of the year. Respondents who did not complete the trip questions or did not specify their numbers of trips (i.e., they simply checked that they had visited a given lake) were excluded (224 observations). Lastly, anyone reporting more than fifty-two total single-day trips to the 129 lakes were excluded (133 observations). In the analysis that follows, only single-day trips are included to avoid the complexity of modeling multiple-day visits. Defining the number of choice occasions as fifty-two allows for one trip per week to one of the 129 Iowa lakes. While the choice of fifty-two is arbitrary, it seems a reasonable cut-off for the total number of allowable single-day trips for the season.[7] This last step eliminated approximately three percent of the returned surveys. Finally, because of the large number of respondents, the overall sample was randomly divided into three segments; specification, estimation, and prediction portions, each component using just under 1,290 observations. Once the estimation stage is reached, the results will be free from any form

[6] Complete details of the survey design and implementation can be found in [2].

[7] Sensitivity analysis, raising the allowable number of trips per year above fifty-two, indicated that the results were not sensitive to the choice of this cut-off.

of pretest bias and the standard errors will be not be biased by the extensive specification search.[8]

Table 1 provides summary statistics for the full sample regarding trip and the socio-demographic data obtained from the survey. The average number of total single-day trips for all 129 lakes is 6.68, varying from some respondents taking zero trips and others taking fifty-two trips. In general, the survey respondents are more likely to be older, male, have a higher income, and to be more educated than the general population. Schooling is entered as a dummy variable equaling one if the individual has attended or completed some level of post-high school education.

The physical water quality measures used in modeling recreational lake usage in Iowa were gathered by Iowa State University's Limnology Laboratory as part of the ongoing state lake monitoring program. Table 2 provides a listing of the water quality attributes and 2002 summary statistics for the 129 lakes used in our analysis. All of the physical water quality measures are the average values for the 2002 season. Samples were taken from each lake three times throughout the year, in spring/early summer, mid-summer, and late summer/fall to cover the range of seasonal variation.

Each of the water quality measures help to characterize a distinct aspect of the lake ecosystem. Secchi transparency is one of the most widely applied limnological parameters and approximately reflects the lake depth at which the bottom of the lake can still be seen. Chlorophyll is an indicator of phytoplankton plant biomass which leads to greenness in the water. Three nitrogen levels were gathered. Total nitrogen is the sum of all dissolved and particulate forms. NH3+NH4 measures ammonium nitrogen that derive from fertilizer or anaerobic conditions and can be toxic. NO3+NO2 measures the nitrate level in the water that derives from aerobic nutrient contributions. Total phosphorus is an important indicator

[8] Creel and Loomis [5] use a similar procedure in investigating alternative truncated count data estimators.

of nutrient conditions in freshwater systems and is usually the principal limiting nutrient which determines phytoplankton (algae?) growth. Silicon is important to diatoms, a key food source for aquatic organisms. The hydrogen ion concentration of the water is measured by pH with levels below 6 indicating acid conditions and levels above 8 indicating extreme basicity. As Table 2 notes, all of the pH levels in this sample are clustered between 7.3 and 10. Alkalinity is a reflection of the buffering capacity of the water and is expressed as the concentration of calcium carbonate in the water. Plants need carbon to grow and most carbon comes from alkalinity in lakes; therefore, alkalinity is an indication of the availability of carbon to plant life. Inorganic suspended solids (ISS) consist of soil and silt suspended in the water through erosion, whereas volatile suspended solids (VSS) consists of suspended organic matter. Increases in either ISS or VSS levels decrease water clarity.

Table 2 demonstrates that there is considerable variation in water quality conditions throughout the state. For example, average Secchi depth varies from a low of 0.09 meters (or 3.5 inches) to a high of 5.67 meters (over 18 feet). Total phosphorus varies from 17 to 453 μg/L, spanning the range of concentrations seen in the world (Arbuchle and Downing, [1]). An additional unique aspect of Iowa lakes is that the diversity of land uses in the watershed contributing to them leads to a low degree of collinearity among the water quality measures.

In addition to trip and water quality data, two other data sources were used. First, the travel costs, from each survey respondent's residence to each of the 129 lakes, were needed. The out-of-pocket component of travel cost was computed as the round-trip travel distance multiplied by \$0.25 per mile.[9] The opportunity cost of time was calculated as one-third the estimated round-trip travel time multiplied by the respondent's average wage rate. Table 3 provides summary statistics for the resulting travel cost variable. The average price of a recreational trip to a lake is \$136, although perhaps a more meaningful statistic is the

[9] *PCMiler (Streets Version 17)* was used to compute both round-trip travel distance and time.

average price of a lake visit, $85.

Second, lake site characteristics were obtained from the Iowa Department of Natural Resources [11]. Table 3 provides a summary of these site characteristics. As Table 3 indicates, the size of the lakes varies considerably, from 10 acres to 19,000 acres. Four dummy variables are included to capture different amenities at each lake. The first is a "ramp" dummy variable which equals one if the lake has a cement boat ramp, as opposed to a gravel ramp or no boat ramp at all. The second is a "wake" dummy variable that equals one if motorized vessels are allowed to travel at speeds great enough to create wakes and zero otherwise. About sixty-six percent of the lakes allow wakes, whereas thirty-four percent of lakes are "no wake" lakes. The "state park" dummy variable equals one if the lake is located adjacent to a state park, which is the case for 38.8 percent of the lakes in our study. The last dummy variable is the "handicap facilities" dummy variable, which equals one if handicap amenities are provided, such as handicap restrooms or paved ramps. A concern may be that handicap facilities would be strongly correlated with the state park dummy variable. However, while fifty of the lakes in the study are located in state parks and fifty have accessible facilities, only twenty six of these overlap. Finally, a "fish index" variable is included that varies from zero to four representing the number of fish species for which the lake is considered one of the "top 10" sites for the species in the state.[10]

3 The Model

The mixed logit model was chosen because it exhibits many desirable properties, including that "...it allows for corner solutions, integrates the site selection and participation decisions in a utility consistent framework, and controls for the count nature of recreation demand" (Herriges and Phaneuf, [9]).

[10] The candidate fish species are Bluegill, Brappie, large-mouth bass, catchfish, bullhead, and walleye. See [12] for details.

Assume the utility of individual i choosing site j on choice occasion t is of the form

$$U_{ijt} = V\left(X_{ij};\beta_i\right) + \varepsilon_{ijt},\ i = 1, ..., N;\ j = 0, .., J;\ t = 1, ..., T \tag{1}$$

where V represents the observable portion of utility, and from the perspective of the researcher, ε_{ijt}, represents the unobservable portion of utility. A mixed logit model is defined as the integration of the logit formula over the distribution of unobserved random parameters (Revelt and Train, [19]). If the random parameters, β_i, were known then the probability of observing individual i choosing alternative j on choice occasion t would follow the standard logit form

$$L_{ijt}\left(\beta_i\right) = \frac{\exp\left(V_{ijt}\left(\beta_i\right)\right)}{\sum\limits_{k=0}^{J} \exp\left[V_{ikt}\left(\beta_i\right)\right]}. \tag{2}$$

Since the β_i's are unknown, the corresponding unconditional probability, $P_{ijt}\left(\theta\right)$, is obtained by integrating over an assumed probability density function for the β_i's. The unconditional probability is now a function of θ, where θ represents the estimated moments of the random parameters. This repeated Mixed Logit model assumes the random parameters are $i.i.d.$ distributed over the individuals so that

$$P_{ijt}\left(\theta\right) = \int L_{ijt}\left(\beta\right) f\left(\beta|\theta\right) d\beta. \tag{3}$$

No closed-form solution exists for this unconditional probability and therefore simulation is required for the maximum likelihood estimates of θ.[11]

Following Herriges and Phaneuf [9], a dummy variable, D_j, is included which equals one for all of the recreation alternatives ($j = 1, \ldots, J$) and equals zero for the stay-at-home option ($j = 0$). Including the stay-at-home option allows a complete set of choices, including in the population those individuals who always "stay at home" on every choice occasion and

[11] Randomly shifted and shuffled uniform draws are used in the simulation process (Hess, Train, and Polak, [10]). The number of draws used in the simulation is 750.

do not visit any of the sites. It is convenient to partition the individual's utility into the stay-at-home option or choosing one of the J sites, with

$$U_{ijt} = \begin{cases} \tilde{\beta}' z_i + \varepsilon_{i0t} \\ \beta_i' x_{ij} + \alpha_i + \varepsilon_{ijt}, \ j = 1, ..., J, \end{cases} \tag{4}$$

where α_i is the random parameter on the dummy variable, D_j, which does not appear since it equals one for $j = 1, ..., J$ and zero for $j = 0$. The vector z_i contains socio-demographic data such as gender, age, and education, and x_{ij} represents the site characteristics that vary across the lakes, including attributes such as facilities at the lake as well as water quality measures. Notice that the parameters associated with the socio-demographic data are not random as this information does not vary across the sites.[12]

The random coefficient vectors for each individual, β_i and α_i, can be expressed as the sum of population means, b and a, and individual deviation from the means, δ_i and γ_i, which represents the individual's tastes relative to the average tastes in the population (Train, [20]). Therefore, we can redefine

$$\beta_i' x_{ij} = b' x_{ij} + \delta_i' x_{ij} \tag{5}$$

$$a_i = a + \gamma_i \tag{6}$$

and then the partitioned utility is

$$U_{ijt} = \begin{cases} \tilde{\beta}' z_i + \eta_{i0t} \\ b_i' x_{ij} + a + \eta_{ijt}, \ j = 1, ..., J, \end{cases} \tag{7}$$

where

$$\eta_{ijt} = \begin{cases} \varepsilon_{i0t} & i = 1, ..., N; \ t = 1, ..., T \\ \delta_i' x_{ij} + \gamma_i + \varepsilon_{ijt}, \quad j = 1, ..., J; \ i = 1, ..., N; \ t = 1, ..., T \end{cases} \tag{8}$$

is the unobserved portion of utility. This unobserved portion is correlated over sites and trips because of the common influence of the terms δ_i and γ_i, which vary over individuals. For example, an individual who chooses the stay-at-home option for all choice occasions

[12] It is possible to interact the socio-demographic data with the sites if one believed, for example, that age would affect which lake was chosen.

would have a negative deviation from a, the mean of α_i, while someone who takes many trips would have a positive deviation from a, allowing the marginal effect to vary across individuals. However, the parameters do not vary over sites or choice occasions; thus, the same preferences are used by the individual to evaluate each site at each time period. Since the unobserved portion of utility is correlated over sites and trips, the familiar IIA assumption does not apply for mixed logit models.

In particular, we model the utility individual i receives from choosing lake j on choice occasion t as

$$U_{ijt} = \begin{cases} \tilde{\beta}' z_i + \varepsilon_{i0t} \\ -\beta^P P_{ij} + \beta^{q'} Q_j + \beta_i^{a'} A_j + \alpha_i + \varepsilon_{ijt}, \ j = 1, ..., J \end{cases}, \tag{9}$$

where the vector z_i consists of socio-demographic characteristics, P_{ij} is the travel cost from each Iowan's residency to each of the 129 lakes, as calculated with PCMiler. The vector Q_j denotes the physical water quality measures and A_j represents the attributes of the lake. As shown in equation (9), notice that the parameters on the lake attributes and the dummy variable, D_j, are random.

4 Specification and Estimation

While the mixed logit model provides the general framework for our analysis, it does not determine the specific variables to use in the model (e.g., which water quality measures) or the functional form they should take in equation (9) (e.g., linear versus logarithmic). Moreover, economic theory provides little or no guidance in terms of these choices. In order to investigate the model specification issue, we divided the full survey sample into three portions, with one portion each dedicated to model specification, estimation and out-of-sample prediction. We begin this section with a discussion of the model specification process.

4.1 Specification

There are, of course, a large number of potential model specifications given the range of water quality, site characteristics, and household characteristics identified in Tables 1 through 3. We focus on modeling the role of water quality characteristics in determining recreation demand patterns, holding constant the manner in which both socio-demographics and other site characteristics impact preferences. Specifically, socio-demographic characteristics are assumed to enter through the "stay-at-home" option. They include age and household size, as well as dummy variables indicating gender and college education (See Table 1). A quadratic age term is included in the model to allow for nonlinearities in the impact of age. Site characteristics, identified in Table 3, are included with random coefficients. This is to allow for heterogeneity in individual preferences regarding site characteristics, such as wake restrictions and site facilities, observed in previous studies (e.g., Train, [20]). For example, some households may prefer to visit less developed lakes with wake restrictions in place, while others are attracted to sites allowing the use of motorboats, jet skis, etc. It is assumed that the random parameters (β_i^a) are each normally distributed with the mean and dispersion of each parameter estimated.

Even restricting our attention to the water quality characteristics in Table 2, there are a large number of potential model specifications. We focus on five groups of water quality characteristics for the Q_j in equation (9):

- Secchi depth;
- Chlorophyll;
- Nutrients (Total Nitrogen and Total phosphorus);
- Suspended Solids (Inorganic and Organic); and

- Bacteria (Cyanobacteria and Total).

The first four characteristic groups directly impact the visible features of the water quality, making it more likely that households respond to them. Bacteria is included because surveyed households report it to be the single most important water quality concern (See [2]).

Our initial intent was to consider three possible specifications for each of these groups of variables: inclusion linearly, inclusion logarithmically, or exclusion. However, preliminary analysis indicated that these variables individually and as groups were consistently significant at a five percent level or better. Thus, we chose to focus on determining whether each group of factors should enter the model in a linear or logarithmic fashion. This required estimating a total of $2^5 = 32$ versions of the model. The preferred model (see [8] for details) has Secchi depth and suspended entering the model linearly, with the remaining variables entering in a logarithmic fashion. This model is referred to as Model A below. A more complex model, including the remaining water quality variables of pH, alkalinity, silicon, nitrates, and ammonium nitrogen) is referred to as Model B below. These additional variables are entered in a linear form, except for pH which is included quadratically.

4.2 Estimation Results

Given the results from the specification search, three models were using the second third of the sample: Models A and B and a model including only Secchi depth as a measure of water quality (referred to as Model C hereafter). We include Model C to illustrate the consequences of relying on a single measure of water quality, in this case one that is often available to analysts. The resulting parameter estimates are presented in two Tables, 4a and 4b. For all three models, the coefficients for the socio-demographic data, price, and the random coefficients on the amenities are given in Table 4a. Table 4b lists the coefficients for

the physical water quality measures in all three models. All of the coefficients are significant at the one percent level except for a few of the socio-demographic data. For model B, with thirteen physical water quality measures, only the "male" dummy variable is not significant. In model A, household size and the quadratic term on age are insignificant. Note that the socio-demographic data are included in the conditional indirect utility for the stay-at-home option. Therefore, higher-educated individuals, and larger households are all more likely to take a trip to a lake. Age has a convex relationship with the stay-at-home option and therefore has a concave relationship with trips. For model B, the peak occurs at about age 37, which is consistent with the estimate of larger households taking more trips, as at this age the household is more likely to include children. The price coefficient is negative as expected and virtually identical in all three models.

Turning to the site amenities, again all of the parameters are of the expected sign. As the size of a lake increases, has a cement boat ramp, gains handicap facilities, or is adjacent to a state park, the average number of visits to the site increases. Notice, however, the large dispersion estimates. For example, in model A the dispersion on the size of the lake indicates 11.1 percent of the population prefers a smaller lake, possibly someone who enjoys a more private experience. The large dispersion on the "wake" dummy variable seems particularly appropriate given the potentially conflicting interests of anglers and recreational boaters. Anglers would possibly prefer "no wake" lakes, while recreational boaters would obviously prefer lakes that allow wakes. It seems the population is almost evenly split, with 56.9 percent preferring a lake that allows wakes and 43.1 percent preferring a "no wake" lake. Lastly, the mean of α_i, the trip dummy variable, is negative, indicating that on average the respondents receive higher utility from the stay-at-home option, which is expected considering the average number of trips is 6.7 out of a possible 52 choice occasions.

The physical water quality coefficients are reported in Table 4b and are relatively stable

across the two models. For all three models, the effect of Secchi depth is positive, while in both Models A and B organic and inorganic (volatile) suspended solids have a negative impact, indicating the respondents strongly value water clarity. However, the coefficient on chlorophyll is positive, suggesting that on average respondents do not mind some "greenish" water. The negative coefficient on total phosphorus, the most likely principal limiting nutrient, indicates higher algae growth leads to fewer recreational trips. High total nitrogen levels also have a statistically significant and negative impact on recreational utility associated with a site.

Continuing with the additional measures in model B, alkalinity has a positive coefficient, consistent with alkalinity's ability to both act as a buffer on how much acidification the water can withstand before deteriorating and as a source of carbon, keeping harmful phytoplankton from dominating under low CO_2 stress. Since all of the lakes in the sample are acidic (i.e., pH greater than seven), a positive coefficient for alkalinity is expected. The positive coefficient on silicon is also consistent since silicon is important for the growth of diatoms, which in turn are a preferred food source for aquatic organisms. Lastly, pH is entered quadratically, reflecting the fact that low or high pH levels are signs of poor water quality. However, as mentioned, in our sample of lakes all of the pH values are normal or high. The coefficients for pH show a convex relationship (the minimum is reached at a pH of 8.2) to trips, indicating that as the pH level rises above 8.2, trips are predicted to increase. This is the opposite of what we expected.

5 Welfare Calculations

Given the random parameters β_i, the conditional compensating variation associated with a change in water quality from Q to Q' for individual i on choice occasion t is

$$CV_{it}(\beta_i) = \frac{-1}{\beta^p}\left\{\ln\left[\sum_{j=0}^{J}\exp\left(V_{ijt}\left[Q';\beta_i\right]\right)\right] - \ln\left[\sum_{j=0}^{J}\exp\left(V_{ijt}\left[Q;\beta_i\right]\right)\right]\right\}$$

which is the compensating variation for the standard logit model. The unconditional compensating variation does not have a closed form, but it can be simulated by

$$CV_{it} = \frac{1}{R}\sum_{r=1}^{R}\frac{-1}{\beta^p}\left\{\ln\left[\sum_{j=0}^{J}\exp\left(V_{ijt}\left[Q';\beta_i^r\right]\right)\right] - \ln\left[\sum_{j=0}^{J}\exp\left(V_{ijt}\left[Q;\beta_i^r\right]\right)\right]\right\}$$

where R is the number of draws and r represents a particular draw from its distribution. The simulation process involves drawing values of β_i and then calculating the resulting compensating variation for each vector of draws, and finally averaging over the results for many draws. Following Von Haefen [22], 2,500 draws were used in the simulation.

Three water quality improvement scenarios are considered with the results from model A used for all the scenarios. The first scenario improves all 129 lakes to the physical water quality of West Okoboji Lake, the clearest, least impacted lake in the state. Table 5 compares the physical water quality of West Okoboji Lake with the average of the other 128 lakes. All of West Okoboji Lake's measures are considerably improved over the other 128. For example, West Okoboji Lake has slightly over five times the water clarity, measured by Secchi depth, of the other lakes. Given such a large change, the annual compensating variation estimate of $249 for every Iowa household seems reasonable (Table 7). Aggregating to the annual value for all Iowans simply involves multiplying by the number of households in Iowa, which is 1,153,205.[13] Table 7 also reports the average predicted trips before and after the water quality improvement. Improving all 128 lakes to the physical water quality

[13] Number of Iowa households as reported by Survey Sampling, Inc., 2003.

of West Okoboji Lake leads to an increase in average number of trips to 15.2. As expected, the predicted trips to West Okoboji Lake fall by 19.8 percent, from 0.39 average trips per Iowa household to 0.31. Iowans could then choose the nearest lake with the attributes they prefer, instead of traveling further to West Okoboji Lake.[14]

The next scenario is a less ambitious, more realistic plan of improving nine lakes to the water quality of West Okoboji Lake (see Table 5 for comparison). The state is divided into nine zones with one lake in each zone, allowing every Iowan to be within a couple of hours of a lake with superior water quality. The nine lakes were chosen based on recommendations by the Iowa Department of Natural Resources for possible candidates of a clean-up project. The annual compensating variation estimate is $40 for each Iowa household. As expected, this estimate is 16.0 percent of the value if all lakes were improved, even though the scenario involves improving only 7.0 percent of the lakes. This suggests location of the improved lakes is important and, to maximize Iowan's benefit from improving a few lakes, policymakers should consider dispersing them throughout the state.

The last scenario is also a policy-oriented improvement. Currently of the 129 lakes, 65 are officially listed on the EPA's impaired waters list. TMDLs are being developed for these lakes and by 2009 the plans must be in place to improve the water quality at these lakes enough to remove them from the list. Therefore, in this scenario, the 65 impaired lakes would be improved to the median physical water quality levels of the 64 non-impaired lakes. Table 8 compares the median values for the non-impaired lakes to the averages of the impaired lakes. The table indicates that the median values of the non-impaired lakes seem an appropriate choice, with physical water quality measures higher than the averages of the 65 impaired lakes but much below those of West Okoboji Lake. This scenario is valued considerably lower than the first two water quality improvement scenarios. The estimated compensating

[14] West Okoboji Lake, while one of the most popular lakes in the state currently, is far from most population centers in Iowa.

variation per Iowa household is $151. Consistent with this, the predicted trips only increase three percent over the predicted trips with no improvement in water quality. A reasonable conclusion is that Iowans have an abundance of lakes at this threshold level, and bringing the low-quality lakes up to this level is not much of a benefit.

6 Conclusions

The first-year survey of the Iowa Lakes Project gathered information about the recreational behavior of Iowans at 129 of Iowa's principal lakes. These data were combined with extensive physical water quality measures from the same set of lakes gathered by the Iowa State University Limnology Lab. Our analysis, which employs the repeated mixed logit framework, shows that individuals are responsive to physical water quality measures, and it is possible to base willingness-to-pay calculations on improvements in these physical measures. In particular we considered three improvement scenarios, with the results suggesting that Iowans value more highly a few lakes with superior water quality rather than all recreational lakes that have only adequate levels (i.e., sufficient to not be listed as impaired by the Environmental Protection Agency).

A number of important practical findings come directly from this work. Limnologists and other water quality researchers should be interested in the results of this paper, since the general belief is that visitors care about water clarity as measured by Secchi depth (how many meters beneath the surface of the water a Secchi dish is visible) or water quality in general. By estimating the partial effects of a list of physical measures, we have determined which measures significantly affect recreationists' behavior. Limnologists and water resource managers can use this information about what physical lake attributes visitors' trip behavior responds to in designing projects for water quality improvements. Our results indicate water clarity is very important as evidenced by the Secchi dish and suspended solids parameters.

Also, high concentrations of nutrients, in general, are found to decrease recreational trips.

The findings of this study also have direct relevance for environmental protection managers and citizens concerned with water quality in that they can be used to prioritize clean-up activities to generate the greatest recreational benefits for a given expenditure. Not only can the findings be used to determine which lakes to target and in what order to clean them but also the most efficient levels of improvement can be identified.

Table 1. 2002 Iowa Lakes Survey Summary Statistics

Variable	Mean	Std. Dev.	Min.	Max.
Total Day Trips	6.44	10.22	0	52
Male	0.68	0.46	0	1
Age	53.34	16.09	15	82
School	0.66	0.46	0	1
Household Size	2.61	1.30	1	12

Table 2. Water Quality Variables and 2002 Summary Statistics

Variable	Mean	Std. Dev.	Min.	Max.
Secchi Depth (m)	1.17	0.92	0.09	5.67
Chlorophyll (ug/l)	41	38	2	183
NH3+NH4 (ug/l)	292	159	72	955
NO3+NO2 (mg/l)	1.20	2.54	0.07	14.13
Total Nitrogen (mg/l)	2.20	2.52	0.55	13.37
Total Phosphorus (ug/l)	106	81	17	453
Silicon (mg/l)	4.56	3.24	0.95	16.31
pH	8.50	0.33	7.76	10.03
Alkalinity (mg/l)	142	41	74	286
Inorganic SS (mg/l)	9.4	17.9	0.6	177.6
Volatile SS (mg/l)	9.4	7.9	1.6	49.9
Cyanobacteria (mg/l)	295.8	833.1	0.01	7178.1
Total Bacteria (mg/l)	304.8	835.2	3.99	7178.6

Table 3. Summary Statistics for Lake Site Characteristics

Variable	Mean	Std. Dev.	Min.	Max.
Travel Cost	135.79	29.47	94.12	239.30
Acres	672	2,120	10	19,000
Ramp	0.86	0.35	0	1
Wake	0.66	0.47	0	1
State Park	0.39	0.49	0	1
Handicap Facilities	0.39	0.49	0	1
Fish Index	1.01	1.14	0	4

Table 4a. Repeated Mixed Logit Model Parameter Estimates (Std. Errs in Parentheses)[a]

	Model A: 8 WQ Measures		Model B: 13 WQ Measures		Model C: Secchi Depth	
Variable	Mean	Dispersion	Mean	Dispersion	Mean	Dispersion
Male	−7.88*		−5.93*		−6.65*	
	(0.56)		(0.57)		(0.45)	
Age	1.13*		0.31*		−0.23*	
	(0.09)		(0.09)		(0.08)	
Age^2	−0.007*		−0.001		0.006*	
	(0.0008)		(0.0008)		(0.0007)	
School	−2.12*		−3.09*		5.81*	
	(0.55)		(0.58)		(0.53)	
Household	0.75*		−0.29		−0.68*	
	(0.20)		(0.22)		(0.18)	
Price	−0.49*		−0.49*		−0.49*	
	(0.0012)		(0.001)		(0.001)	
Log(Acres)	4.70*	3.81*	4.42*	4.34*	4.02*	3.00*
	(0.09)	(0.07)	(0.10)	(0.08)	(0.07)	(0.06)
Ramp	12.49*	16.86*	12.71*	15.58*	11.51*	17.30*
	(0.93)	(0.90)	(0.89)	(0.86)	(0.88)	(0.65)
Facilities	2.07*	13.46*	2.01*	12.89*	1.37*	13.95*
	(0.29)	(0.34)	(0.28)	(0.33)	(0.26)	(0.29)
State Park	3.90*	12.35*	3.50*	11.81*	4.04*	13.99*
	(0.32)	(0.28)	(0.31)	(0.27)	(0.31)	(0.27)
Wake	3.80*	11.73*	1.94*	11.89*	3.88*	13.98*
	(0.31)	(0.27)	(0.34)	(0.26)	(0.30)	(0.34)
Fish Index	0.04	6.36*	−0.16	6.06*	−0.12	5.41*
	(0.12)	(0.12)	(0.13)	(0.13)	(0.12)	(0.11)
α	−11.90*	2.76*	−11.92*	2.59*	−11.80*	2.76*
	(0.05)	(0.04)	(0.05)	(0.04)	(0.04)	(0.03)

* Significant at 1% level.

[a] All of the parameters are scaled by 10, except α (which is unscaled).

Table 4b. Repeated Mixed Logit Model Parameter Estimates (Std. Errs. in Parentheses)[a]

Variable	Model A: 8 Water Quality Measures	Model B: 13 Water Quality Measures	Model C: Secchi Depth Only
Secchi Depth (m)	2.40* (0.10)	2.44* (0.10)	1.77* (0.06)
Log(Chlorophyll (ug/l))	2.37* (0.16)	2.97* (0.19)	
NH3+NH4 (ug/l)		0.003* (0.0007)	
NO3+NO2 (mg/l)		−0.14 (0.09)	
Log(Total Nitrogen (mg/l))	−1.16* (0.10)	−1.80* (0.34)	
Log(Total Phosphorus (ug/l))	−2.83* (0.17)	−4.36* (0.22)	
Silicon (mg/l)		0.33* (0.04)	
pH		−109.59* (10.01)	
pH^2		6.41* (0.59)	
Alkalinity (mg/l)		0.05* (0.003)	
Inorganic SS (mg/l)	−0.006 (0.005)	−0.027* (0.006)	
Volatile SS (mg/l)	−0.02 (0.02)	−0.03 (0.02)	
Log(Cyanobacteria)	−2.44* (0.12)	−2.27* (0.14)	
Log(Total Bacteria)	3.44* (0.15)	2.92* (0.17)	
LogLik	-37,626.94	-37,579.04	-37,759.20

*Significant at the 1% level.

[a] All of the parameters are scaled by 10.

Table 5. West Okoboji Lake vs. the other 128 Lakes

	West Okoboji Lake	Averages of the other 128 Lakes	Averages of the 9 Zone Lakes
Secchi Depth (m)	5.67	1.14	1.23
Chlorophyll (ug/l)	2.63	41.01	40.13
Total Nitrogen (mg/l)	0.86	2.21	3.64
Total phosphorus (ug/l)	21.28	105.99	91.11
Inorganic SS (mg/l)	1.00	9.45	9.52
Volatile SS (mg/l)	1.79	9.38	8.42
Cyanobacteria (mg/l)	22.00	295.76	948.00
Total Bacteria (mg/l)	24.06	304.78	953.37

Table 6. 64 Non-impaired Lakes vs. the 65 Impaired Lakes

	Median of the 64 Non-impaired Lakes	Averages of the 65 Impaired Lakes
Secchi Depth (m)	1.27	0.81
Chlorophyll (ug/l)	23.25	56.67
Total Nitrogen (mg/l)	1.11	2.31
Total phosphorus (ug/l)	58.79	139.91
Inorganic SS (mg/l)	3.51	14.78
Volatile SS (mg/l)	6.02	12.93
Cyanobacteria (mg/l)	42.96	516.96
Total Bacteria (mg/l)	47.48	528.65

Table 7. Annual Compensating Variation Estimates

a. Using Model A

Average CV	All 128 Lakes Improved to W. Okb.	9 Zone Lakes Improved to W. Okb.	65 Impaired Lakes Improved to Median
Per choice occasion	$4.80	$0.77	$0.29
Per Iowa household	$249.43	$40.12	$15.06
For all Iowa households	$287,648,000	$46,261,000	$17,363,000
Predicted Trips (11.04 with current water quality)	15.22	11.72	11.33

b. Using Model C

Average CV	All 128 Lakes Improved to W. Okb.	9 Zone Lakes Improved to W. Okb.	65 Impaired Lakes Improved to Median
Per choice occasion	$3.95	$0.64	$0.13
Per Iowa household	$205.40	$33.05	$6.82
For all Iowa households	$236,863,000	$38,117,000	$7,862,000
Predicted Trips (11.23 with current water quality)	14.75	11.80	11.37

References

[1] Arbuckle, K.E. and J.A. Downing (2001). "The influence of watershed land use on lake N:P in a predominantly agricultural landscape." *Limnology and Oceanography* **46**: 970-975.

[2] Azevedo, C.D., K.J. Egan, J.A. Herriges, and C.L. Kling (2003). *Iowa Lakes Valuation Project: Summary and Findings from Year One.* Final Report to the Iowa Department of Natural Resources, August.

[3] Bockstael, N.E., W.M. Hanemann, and I.E. Strand (1986). *Measuring the Benefits of Water Quality Improvements Using Recreationd Demand Models*, report presented to the Environmental Protection Agency under cooperative agreement CR-811043-01-0, Washington, D.C.

[4] Chen, H. Z., F. Lupi, and J. P. Hoehn (1999). "An Empirical Assessment of Multinomial Probit and Logit Models for Recreation Demand," in Herriges, J.A., and C. L. Kling, *Valuing Recreation and the Environment: Revealed Preference Methods in Theory and Practice*, Cheltenham, UK: Edward Elgar, pp. 141-161.

[5] Creel, M.D., and J.B. Loomis (1990). "Theoretical and Empirical Advantages of Truncated Count Data Estimators for Analysis of Deer Hunting in California." *American Journal of Agricultural Economics* **72**(2): 434-41.

[6] Creel, M.D., and J.B. Loomis (1992). "Modeling Hunting Demand in the Presence of a Bag Limit, with Tests of Alternative Specifications." *Journal of Environmental Economics and Management* **22**(2): 99-113.

[7] Dillman, D. A. (1978) *Mail and Telephone Surveys – The Total Design Method*, New York: Wiley.

[8] Egan, K.J., J. A. Herriges, C. L. Kling, and J. A. Downing (2007). "Valuing Water Quality as a Function of Water Quality Measures." Working paper.

[9] Herriges, J., and D. Phaneuf (2002). "Introducing Patterns of Correlation and Substitution in Repeated Logit Models of Recreation Demand." *American Journal of Agricultural Economics* **84**: 1076-1090.

[10] Hess, S., K. Train, and J. Polak (2003). "On the use of randomly shifted and shuffled uniform vectors in the estimation of a Mixed Logit model for vehicle choice." Working paper.

[11] Iowa Department of Natural Resources (2004a), *Fishing Guide for Iowa Lakes.*

[12] Iowa Department of Natural Resources (2004b), *Iowa Conservationist,* **63**(2): 7-25.

[13] Leamer, E.E. (1983). "Let's Take the Con out of Econometrics." *American Economic Review* **73**(1): 31-43.

[14] McFadden, D., and K. E. Train (2000). "Mixed MNL Models for Discrete Response." *Journal of Applied Econometrics* **15**(September/October): 447-70.

[15] Morey, E.R., R. D. Rowe, and M. Watson (1993). "A Repeated Nested-Logit Model of Atlantic Salmon Fishing." *American Journal of Agricultural Economics* **75**(August): 578-592.

[16] Parsons, G.R., E. C. Helm, and T. Bondelid (2003). "Measuring the Economic Benefits of Water Quality Improvements to Recreational Users in Six Northeastern States: An Application of the Random Utility Maximization Model." Working paper, University of Delaware, July.

[17] Parsons, G.R., and M.J. Kealy (1992). "Randomly Drawn Opportunity Sets in a Random Utility Model of Lake Recreation." *Land Economics* **68**(1): 93-106.

[18] Phaneuf, D.J., C.L. Kling, and J.A. Herriges (2000). "Estimation and Welfare Calculations in a Generalized Corner Solution Model with an Application to Recreation Demand." *The Review of Economics and Statistics* **82**(1) (February): 83-92.

[19] Revelt, D., and K. Train (1998). "Mixed Logit with Repeated Choices: Households' Choices of Appliance Efficiency Level." The Review of Economics and Statistics 80: 647-57.

[20] Train, K. (1998). "Recreation Demand Models with Taste Differences Over People." *Land Economics* **74** (2) (May): 230-239.

[21] U.S. Environmental Protection Agency (USEPA) (2000). "Nutrient Criteria Technical Guidance Manual: Lakes and Reservoirs." Office of Water, Office of Science and Technology, Report EPA-822-B00-001, Washington, D.C.

[22] Von Haefen, R. (2003). "Incorporating observed choice into the construction of welfare measures from random utility models." Journal of Environmental Economics and Management 45: 145-65.

Cost-effective provision of ecosystem services from riparian buffer zones

Heidi J. Albers
Oregon State University,
Forest Resources/College of Forestry

Stephen C. Newbold
U.S. Environmental Protection Agency,
National Center for Environmental Economics

R. David Simpson
Johns Hopkins University,
Paul H. Nitze School of Advanced International Studies

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Abstract

The main objective of this paper is to determine the optimal spatial arrangement of riparian buffer zones (RBZs) for protecting salmon species on a branched river network. First, we develop a stylized analytical model of salmon survival on a stream network with three reaches, which reveals some of the general conditions under which RBZs should be concentrated on a few reaches or spread more evenly among multiple reaches. Next, we present a series of simulation experiments based on a calibrated version of the model, which allows us to numerically solve for the optimal distribution of RBZs under a range of initial conditions and parameter values. These simulation experiments reinforce the intuition behind our analytical results and help to further illuminate the quantitative influence of the key parameters of the model, including the spatial distribution of spawning areas, initial temperature conditions, and the size of the conservation budget. This research can help to improve riparian buffer protection guidelines and inform the design of economic incentive schemes for meeting TMDL requirements for a variety of non-point source pollutants including nutrients and high water temperatures.

1 Introduction

Riparian buffer zones are stream-side strips of natural vegetation that serve a variety of ecosystem functions, including filtering polluted non-point source runoff before it enters streams, preventing erosion by stabilizing stream banks, providing nutrients and microhabitat structure from inputs of leaf litter and coarse woody debris, and regulating stream temperature by shading the water surface. These functions in turn contribute to a variety of ecosystem services, including the protection of valued fish stocks and other aquatic species. In this paper we focus on the role of riparian buffer zones in regulating stream temperatures and thereby contributing to the survival of cold water migratory fish species such as salmon.

When Meriwether Lewis and William Clark visited what is now the northwestern U. S. in the early nineteenth century Clark remarked that the number of salmon they encountered was "incredible to say."[1] The explorers grew heartily sick of the ubiquitous fish. Expedition member Patrick Gass recorded that "Being again reduced to fish . . . we made an experiment to vary our food by purchasing a few dogs, and ... felt no disrelish to this new dish."[2] It is ironic that two centuries later several runs of Pacific Salmon are now listed as Federal threatened or endangered species. This situation has led to a number of calls to conserve them. As one might expect of a commercially valuable species with a complicated life cycle ranging over thousands

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[1] Journal of William Clark, 17 October 1805. Available online at http://www.ccrh.org/comm/river/docs/lcww.htm accessed 7 April 2007.

[2] Journal of Patrick Gass, 10 October 1805. The explorers' purchases did create some cultural conflicts, however. The natives were revolted. Gass went on to note that the Nez Perce with whom they traded "have great numbers of dogs, which they employ for domestic purposes, but never eat; and our using the flesh of that animal soon brought us into ridicule as dog-eaters." Quoted in "Lewis and Clark from Other Side", online at http://redwebz.org/modules.php?name=News&file=article&sid=1408 accessed 6 April 2007. Part of the Lewis and Clark expedition's aversion to salmon may also be attributed to ignorance of their biology. Salmon die shortly after spawning and the explorers, on seeing so many dead fish, presumed they must have been diseased.

of kilometers of both fresh and saltwater habitat, the conservation options are as diverse as is the range of the fish. Policies advocated include:

- land use restrictions and best management practices intended to preserve the upstream habitats in which adult fish spawn, eggs hatch, and young fish grow;
- total maximum daily load (TMDL) restrictions on various pollutants that may adversely affect salmon survival, including high temperatures;
- restricting water use to assure adequate flows over dam spillways, road transport of juvenile fish around generating turbines, and—the most extreme option—breaching major dams on the Snake, Columbia, and other rivers;
- programs to control predation from Caspian terns and other animals that feed on salmon in the Columbia Estuary and elsewhere (USACE 1999, Roby *et al.* 2002); and
- reducing fishing pressures on mature fish in marine environments.

The existence of so diverse a range of options guarantees that any particular choice will be controversial. The (opportunity) costs borne by a specific group, be it developers, ranchers, fisherman, or others, differ markedly depending on the option, which guarantees disagreement about the appropriate policy choice. In addition, the best approach may well be a collection of specific policies, but coordinating policy over such a broad range of options, actors, and locations can overwhelm governance systems. The policy of the United States Fish and Wildlife Service in its interpretation and enforcement of the Endangered Species Act suggests a societal commitment to the preservation of Pacific Salmon. Salmon conservation policy, then, might be considered a classic candidate for cost-effectiveness analysis. What is the least expensive way to achieve this objective? What combination of the options under consideration is the best?

Both natural scientists and economists have weighed in on this topic. The scales at which different authors have addressed the issues have varied greatly. Some researchers have developed models to determine the optimal buffer width at a location given the local terrestrial and stream conditions at that point (e.g., Weller *et al*. 1998, Sparovek *et al*. 2002, Matero 2004, Frimpong *et al*. 2005, Tomer *et al*. 2005). This strand of research has inspired policy guidance on the width of RBZs and the composition of vegetation and placement of woody debris within the streams. Moreover, some jurisdictions offer incentives for landowners to establish and maintain RBZs, such as the federal-state partnerships between the USDA and Oregon and Washington that uses the Conservation Reserve Enhancement Program to protect salmon and trout through RBZs.[3]

However, by focusing exclusively on local conditions, these studies and associated policy recommendations take no account of the larger scale, landscape level spatial interactions that also can have a strong influence on the overall effectiveness of RBZs. Considering that conservation budgets often are severely limited, RBZ protection and restoration efforts should be spatially targeted in such a way that the manager's overall conservation goals are achieved as cost-effectively as possible.

Other researchers have developed larger-scale models that explicitly account for the spatial arrangement of buffers in the landscape, typically to maximize the mass of pollutants filtered by RPZs before they enter a river system (e.g., Ferarro 2001, Azzaino *et al*. 2002, and Cerucci and Conrad 2003, Yeo *et al*. 2004). These studies take advantage of the spatial heterogeneity in terrestrial conditions that contribute to NPS pollution at a basin-wide scale to

[3] www.fsa.usda.gov/dafp/cepd/crep.htm

increase the overall cost-effectiveness of RBZ expenditures, but they typically ignore any in-stream processes such as water flows, pollution decay or dilution, or species exposures and mortality rates that also may have strong effects on the provision of ecosystem services.

Research by Wu and colleagues has begun to address the question of optimal allocations of conservation expenditures in the presence of threshold effects in the ecosystem service production functions (Wu and Boggess 1999, Wu *et al*. 2000, Wu and Skelton-Groth 2002). Using the John Day Watershed in Oregon as a case study, Wu and colleagues combined data on standing stocks of fish per reach, a detailed water temperature and flow model, and a cost effectiveness analysis to identify priority reaches for RBZs given various management goals. Their central result places RBZs in fish-bearing, upper reaches of the watershed. Although this research provides useful information and integrates economic and ecological data, the focus on a particular river system and their use of a complex water temperature simulation model make it difficult to identify the drivers of their main results and derive generalizable lessons for other watersheds. Furthermore, Wu and colleagues use data on the relative abundances of spawning salmon as a measure of habitat quality for targeting conservation efforts. No model of salmon migrations is included, so their analysis does not account for the key spatial interaction effects inherent in such a system. In this paper we develop a stylized model of fish and water movement to determine the optimal placement of RBZs in a watershed as a clear function of the parameters of the model. Our policy goal of maximizing the number of fish that survive to the mouth of the river system recognizes that the salmonid lifecycle requires movement through connected reaches in the watershed and accounts for spatial externalities of RBZ restoration.

Another recent contribution to the economic literature on salmon conservation falls, in a sense, at the other end of the spectrum relative to the work of Wu and colleagues. Rather than focusing on watershed protection measures along low order streams, Halsing and Moore (2006) consider 76 management options along the mainstem of the Columbia River for salmon recovery and rank them by their costs effectiveness. The options considered include various combinations of strategies for managing or removing dams, controlling predators, and restricting fishing effort; they do not consider interventions focusing on spawning habitat, as they are unable to obtain sufficient data. Nevertheless, Halsing and Moore's analysis seems a powerful motive for applying cost-effectiveness analysis to upstream habitat conservation options. In their work they find that between 80 and 90 percent of recovery options are clearly dominated by lower-cost alternatives. It would seem, then, quite likely that careful consideration of upstream recovery options via habitat conservation might either reveal them to dominate higher cost strategies in Halsing and Moore's set of preferred alternatives, or to be dominated by them. We are now reporting work in progress that has not yet been fully calibrated to facts on the ground, but we expect the gains from that calibration and comparison to other alternatives will be quite informative to policy.

2 An Analytical Model of Optimal Conservation Efforts

In this section we develop an analytical model of salmon survival on a simplified river network. A set of conditions will emerge that indicate whether expenditures on RBZ protections should be targeted toward a single reach or spread more evenly among multiple reaches. These conditions also will provide a foundation for interpreting some of the numerical results in the following section. Let us consider, then, a simple river network consisting of a

downstream reach, which we will identify with a subscript 3 in the following exposition, and two upstream reaches, identified with subscripts 1 and 2. See Figure 1. Note that this is a very general specification; it could describe any situation above and below the confluence of two streams.

Let N_i denote the number of young salmon hatched in reach *i*. Now suppose that policy makers can devote expenditures, denoted by x_i, along each reach to enhance salmon habitat. Note that these expenditures (or more generally "conservation efforts") could come in a variety of forms, e.g., dollar outlays on restoration projects, opportunity costs of habitat protections, the amount of land set aside or restored to RBZs, etc.

We will suppose that fish survival along reach *i*, p_i, can be described by a logistic function of expenditures on that reach (to allow for threshold effects that are thought to characterize many dose-response functions) as well as expenditures on upstream reaches (because water flows downstream, water quality in upstream reaches influences water quality in downstream reaches). Thus, for our three-reach river network, we have

$$p_1 = \frac{k_1}{1+\beta_1 e^{-\alpha_1 x_1}}, \tag{1}$$

$$p_2 = \frac{k_2}{1+\beta_2 e^{-\alpha_2 x_2}}, \tag{2}$$

and

$$p_3 = \frac{k_3}{1+\beta_3 e^{-(\gamma_1 x_1 + \gamma_2 x_2 + \alpha_3 x_3)}}, \tag{3}$$

where α_i determines the influence of expenditures on reach *i* on fish survival on reach *i*, γ_i determines the influence of expenditures on reach *i* on fish survival on the downstream reach, and k_i and β_i determine the minimum and maximum fish survival on reach *i*.

The following properties of the logistic specification, $p(x) = \dfrac{k}{1 + \beta e^{-\alpha x}}$, will be useful for deriving some of our main qualitative results:

$$p(0) = \frac{k}{1+\beta},$$

$$\lim_{x \to \infty} \quad p(x) = k,$$

$$\frac{\partial p}{\partial x} = \alpha p \left(\frac{k-p}{k} \right),$$

and

$$\frac{\partial^2 p(x)}{\partial x^2} = \alpha^2 p \left(\frac{k-p}{k} \right) \left(\frac{k-2p}{k} \right);$$

that is, the function has an inflection point at $x = \ln(\beta/\alpha)$, where $p = k/2$. We will find it easier to demonstrate our subsequent results if we first invert expressions (1) through (3). Doing so, we have

$$x_1 = \frac{1}{\alpha_1} \ln \frac{\beta_1 p_1}{k_1 - p_1}, \tag{4}$$

$$x_2 = \frac{1}{\alpha_2} \ln \frac{\beta_2 p_2}{k_2 - p_2}, \tag{5}$$

and, since

$$\gamma_1 x_1 + \gamma_2 x_2 + \alpha_3 x_3 = \ln \frac{\beta_3 p_3}{k_3 - p_3},$$

$$x_3 = \frac{1}{\alpha_3}\ln\frac{\beta_3 p_3}{k_3 - p_3} - \left(\frac{\gamma_1}{\alpha_1\alpha_3}\ln\frac{\beta_1 p_1}{k_1 - p_1} + \frac{\gamma_2}{\alpha_2\alpha_3}\ln\frac{\beta_2 p_2}{k_2 - p_2}\right). \tag{6}$$

Using these facts, the total cost of the overall conservation program is

$$x_1 + x_2 + x_3 = \frac{1}{\alpha_3}\ln\frac{\beta_3 p_3}{k_3 - p_3} + \frac{\alpha_3 - \gamma_1}{\alpha_1\alpha_3}\ln\frac{\beta_1 p_1}{k_1 - p_1} + \frac{\alpha_3 - \gamma_2}{\alpha_2\alpha_3}\ln\frac{\beta_2 p_2}{k_2 - p_2}. \tag{7}$$

In general we might consider the problem of conserving fish as a constrained optimization exercise: given a specified limit on the sum of expenditures in expression (7), how would we allocate those expenditures across reaches to maximize the expected number of surviving fish? Such a problem is typically set up as a Lagrangean maximization exercise with a multiplier that reflects the shadow value of the constraint, which reveals the cost of saving the "marginal fish." For our expositional purposes here, however, it will be convenient to treat a somewhat simpler problem. Suppose that the manager is offered a price per fish saved and then maximizes "profits," defined as the price times the expected number of fish saved less the expenditures devoted to saving them. The answer derived using this approach will be exactly the same as that which would emerge from more general problem when the "price" is exactly equal to the shadow value of the multiplier, but by phrasing the problem in this way we simplify required calculations below.[4] So, suppose that the manager's objective is to maximize

$$\phi p_3\left(p_1 N_1 + p_2 N_2\right) - \left(\frac{1}{\alpha_3}\ln\frac{\beta_3 p_3}{k_3 - p_3} + \frac{\alpha_3 - \gamma_1}{\alpha_1\alpha_3}\ln\frac{\beta_1 p_1}{k_1 - p_1} + \frac{\alpha_3 - \gamma_2}{\alpha_2\alpha_3}\ln\frac{\beta_2 p_2}{k_2 - p_2}\right), \tag{8}$$

[4] The reason for this is that derivation of second-order (sufficient) conditions for an optimum in a Lagrangean optimization problem require verification of determinant conditions on a 4 x 4 bordered Hessian matrix of second derivative. By assuming a fixed price of fish we can confine our attention to the 3 x 3 sub-matrix of second derivatives in what follows.

where ϕ is the price of fish. Note the implication of inverting expressions (1), (2), and (3) to derive (4), (5), and (6) , and, by extension, (7) and (8). We treat the manager as choosing probabilities of survival along each reach and, in so doing, committing to the expenditures required to achieve those survival probabilities. First-order conditions for this optimization problem are, with respect to p_1,

$$\phi p_3 N_1 - \frac{\alpha_3 - \gamma_1}{\alpha_1 \alpha_3} \frac{k_1}{p_1 \left(k_1 - p_1 \right)} \leq 0 , \tag{9}$$

with respect to p_2,

$$\phi p_3 N_2 - \frac{\alpha_3 - \gamma_2}{\alpha_2 \alpha_3} \frac{k_2}{p_2 \left(k_2 - p_2 \right)} \leq 0 , \tag{10}$$

and with respect to p_3,

$$\phi p_1 N_1 + \phi p_2 N_2 - \frac{k_3}{\alpha_3 p_3 \left(k_3 - p_3 \right)} \leq 0 . \tag{11}$$

Each of these conditions could be an inequality. None of the p_i's can be less than $k_i / \left(1 + \beta_i \right)$. In other words, it is not possible to free up further resources to apply along reach *j* by sacrificing survival along reach *i* if the latter is already as low as $k_i / \left(1 + \beta_i \right)$.

Note also that the left-hand sides of expressions (9) and (10) would be positive if $\alpha_3 < \gamma_i$, $i = 1,2$. Referring back to expressions (1) and (2), this would occur if one unit of expenditure on one of the *upstream* reaches had a greater impact on the survival of fish in the *downstream* reach than did one unit of expenditure there. To give an example, this might be the case if controlling temperature or pollution upstream were more important to survival downstream than were expenditures made downstream to control mortality there directly.

To consider further cases under which the first-order conditions might *not* hold as equalities, suppose that expression (9) *is* an equality,

$$\phi N_1 = \frac{\alpha_3 - \gamma_1}{\alpha_3 \alpha_1} \frac{k_1}{p_3 p_1 \left(k_1 - p_1\right)}, \tag{12}$$

and consider the circumstances under which this would imply a contradiction. The denominator on the right-hand side of equation (12) would reach its maximum when p_3 equals its maximum value k_3 and p_1 approaches $k_1/2$. (As will be demonstrated momentarily, satisfaction of the second-order conditions for maximization requires that $p_1 > k_1/2$.) Hence the *minimum* possible value of the right-hand side of (12) is $\frac{\alpha_3 - \gamma_1}{\alpha_1 \alpha_3} \frac{4}{k_1}$. There seems no reason *a priori* to rule out values of the parameters α_1, α_3, γ_1, and k_1 such that certain combinations of the number of fish hatched on reach 1, N_1, and the value assigned to surviving fish, ϕ, were not large enough to justify positive expenditures along reach 1.

Let us proceed for now, however, on the assumption that the first-order conditions (9) – (11) are equalities and continue to characterize the conditions under which interior solutions obtain; that is, the conditions under which it is optimal to devote some expenditures on RBZ protections on all three reaches (and of course when these conditions are not satisfied a corner solution will be optimal). Combining (9) and (10) with (11) when each is an equality,

$$\frac{\alpha_3 - \gamma_1}{\alpha_1} \frac{k_1}{k_1 - p_1} + \frac{\alpha_3 - \gamma_2}{\alpha_2} \frac{k_2}{k_2 - p_2} - \frac{k_3}{k_3 - p_3} = 0 . \tag{13}$$

While expressions for the relationships between survival probabilities can be reasonably compact, closed-form solutions for the probabilities themselves are not. In general they admit

multiple roots, begging the question of which root corresponds to the optimal allocation of expenditures. To further analyze this question, we derive the second-order conditions for optimization. The matrix of second derivatives of the objective function is

$$\begin{pmatrix} \frac{\alpha_3-\gamma_1}{\alpha_1\alpha_3}\frac{k_1(k_1-2p_1)}{p_1^2(k_1-p_1)^2} & 0 & \phi N_1 \\ 0 & \frac{\alpha_3-\gamma_2}{\alpha_2\alpha_3}\frac{k_2(k_2-2p_2)}{p_2^2(k_2-p_2)^2} & \phi N_2 \\ \phi N_1 & \phi N_2 & \frac{k_2(k_3-2p_3)}{\alpha_3 p_3^2(k_2-p_3)^2} \end{pmatrix} \tag{14}$$

and simplifying by substituting from the first-order conditions,

$$\phi\begin{pmatrix} p_3N_1\frac{k_1-2p_1}{p_1(k_1-p_1)} & 0 & N_1 \\ 0 & p_3N_2\frac{k_2-2p_2}{p_2(k_2-p_2)} & N_2 \\ N_1 & N_2 & (p_1N_1+p_2N_2)\frac{k_3-2p_3}{p_3(k_3-p_3)} \end{pmatrix}. \tag{15}$$

For the second-order conditions to be satisfied each of the diagonal terms must be negative. This requires that each p_i exceeds its corresponding $k_i/2$. In other words, the optimal expenditure along each reach will be positive if diminishing returns obtain along each.

Another necessary condition for the second-order conditions to be satisfied is that the principle minors must alternate in sign, which implies that

$$p_3^2N_1N_2\frac{k_1-2p_1}{p_1(k_1-p_1)}\frac{k_2-2p_2}{p_2(k_2-p_2)}>0,$$

$$(p_1N_1+p_2N_2)\frac{k_2-2p_2}{(k_2-p_2)}\frac{k_3-2p_3}{(k_3-p_3)}>p_2N_2,$$

and

$$\left(p_1N_1 + p_2N_2\right)\frac{k_1 - 2p_1}{\left(k_1 - p_1\right)}\frac{k_3 - 2p_3}{\left(k_3 - p_3\right)} > p_1N_1 .$$

A sufficient condition for the first of these conditions to be met is that the conditions on the diagonal elements are met. A sufficient condition for the second and third conditions to be met is that each p_i exceeds its corresponding $2k_i/3$.

The final necessary condition for the second-order conditions to be satisfied is that the determinant of the entire matrix is negative, which will be true if

$$\left(p_1N_1 + p_2N_2\right)\frac{k_1 - 2p_1}{p_1\left(k_1 - p_1\right)}\frac{k_2 - 2p_2}{p_2\left(k_2 - p_2\right)}\frac{k_3 - 2p_3}{\left(k_3 - p_3\right)} < N_2\frac{k_1 - 2p_1}{p_1\left(k_1 - p_1\right)} + N_1\frac{k_2 - 2p_2}{p_2\left(k_2 - p_2\right)} . \qquad (16)$$

A *sufficient* condition is that each p_i exceeds its corresponding $2k_i/3$.

These analytical results may not be immediately transparent, so it may be useful to attempt an intuitive explanation of our findings. Recall that the logistic function has an inflection point at $p_i = k_i/2$. It is not surprising, then, that $k_i/2$ is an important benchmark for our results. If none of the p_i's were as large as its corresponding $k_i/2$, there would be increasing returns to additional expenditures on each reach, and all expenditures should be dedicated to the reach for which the marginal contribution to the probability of survival is highest.

These results also reveal that as more money is allocated to restoring salmon—in our model, as a higher price, ϕ, is put on the salmon that survive—it will make sense to allocate expenditures across more than one reach: diminishing returns will eventually set in along the first reach to which expenditures are allocated. The other conditions we derive for an optimum

indicate that the probability of survival must rise to some level discretely greater than $k_i/2$ along each of two reaches before it becomes optimal to allocate expenditures across all three reaches simultaneously. Heuristically, it will be the case that the marginal product of expenditure in enhancing the likelihood of survival along each of the reaches to which positive expenditures are allocated is declining when the probability of fish survival along each of two reaches receiving positive expenditure is greater than $k_i/2$. However, that marginal product still may exceed the marginal product of expenditure in enhancing the likelihood of survival along the last of three reaches to which effort is allocated.

That an interior solution requires decreasing returns to each input is a familiar condition for any constrained optimization problem, but these second order conditions play a key role in determining the nature of the optimal allocation of RBZs and they arise due to the particular structure of our model system: with threshold effects in the survival functions along each reach, under some conditions the objective function will exhibit increasing returns. Thus, as in other ecosystem management problems, the non-convex ecological production functions require special attention and can give rise to outcomes such as "specializing" or focusing policy on one area even when the two areas are identical (Swallow *et al*. 1997, Wu and Boggess 1999), and can lead to difficulties for management including multiple local optima and the inability of decentralized incentive schemes to generate efficient outcomes (Dasgupta and Maler 2004).

3 Simulation Experiments

In this section we present simulation results from a numerical version of the model analyzed above. This requires calibration of the logistic function parameter values to represent

the stream temperature-fish survival relationship (plus some modest changes in notation in the process, which we point out as they arise). Specifically, we calibrated the model to match the following temperature dose-response relationship estimated by McHugh *et al*. (2003) using data from the Snake River in Oregon:

$$S_t = \exp\left[-\left(\frac{T_t}{27.0271}\right)^{10.74}\right], \tag{17}$$

where S_t is the daily survival rate of spring Chinook salmon and T_t is the average daily temperature. To estimate survival probabilities for the entire portion of the life cycle during which young salmon migrate from their spawning streams to the mainstem, McHugh *et al*. assumed that daily survival probabilities are independent, which implies that the overall survival probability is the product of the daily survival probabilities that constitute the full duration of this portion of the life cycle.

To calibrate our simulation model, we translated this resulting seasonal survival probability to a logistic form, where the average seasonal temperature was the argument of the logistic function. (We assumed a day-to-day coefficient of variation of 0.3, which also is loosely consistent data shown in McHugh *et al*.) Figure 2 shows the match between the McHugh model and the logistic model with $\alpha = 1.25$ and $\beta = 1/5{,}700{,}000$. The match is not exact, but it is close enough to ensure that our numerical simulation model will produce realistic results.

Next, we assume a simple linear relationship between riparian shading and average stream water temperatures. On the two upstream reaches the average stream temperatures are

$$T_i = T_i^{\max} + \left(T_i^{\min} - T_i^{\max}\right) x_i \quad i = 1, 2, \tag{18}$$

and on the downstream reach the average stream temperature is

$$T_3 = T_3^{\max} + \left(T_3^{\min} - T_3^{\max}\right)\left(x_3 + \gamma_1 x_1 + \gamma_2 x_2\right), \tag{19}$$

where x_i is the fraction of reach *i* with restored riparian buffers, $T_i^{\max}$ is the current stream temperature on reach *i* (with no riparian restoration), $T_i^{\min}$ is the stream temperature on reach *i* under pristine conditions (if riparian buffers were fully restored along all three reaches), and γ_i is the influence of riparian restoration along (upstream) reach *i* ($i = 1,2$) on the temperature of the downstream reach. Note that our control variables here, the x_i's, represent the fraction of the reach where buffers are restored and so are bounded between 0 and 1 (rather than representing "expenditures" as in the previous section, which were unbounded from above). Furthermore, note that it is the combination of parameters $T_i^{\max}$, $T_i^{\min}$, α_i, and β_i that determine the upper and lower bounds on fish survival along each reach rather than the ("reduced form") parameters k_i, α_i, and β_i used to derive our qualitative analytical results above.

In this application the γ_i's summarize the rate at which heat is gained and lost from the flowing water, and their values will depend on a variety of physical factors including latitude, water flow rates, reach widths, and air temperatures. We expect the γ_i's to be positive but less than one—riparian restoration on a reach should have a stronger influence on the water temperature in that reach than restoration upstream—but otherwise we have no basis for estimating a realistic value for these parameters. Thus, we present results based on a range of γ values, from 0 to greater than 0.5.

The survival probability for salmon migrating through a reach is based on the logistic function, as in the previous section, but here it is calibrated to match the McHugh *et al*. model, and so the argument of the logistic function is the average stream temperature, T_i,

$$p_i = \frac{1}{1+\beta_i e^{\alpha_i T_i}} . \tag{20}$$

The total number of salmon that survive the migration to the downstream end of reach three, *F*, is the number of salmon hatched in reach 1 that survive migration down reaches one and three, the number hatched in reach 2 that survive migration down reaches two and three, and the number hatched in reach 3 that survive migration down reach three, i.e.,

$$F = \left(N_1 p_1 + N_2 p_2 + N_3\right) p_3 . \tag{21}$$

Equation (21) is the objective function, which will be maximized by choosing x_1, x_2, and x_3 in Equations (18) and (19) subject to a constraint on the sum of the *x*'s, i.e., $x_1 + x_2 + x_3 \le X$, which essentially assumes a homogeneous and fixed marginal cost of buffer restoration. (And from the perspective of the dual problem, the sum of the *x*'s chosen optimally give the lowest possible cost of riparian buffer restoration for achieving the resulting level of salmon production, *F*.)

As discussed in the previous section, this optimization problem may be non-convex (here depending on the values of the $T_i^{\max}$'s), so a simple optimization approach using a gradient-based search or first-order conditions alone may not guarantee the globally optimal solution. Thus, we use an exhaustive combinatorial search method—checking every possible combination of *x*'s that sum to the budget constraint, *X* (in increments of *X*/500)—to ensure the globally optimal solution for any combination of parameters and starting values. This approach

is straight-forward and feasible for our current example with only three reaches, but the curse of dimensionality would quickly set in as more reaches are added. Thus, more efficient optimization algorithms will be needed to address real-world cases on river systems that may consist of dozens of reaches. In the meantime, the stylized examples analyzed here can help to build some intuition about what the optimal configuration of riparian buffer restoration will look like under various conditions.

Preliminary sensitivity analyses revealed that, in addition to interior solutions, virtually all combinations of corner solutions are possible. Rather than presenting an exhaustive accounting of all possible combinations, we show in Figures 3 through 5 a few key results that illustrate the effects of the most important model parameters. In each figure, the upper panels graph the fraction of each reach where RBZs are restored (the vertical axis) against the size of the budget (the horizontal axis). The corresponding lower panels in each figure show the number of fish that survive to the mouth of the system at each budget level.

First, consider Figure 3, which illustrates the effect of the initial temperature conditions on the optimal distribution of RBZ restoration. In Panel A the initial temperatures are just beyond the inflection point of the logistic survival curve (refer back to Figure 2), so decreasing returns prevail on each reach. However, since the objective is a multiplicative function of survival along reaches 1 and 3 and 2 and 3, i.e., RBZ restoration on reach 3 enhances the survival prospects of fish hatched on both reaches 1 and 2, when the budget is small it is optimal to target all restoration effort to the downstream reach. If the budget is large enough, then an interior solution is optimal; restoration effort should be split between the three reaches, with equal amounts devoted to each of the two upstream reaches. In Panel B, the initial

temperatures are well below the inflection point of the logistic survival curve, so increasing returns prevail on each reach initially. Thus, when the budget is large enough to justify some restoration effort being devoted upstream, that effort should be targeted to one of the two upstream reaches rather than split evenly between them (even though the initial conditions are the same on the two upstream reaches). At still higher budgets the threshold for the upstream focal reach is achieved and RBZ expenditures occur on all three reaches.

Figure 4 illustrates the influence of the strength of the spillover effects from upstream RBZ restoration on the optimal allocation of restoration effort. In Panel A, γ equals 0.25, which means one unit of RBZ restoration on either reach 1 or 2 will reduce the average temperature on the downstream reach by the same amount as 0.25 units of RBZ restoration on reach 3. These results are qualitatively similar to those in Panel B of Figure 2 where there were no spillover effects, but the switching points between the various corner solutions occur at lower levels of the budget. In Panel B of Figure 4, γ is increased even further to 0.51, which is sufficient to cause a qualitative change in the optimal pattern of RBZ restoration: now at low budgets RBZ restoration efforts should be targeted to one of the two upstream reaches rather than the downstream reach. This concentration of effort occurs because the spillover effects are strong and there are increasing returns to RBZ restoration initially.

Next, Figure 5 illustrates the influence of one of the key spatial features of the objective function on the optimal allocation of RBZ restoration. Here all spawning occurs on only one of the two upstream reaches and there are no spillover effects, so the branching feature of the system is effectively eliminated. Thus, this example isolates the serial (multiplicative) nature of overall fish survival due to the fact that salmon must travel through multiple reaches on their

journey to and from the ocean. In our simplified case, the number of fish that survive to the downstream end of reach 3 is the number hatched in reach 1 that survive their migration through both reach 1 and then reach 3. In Panel A the initial temperatures are above the inflection point of the logistic survival function, so, as we might expect, restoration effort is split evenly between the two reaches. However, in Panel B the initial temperatures are well below the inflection point, so we might expect that, at least when the budget is low, restoration effort should be targeted towards one of the two reaches rather than split evenly between them. However, it turns out that the multiplicative nature of the objective function leads to a switch in the sign of the second derivative compared to the analogous case with an additive objective function.[5] This further highlights the influence of the particular spatial structure of our study system on the qualitative nature of our results. This finding also qualifies one of Wu's key analytical results—that at low budgets and in the presence of threshold effects optimal conservation efforts should be spatially aggregated rather than dispersed (Wu and Boggess 1999)—which apparently depends in part on the fact that their objective was an additive function of watershed-level outputs.

Finally, note that in several of the cases illustrated here, the optimal amount of RBZ restoration does not increase monotonically along each reach as the budget increases (Panel B of Figure 3, and Panels A and B of Figure 4). This can have important consequences for the manager whose budget is not a once-and-for-all endowment but rather comes in regular (typically annual) installments that are often provisional, i.e., not guaranteed and conditional on

[5] The reader can confirm this by comparing a simple "serial" case, $F = p_1(x_1)p_2(X - x_1)$, with an analogous "parallel" case, $F = p_1(x_1) + p_2(X - x_1)$, where the $p_i(\bullet)$'s are based on the logistic function, and taking first and second derivatives of each with respect to x_1.

a variety of factors out of the manager's control. In these cases, if the manager targets funds as they become available to RBZ restoration in areas with the currently highest marginal benefits, and if RBZ restoration projects are not easily reversible, then the manager may find she is stuck with a sub-optimal pattern of RBZs. Thus, in the general case a forward-looking dynamic programming approach will be needed to solve this problem.

4 Conclusions

The ecological and policy literature on salmonid conservation and RBZs contains sometimes conflicting ideas about where to place RBZs in a watershed. Some recommend that RBZs be placed along reaches with warm water that is close to threshold or lethal levels, while others recommend that they be placed upstream where they keep cool water cool. Some state and federal regulations prohibit harvesting trees from riparian areas of salmonid-bearing streams and innovative programs encourage further riparian management of forests (Oregon Forest Practices Act, Kline *et al*. 2000). Recent research has emphasized the contribution of fishless headwater streams to downstream habitat and have called for RBZs in those reaches (IMST 1999, Thompson *et al*. 2007). It appears that there are scientific reasons to have RBZs in many parts of a watershed but, in a world of limited budgets, prioritizing amongst locations is difficult.

The stylized model developed in this paper identifies the role of key parameters and initial conditions in determining where in a watershed to invest in RBZs. In different settings as characterized by these parameter values, the optimal placement of RBZs varies. It seems that no simple rule of thumb will apply uniformly across an ecologically heterogeneous region. However, our results can serve as a starting point for developing some general guidelines based

on a few readily observable characteristics of specific cases. For example, when thresholds in survival matter, and when initial conditions are highly degraded and the budget is small, focusing efforts on a subset of reaches conserves more fish than spreading effort across all reaches. Second, when upstream habitat conditions contribute significantly to downstream species protection, RBZ expenditures should create those spillovers by emphasizing upstream riparian conservation. Third, stream reaches that affect many fish—such as downstream migration corridors—or that are particularly important in the probability of survival—such as those in which salmonids live for extended periods in order to grow strong enough to survive the ocean—should be high priorities because the probability of survival there plays a large role in the overall survival rate. With these state-based rules of thumb in mind, there are situations in which it is best to place RBZs downstream, upstream, and in fishless headwaters but also situations in which those placements are not preferred.

Such rules of thumb can be a useful starting point for managers trying to meet landscape or basin scale species conservation goals, but the real power of this framework will come in specific applications to real-world cases. By expanding the model to a full river network and calibrating it to current temperature conditions and fish survival rates, it should be possible to determine the theoretically optimal distribution of riparian buffer protections and then make concrete suggestions for future RBZ restoration efforts.

Current RBZ regulations and conservation incentives typically do not vary across settings except between fish-bearing and fishless streams. Many policies target stream temperatures for different portions of the salmonid lifecycle but tradeoffs across reaches and watersheds and the connectivity of reaches are not considered. Even the establishment of

critical habitat for salmon has faced boundary difficulties that constrain those designations away from linking conservation upstream to downstream in order to take advantage of spillovers and characteristics of salmonid migration. All of this suggests that significant improvements in the cost-effectiveness of RBZ conservation programs should be possible if the key spatial interactions and tradeoffs across watersheds are properly accounted for in the targeting process.

The framework developed in this paper forms the foundation for several next steps. The work presented here is quite preliminary and our first research priority is to further explore this simple branching system with water and fish movement, with particular attention to identifying characteristics of a setting that lead to different allocations of RBZs across a watershed. One direction of future research involves a fuller representation of the habitat needs across the lifecycle of salmonids and the impact of lifecycle characteristics on the relative importance of reaches within a watershed. A somewhat different direction of research incorporates the RBZ framework developed here into a broader model that elucidates tradeoffs between policy options to conserve salmonids, including breaching dams, limiting harvests, and riparian management. In future work we also intend to calibrate the model to a real river system and attempt to solve the larger network optimization problem and the more general dynamic programming problem accounting for uncertain budgets and discounting. This framework also can serve as a foundation for evaluating the effectiveness of various watershed trading schemes or riparian buffer mitigation banking designs.

References

Azzaino Z, Conrad JM, Ferarro PJ. 2002. Optimizing the riparian buffer: Harold Brook in the Skaneateles Lake Watershed, New York. *Land Economics* 78(4):501-514.

Cerucci M, Conrad JM. 2003. The use of binary optimization and hydrologic models to form riparian buffers. *Journal of the American Water Resources Association* 39(5):1167-1180.

Dasgupta P, Maler K-G, eds. 2004. *The Economics of Non-Convex Ecosystems*. Boston: Kluwer Academic Publishers.

Ferarro P. 2001. Cost-effective targeting of riparian buffers in Georgia when water quality benefits are difficult to measure. Georgia State University, Water Policy Working Paper Series #2001-005.

Frimpong EA, Sutton TM, Lim KJ, Hrodey PJ, Engel BA, Simon TP, Lee JG, Le Master DC. 2005. Determination of optimal riparian forest buffer dimensions for stream biota-landscape association models using multimetric and multivariate responses. *Canadian Journal of Fisheries and Aquatic Sciences* 62:1-6.

Halsing DL, Moore MR. 2006. Cost-effective recovery strategies for Snake River Chinook salmon: a biological-economic synthesis. Draft manuscript.

IMST (Independent Multidisciplinary Science Team), 1999. Recovery of Wild Salmonids in Western Oregon Forests: Oregon Forest Practices Act Rules and the Measures in the Oregon Plan for Salmon and Watersheds. Technical Report 1999-1 to the Oregon Plan for Salmon and Watersheds, Governor's Natural Resources Office, Salem, Oregon.

Kline JD, Ralph JA, Johnson RL. 2000. Forest owner incentives to protect riparian habitat in *Ecological Economics* 33:29-43.

Matero J. 2004. Cost-effective measures for diffuse load abatement in forestry. *Silva Fennica* 38(3):333-345.

McHugh P, Budy P, Schaller H. 2004. A model-based assessment of the potential response of Snake River spring-summer chinook salmon to habitat improvements. *Transactions of the American Fisheries Society* 133:622-638.

Oregon Forest Practices Act. http://www.leg.state.or.us/ors/527.html

Roby DD, Collis K, Lyons DE, Craig DP, Adkins JY, Myers AM. 2002. Effects of colony relocation on diet and productivity of Caspian terns. *Journal of Wildlife Management.* 66:662-673.

Sparovek G, Ranieri SBL, Gassner A, De Maria IC, Schnug E, dos Santos RF, Joubert A. 2002. A conceptual framework for the definition of the optimal width of riparian forests. *Agriculture Ecosystems & Environment* 90:169-175.

Swallow SK, TalukdarP, Wear DN. 1997. Spatial and temporal specialization in forest ecosystrem management under sole ownership. *American Journal of Agricultural Economics* 79:311-326.

Thompson J, McDonell J, Landwehr D. 2007. "Riparian Management of Fishless Headwater Streams in the Tongass National Forest." Abstracted in "Conference Program: Riparian Management in Headwater Catchments: Translating Science into Management." February 19-21, 2007. The University of British Columbia. Vancouver, Canada.

Tomer MD, Dosskey MG, Burkart MR, James DE, Helmers MJ. 2005. Placement of riparian forest buffers to improve water quality. 2005 (Association for Temperate Agroforestry) Conference proceedings.

USACE (U.S. Army Corps of Engineers). 1999. Caspian Tern Relocation Pilot Study FY 1999 Management Plan, Clatsop County, Oregon. USACE, Portland District, Portland Oregon.

Weller DE, Jordan TE, Correll DL. 1998. Heuristic models for material discharge from landscapes with riparian buffers. *Ecological Applications* 8(4):1156-1169.

Wu J, Boggess WG. 1999. The optimal allocation of conservation funds. *Journal of Environmental Economics and Management* 38:302-321.

Wu J, Adams RM, Boggess WG. 2000. Cumulative effects and optimal targeting of conservation efforts: steelhead trout habitat enhancement in Oregon. *American Journal of Agricultural Economics* 82:400-413.

Wu J, Skelton-Groth K. 2002. Targeting conservation efforts in the presence of threshold effects and ecosystem linkages. *Ecological Economics* 42:313-331.

Yeo I-Y, Gordon SI, Guldmann J-M. 2004. Integrated hydrological and land-use model. *Earth Interactions* 8(6):1-20.

Figures

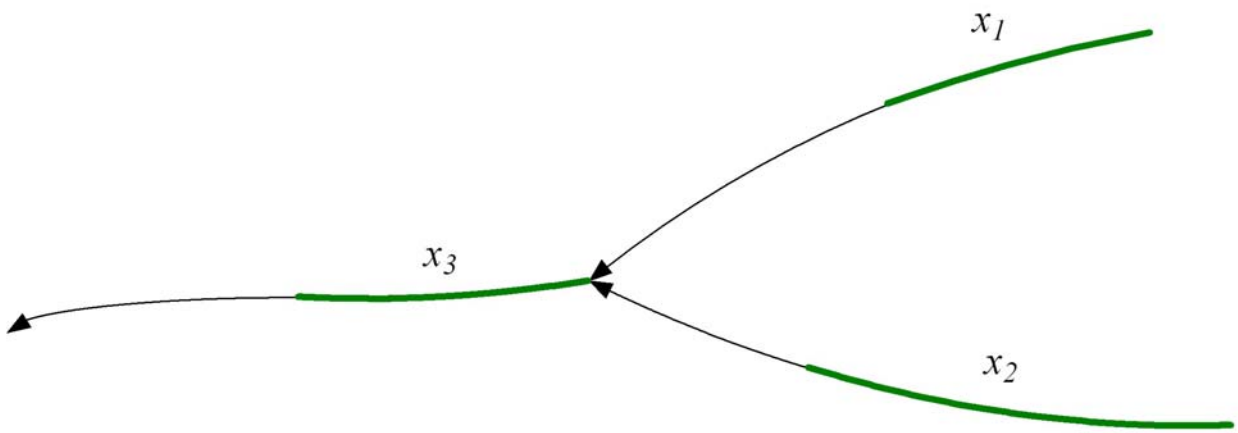


Figure 1 – Schematic representation of a branched stream network, with partial coverage of riparian buffers on each reach.

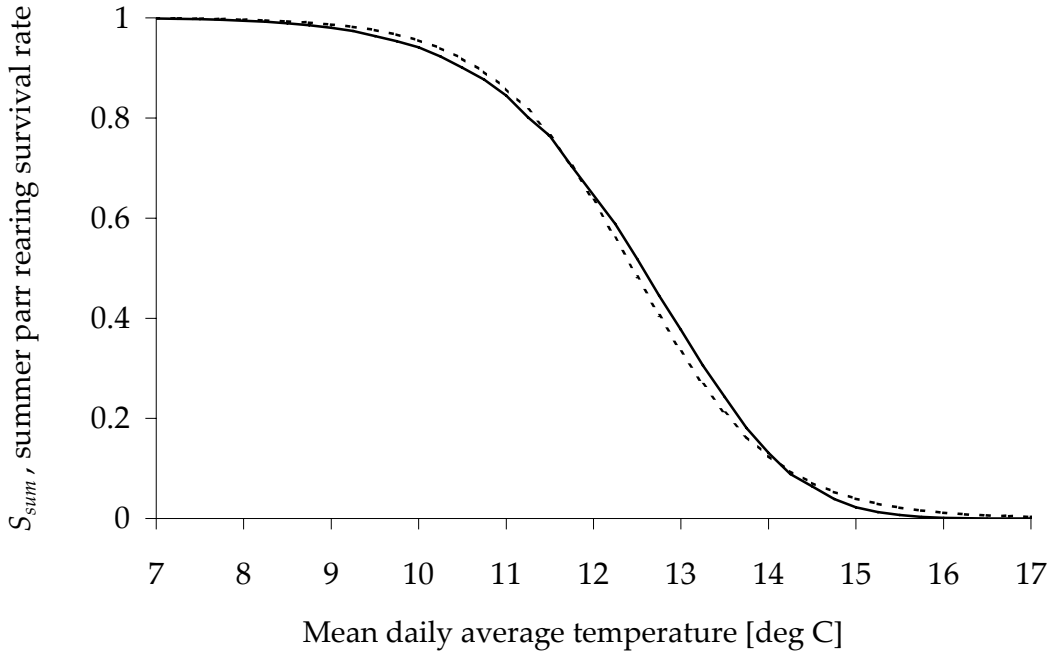
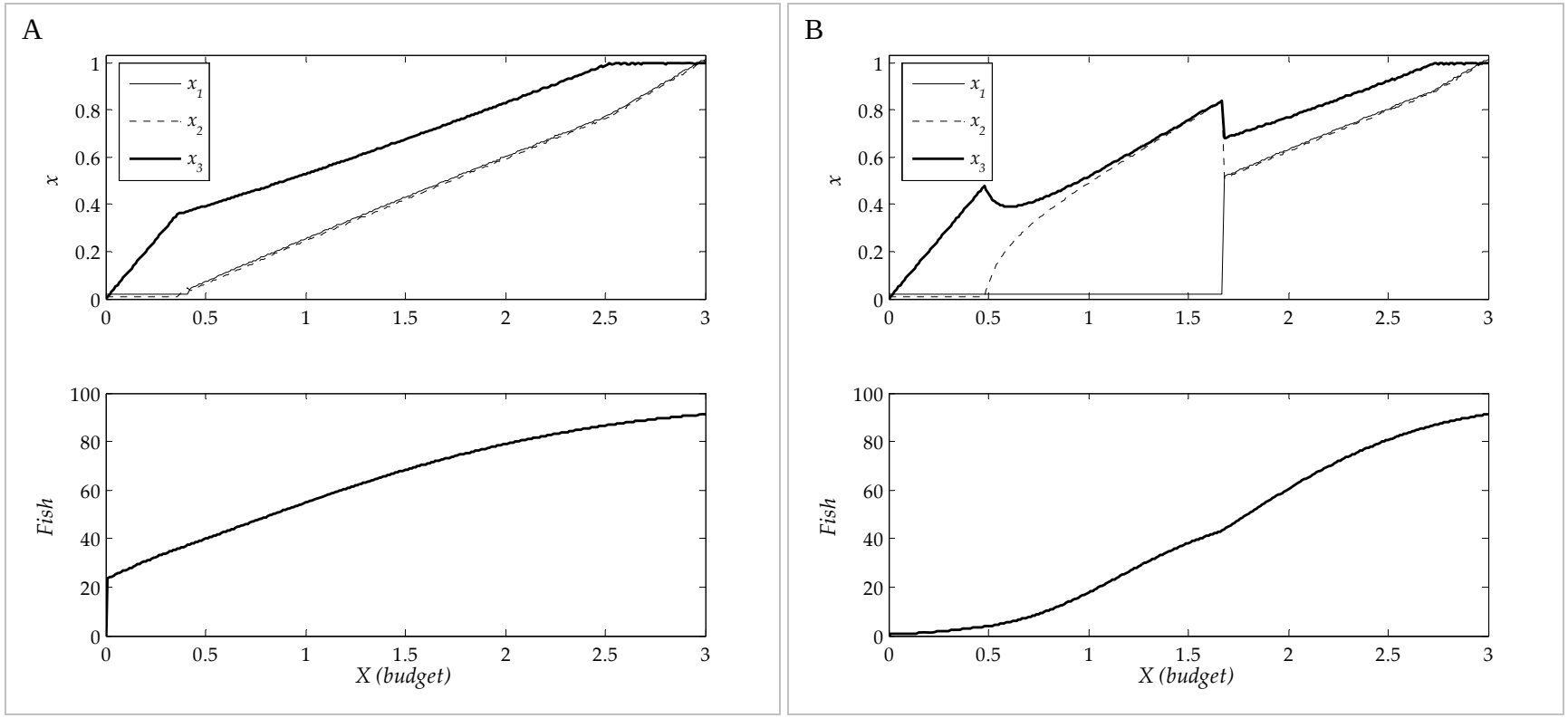


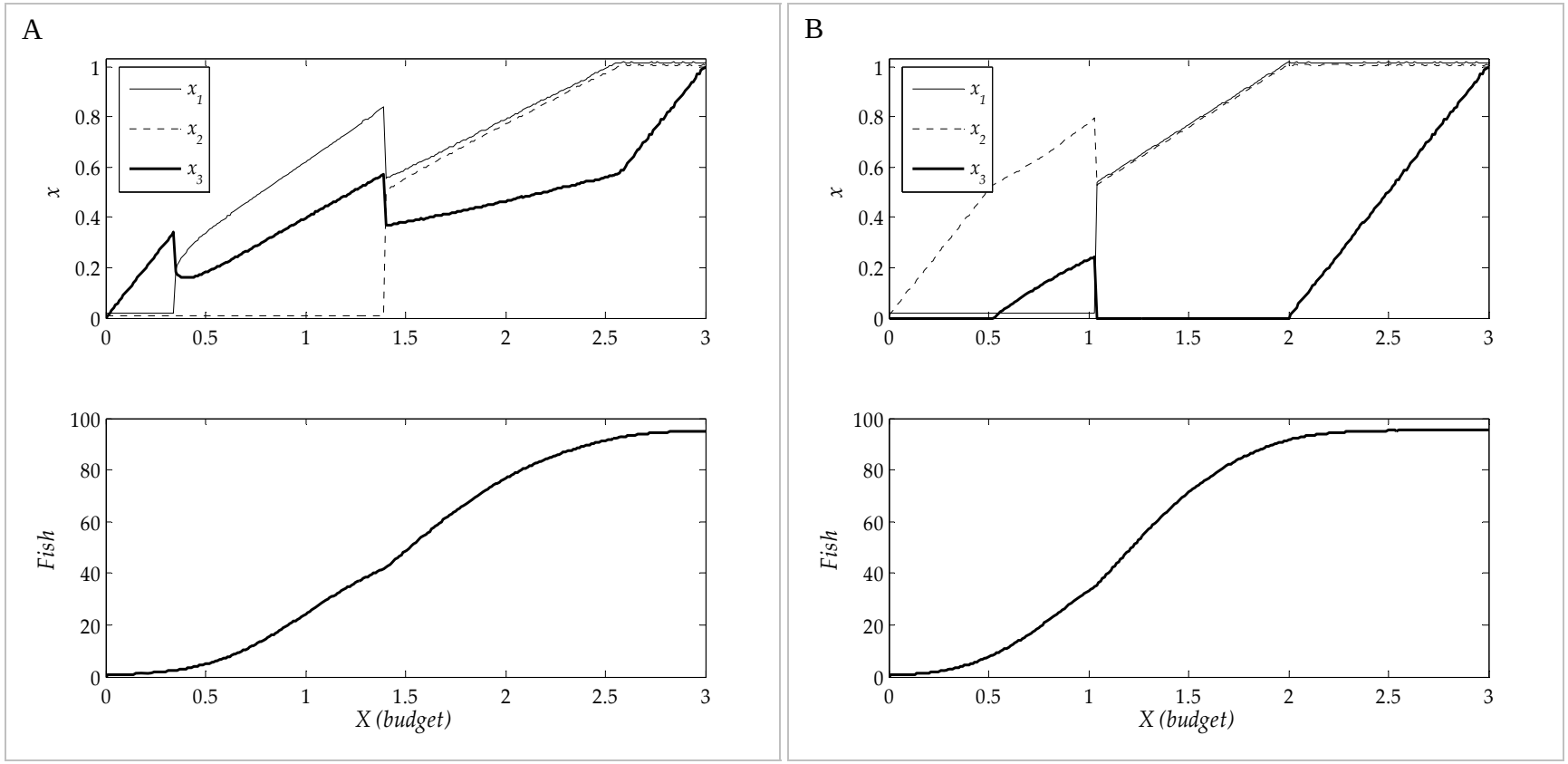
Figure 2 – Seasonal survival probability using the model from McHugh *et al*. (2003) (solid line) and using our calibrated logistic survival function (dotted line).



Reach	1	2	3
N	50	50	0
T^{max}	12.5	12.5	12.5
T^{min}	10	10	10
γ	0	0	--

Reach	1	2	3
N	50	50	0
T^{max}	14.5	14.5	14.5
T^{min}	10	10	10
γ	0	0	--

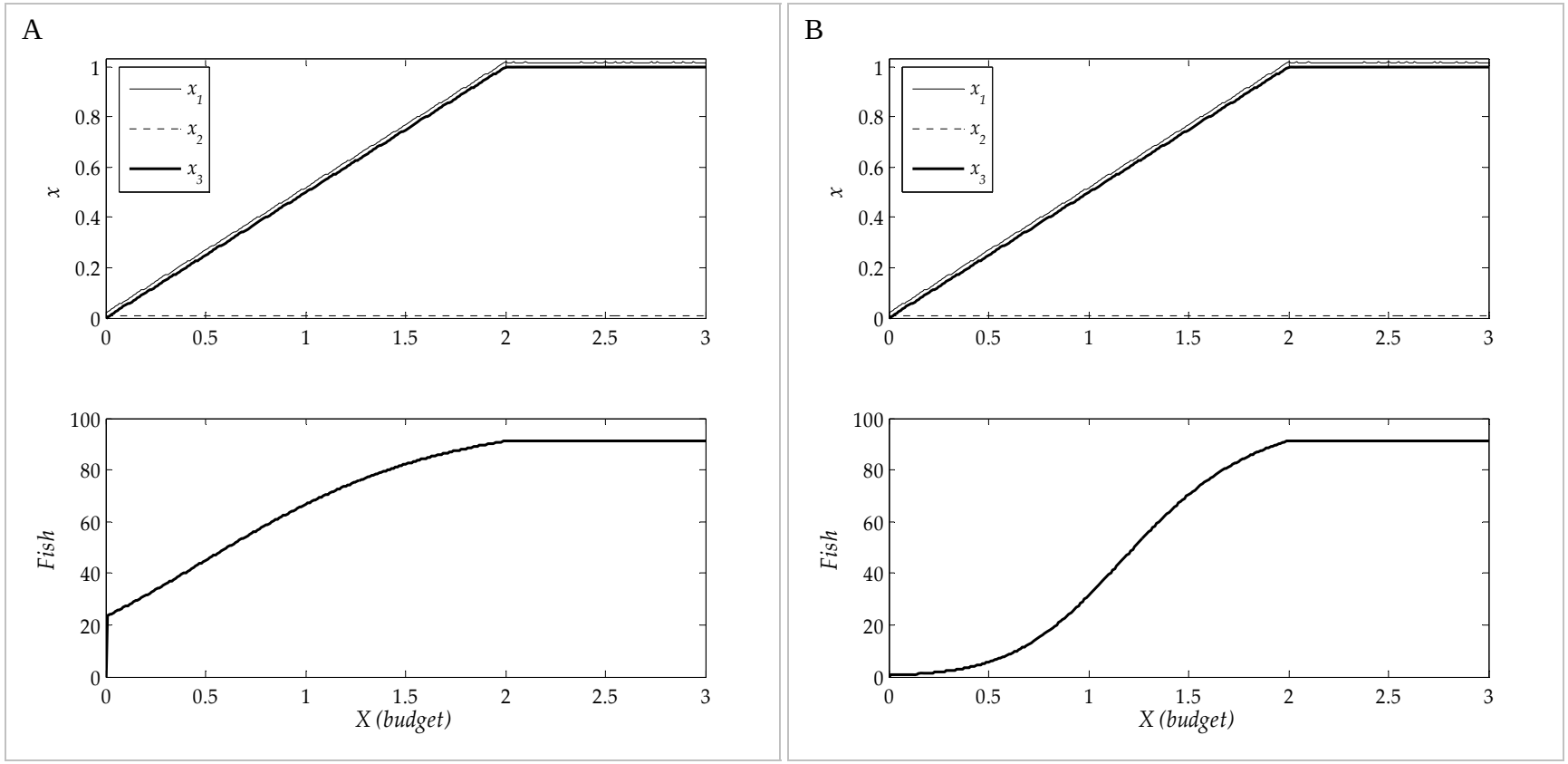
Figure 3 – The influence of initial temperature conditions on the optimal allocation of RBZs.



Reach	1	2	3
N	50	50	0
T^{max}	14.5	14.5	14.5
T^{min}	10	10	10
γ	0.25	0.25	--

Reach	1	2	3
N	50	50	0
T^{max}	14.5	14.5	14.5
T^{min}	10	10	10
γ	0.51	0.51	--

Figure 4 – The influence of the strength of downstream spillovers on the optimal allocation of RBZs.



Reach	1	2	3
N	100	0	0
T^{max}	12.5	12.5	12.5
T^{min}	10	10	10
γ	0	0	--

Reach	1	2	3
N	100	0	0
T^{max}	14.5	14.5	14.5
T^{min}	10	10	10
γ	0	0	--

Figure 5 – The influence of the multiplicative objective function on the optimal allocation of RBZs.

Development of Bioindicator-Based Stated Preference Valuation for Aquatic Resources

Robert J. Johnston
Department of Agricultural and Resource Economics, University of Connecticut

Eric T. Schultz
Department of Ecology and Evolutionary Biology, University of Connecticut

Kathleen Segerson
Department of Economics, University of Connecticut

Elena Y. Besedin
Abt Associates, Inc.

Jessica Kukielka
Department of Agricultural and Resource Economics, University of Connecticut

Deepak Joglekar
Department of Agricultural and Resource Economics, University of Connecticut

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Contact Information: Robert J. Johnston, Connecticut Sea Grant Office, 1080 Shennecossett Rd., Groton, CT 06340. Phone: (860) 405-9278. Email: robert.johnston@uconn.edu.

Abstract

Stated preference (SP) survey techniques are often used to estimate willingness to pay (WTP) for ecological improvements. Describing ecological improvements in a stated preference survey is a challenging task, however, because ecologists often choose assessment endpoints that are relevant to the overall integrity and functioning of the ecosystem but that are not directly linked to economic values or easily understood by survey respondents. This paper presents ongoing progress for an EPA/STAR funded research project, *Improved Valuation of Ecological Benefits Associated with Aquatic Living Resources: Development and Testing of Indicator-Based Stated Preference Valuation and Transfer.* The paper highlights issues related to improving SP estimation of use and non-use values for a specific type of aquatic habitat policy—the restoration of fish passage for diadromous species. We begin by outlining conceptual and practical issues related to the estimation of use and non-use values for aquatic habitat restoration. We then discuss potential limitations in current SP methods used to value aquatic ecosystem changes, and present the framework and theory underlying a proposed, innovative approach to valuation. This approach, denoted Indicator-Based Stated Preference Valuation, seeks to provide more appropriate modeling and representation of aquatic ecosystem attributes within SP valuation, through a more solid grounding in ecological models and a more appropriate communication of ecological change. The paper concludes with an illustration of preliminary survey materials currently under development, grounded in the ISPV framework.

Introduction

Stated preference (SP) valuation methods are often used to estimate the amount individuals would be willing to pay (WTP) for ecological improvements or to avoid a ecological degradation.[1] A number of recent studies, for example, apply SP techniques to assess WTP for policies that affect the ecology of aquatic systems (e.g., Whitehead 1992; Loomis et al. 2000; Johnston et al. 2002; Johnston et al. 2002; Morrison et al. 2002; Boyer and Polasky 2004; Morrison and Bennett 2004; Flores and Shafran 2006). The validity of associated value estimates, however, depends on the appropriate combination of ecological and economic information to derive value estimates (Johnston et al. 2002). A common criticism of SP approaches to aquatic resource valuation is that they do not appropriately reflect the richness and potential complexity of the ecological resources being valued (Simpson 1998; Turner 1999), or are not linked systematically to ecological models (Johnston et al. 2006). There is also disagreement concerning whether this limitation is due to a fundamental and unbridgeable difference between economic and ecological sciences (Turner 1999) or arises from simple miscommunication and/or misinterpretation of ecological models by those seeking to estimate economic values (Simpson 1998).

This paper presents ongoing progress for an EPA/STAR funded research project, *Improved Valuation of Ecological Benefits Associated with Aquatic Living Resources: Development and Testing of Indicator-Based Stated Preference Valuation and Transfer*. The paper highlights issues related to improving SP estimation of use and non-use values for a specific type of aquatic habitat policy—the restoration of fish passage for diadromous species. We begin by outlining conceptual and practical issues related to the estimation of use and non-

[1] In principle, tradeoffs can also be measured using "willingness to accept" measures. However, in practice most applications use WTP because it is more readily estimated.

use values for aquatic habitat restoration in general, and the restoration of fish passage in specific. We then discuss potential limitations in current SP methods used to value aquatic ecosystem changes, and present the framework and theory underlying a proposed, innovative approach to valuation. This approach, denoted Indicator-Based Stated Preference Valuation, seeks to provide more appropriate modeling and representation of aquatic ecosystem health (and change) in SP valuation, through a more solid grounding in ecological models and empirical data and a more appropriate communication of ecological change. Finally, we illustrate preliminary choice experiment questions developed within the ISPV framework, and discuss the differences between such questions and those that are typical within contemporary SP valuation.

Economic Value, Valuation Methods, and Ecological Restoration

Within deliberations for potential natural resource policy changes, natural scientists typically provide data on the components and processes of natural systems. These data may derive from the predictions of ecological models or from the results of empirical field studies of particular systems. From an economic perspective, many of the features of these natural systems represent ecological services that may have value to the public, related to either direct or indirect impacts on human welfare or well-being. The total value realized from policies that restore functions of ecological systems can derive from numerous sources. For example, the restoration of diadromous fish passage yields: (1) potential effects on commercial fisheries may generate market values, (2) impacts on recreational fisheries and other recreational activities (visiting a stream to watch fish runs) may generate substantial non-market use values, as well as (perhaps) market values, (3) restoration of associated ecological functions and condition may generate a wide range of additional use and non-use values.

The remainder of this paper focuses on stated preference valuation of aquatic habitat restoration, and more specifically the restoration of fish habitat or passage in streams and rivers. Stated preference methods, including choice experiments, use surveys to estimate the value people hold for well-defined changes in the quantity or quality of a resource and its services, as measured by what they are willing to give up (in the form of other goods and services they value) in exchange. These approaches in effect construct a hypothetical market using carefully developed survey scenarios. For example, choice experiments present individuals with a set of choices among multiattribute policies, allowing survey respondents to "vote" for the policy they prefer, or to reject all policies in favor of the status quo. By observing respondents' choices over a large number of policies that differ across a range of attributes (including household cost), it is possible to estimate preferences and WTP for different types of restoration programs.

Valid estimation of stated preferences and WTP requires that surveys and estimation methods comply with a wide variety of requirements and guidelines. One of the critical requirements in valuation of ecological restoration is that respondents are provided with sufficient information; analyses of economic value must be based on sound ecological understanding (Bingham et al. 1995), such that respondents can accurately predict the expected influence of the proposed ecological change(s) on their well-being, and hence make rational choices. Put another way, values estimated using SP methods must be viewed as contingent upon the information available to respondents—either as provided by the survey or derived from other sources (e.g., preexisting knowledge) (Bergstrom and Stoll 1989; Bergstrom et al. 1989; Hoehn and Randall 2002; Cameron and Englin 1997). In the view of many ecologists and other natural scientists (as well as some economists), however, stated preference survey instruments often fail to provide sufficient information to enable respondents to understand potential effects

of ecological change on welfare (Spash and Hanley 1995). Moreover, the attributes communicated in these surveys are often ambiguously linked to underlying ecological models or available field study data, providing no clearly defensible means to link estimated preference functions to measurable (or predictable) policy outcomes (Johnston et al. 2006). Finally, the incomplete specification of ecological outcomes may lead respondents to assume particular ecological production function relationships that do not correspond with those identified by ecologists. The following section addresses some of these potential shortcomings, and outlines potential solutions based on the combination of economic stated preference valuation methods with established ecological methods and indicators used to characterize aquatic ecosystem condition.

Valuation of Aquatic Habitat Restoration: Limitations of Current Approaches

Flores and Shafran (2006) review recent studies that seek to estimate values for river and stream restoration. Of 23 studies reviewed (completed since 1999), none address fish passage. Various other studies address improvements in the ecological condition of rivers and watersheds (e.g., Morrison et al. 2002; Holmes et al. 2004; e.g., Morrison and Bennett 2004; Hanley et al. 2006), yet again we are aware of no recently published studies that explicitly address attributes related to the restoration of fish passage through the provision of fish ladders and/or removal of dams. Hence, while aquatic habitat restoration is addressed by a substantial number of past works, the provision of fish passage has been given little attention in the valuation literature.

One of the key issues in survey design is how to define and describe the improvement to ecosystem function (Flores and Shafran 2006, p. 272). While not made explicit by most published SP studies (for an exception, see Holmes et al. 2004), this is most often accomplished

using some form of ecological indicator(s). Ecologists have developed a suite of indices to measure community health and to measure the level of community impacts from stressors, such as species richness and diversity indices and dominance by opportunistic species that is characteristic of a disturbed community (e.g., Davis and Simon 1995; Bortone 2005; Jorgensen et al. 2004). Within stated preference (SP) or survey-based valuation, the role of such ecological indicators is to communicate changes in resource quality or quantity, such that meaningful expressions of value may be elicited (Spash and Hanley 1995). Such information must not only be placed in a format that is readily understood by respondents, but that also provides an accurate representation of the policy change being valued (Mitchell and Carson 1989; Bateman et al. 2002; Spash and Hanley 1995). 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."

Despite the significant attention given to indicator development in the ecological literature, ecological indicators used within SP surveys are often based on *ad hoc* metrics unrelated to formal models of ecosystem change. Measures of change in aquatic living resources presented in SP surveys: (1) rarely correspond to formal indicators presented in the published ecological literature; (2) rarely address uncertainty associated with prediction and measurement; (3) are often ambiguously linked to quantifiable and measurable long-term policy impacts, and; (4) are often based on arbitrary or vague measurement units (Jackson et al. 2000, Ebert and Welsch 2004). For example, Holmes et al. (2004) use an ordinal (hig h, medium, low) "index of naturalness" as an attribute characterizing riparian ecosystems in North Carolina. In the context

of river restoration, Morrison and Bennett (2004) incorporate an attribute communicating the percentage of the river with "healthy riverside vegetation and wetlands." While perhaps easily understandable by respondents, these scales may have little clear meaning within the context of formal ecological models. From the ecologist's perspective, such SP indicators are inadequate.

Indicators commonly used in SP surveys may also fail to communicate critical and welfare-relevant attributes. For example, focus group results reveal that respondents typically view aquatic ecosystem changes relative to historical or pristine conditions (Besedin and Ranson 2004; Johnston et al. 2006b). Common cardinal metrics used in surveys (e.g., 1 million juvenile fish; 20,000 sea birds), in contrast, may have little meaning to respondents.[2] Indeed, with the exception of a few of well-known and tested metrics (e.g., the RFF water quality ladder, Vaughn and Russell 1982), there has been little attention devoted to the development of meaningful (Ebert and Welsch 2004), consistent ecological indicators in SP research.

Using poorly defined indicators may also yield an ambiguous understanding of the goods being valued (i.e., inadequate information regarding specific ecological attributes that influence utility). For example, when presented with an attribute indicating changes in fish populations related to fish passage or other changes, it may be unclear whether resulting WTP estimates reflect a value for the *fish populations themselves*, a value for fish as a proxy for *broader ecological condition*, or a value for fish as a proxy for *some other valued ecosystem function or service*. Do respondents care directly about the loss of a specific number of fish, or do they instead care about other commodities—such as ecosystem health or the probability that fish populations will persist into the future—and use fish numbers as a proxy for the attributes of real concern? Models underlying valuation efforts provide few answers to such questions. In still

[2] This may be one reason why contingent valuation estimates based on such cardinal indicators have been shown to fail scope tests in past assessments (cf. Heberlein et al. 2005).

other cases, SP survey scenarios may confound management *approaches* (what is done) with ecological *outcomes* (what results in terms of measurable ecological change), again leading to the potential for inappropriate respondent assumptions to bias welfare estimates. For example, Yonts et al. (2003) illustrates an SP survey addressing forest biodiversity preservation that characterizes policy outcomes primarily in terms of *what is done* (e.g., the percent of biodiversity reserves, endangered species habitat, and salmon runs either protected or restored; the forest age management approach applied). Left unstated are the measurable ecological improvements that might result, leading to the possibility that respondents may assume an (unstated and likely incorrect) relationship between management actions and ecological outcomes. The resulting ambiguity has significant implications for the validity and transferability of welfare estimates.

The Theoretical Issue

In formal neoclassical economic terms, the question regards the structure of utility. For example, assume that respondent i's utility is of the form $U_i(E(F))$, where $E(\cdot)$ is ecosystem integrity, biodiversity or some other assessment endpoint, and F is the number of diadromous fish (e.g., river herring, shad) passing upstream (or some other measurement endpoint or quantifiable indicator). If the respondent cares about F solely because of its impact on $E(\cdot)$, then a valuation question framed solely in terms of F effectively asks the respondent to directly value changes in the *input* rather than changes in the *output* which influences utility.

The first implication of this is an obvious potential for bias. It is well-known that a preference function of the form $U_i(E(F))$ can be mathematically collapsed to $W_i(F)$. However, if individuals value $E(\cdot)$ directly, the specification of valuation scenarios solely in terms of F 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

F and *E(F)*). As noted by Carson (1998, p. 23), "[r]espondents will tend to fill in whatever details are missing in the CV survey with default assumptions. These may differ considerably from what the researcher perceives." This is particularly true for complex ecological resources and functions, over which respondents may have little baseline information (e.g., Carson 1998; Spash and Hanley 1995). For this reason, it is critical to know if preferences take the form $U_i(E(F))$ rather than the often assumed form $U_i(F)$.

The potential for bias is compounded if utility takes a more complex form such as $U_i(E(F),H(F),F)$, where $H(F)$ represents a second valued ecological attribute influenced by F. For example, $H(\cdot)$ might represent expected quality of recreational fishing for non-diadromous species (e.g., bass), where diadromous fish could have a positive ($\partial H(\cdot)/\partial F > 0$) or negative ($\partial H(\cdot)/\partial F < 0$) marginal effect on recreational fisheries, depending on ecological relationships.[3] Here, F influences utility directly, as well as indirectly its influence on both $E(F)$ and $H(F)$. Moreover, the indirect effects may not be universally positive. In this context, appropriate modeling of utility is critical to obtaining understanding and obtaining unbiased estimates of values for policies that influence F, $E(F)$ and $H(F)$.

The use of oversimplified ecological indicators, moreover, may not yield meaningful valuation for policy purposes. Assume, for example, that ecologists have developed and tested multi-metric indicators of overall ecosystem condition (see additional discussion of these indicators below) that are designed to proxy for true underlying ecosystem condition, $E(\cdot)$, that 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

[3] For example, in some cases diadromous fish may serve as prey for valued recreational species. In other cases diadromous fish may compete for food with various life stages of recreational fish. The balance of such effects may be positive or negative.

survey designer, concerned that respondents will not be able to understand W (or unaware of the appropriate ecological literature), instead uses a simplified indicator (e.g., high-medium-low), which we denote $\widetilde{W}$. The difficulty is that respondents must infer $E(\cdot)$ based on $\widetilde{W}$, with no provided function or information to make this inference; there is no formalized relationship between $\widetilde{W}$ and either W or $E(\cdot)$.[4] Hence, the model estimates WTP for a derived attribute, $\widetilde{W}$, which has no formal ecological meaning or interpretation.

A Proposed Solution: Indicator-Based Stated Preference Valuation

Indicator-Based Stated Preference Valuation (ISPV) is proposed here as a novel approach to welfare evaluation that integrates previously established ecological indicators within SP valuation methodology. ISPV provides a formal basis in established and measurable ecological indicators, a clear structure linking these indicators to specific attributes influencing individuals' well-being, a clear set of criteria to ensure that indicators are consistent and meaningful, and a consequent ability to link welfare measures to measurable and unambiguous policy outcomes. The proposed approach allows the use of measurable indicators within utility-theoretic valuation models, while providing ecological linkages among indicators and assessment endpoints. The result is a multidisciplinary approach to ecological valuation, distinguished from the often *ad hoc*, piecemeal use of ecological information by existing SP valuation.

Figure 1 outlines the basic structural linkages that form the integrated economic-ecological model underlying ISPV. As shown by the illustrated framework, ISPV recognizes that the "assessment endpoints" or fundamental outcomes of fish restoration—as is the case with most environmental policies—are often directly unobservable (US EPA 1998). These are shown

[4] 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. In applied stated preference valuation, the former is (arguably) most common.

as “ecological impacts” in figure 1. For the case of fish passage, ecological impacts may include changes in the survival of fish passing upstream annually, the size and composition of fish populations, and community-level characteristics such as diversity, trophic structure, productivity and resilience. These impacts also represent the attributes that influence people’s well-being (utility).

While often not directly observable, ecological impacts may be measured (and represented) through a set of matching ecological indicators (or measurement endpoints) developed by natural scientists (US EPA 1998). Some of these indicators may be simple and relatively direct—such as the use of fish counter data to measure the number of fish passing upstream. Others are more complex, less direct, and require additional interpretation—such as indices of biological diversity and overall ecological condition. These indicators are mapped to the underlying impacts through established (although sometimes controversial) ecological models (Davis and Simon 1995; Jorgensen et al. 2004; Bortone 2005).

For example, the ecological literature provides numerous integrative, multimetric indices that formally assimilate multiple ecosystem components and are widely used as indicators of ecosystem health or condition (e.g., Davis and Simon 1995; Jorgensen et al. 2004; e.g., Bortone 2005). These indicators (e.g., Index of Biotic Integrity (IBI) and the Estuarine Biotic Integrity Index (EBI)) represent measurement endpoints that can be calculated from data on ecological inputs and can serve as proxies for assessment endpoints that affect utility. Ecologists who have been focused more specifically on species at risk have also developed methods (e.g., population viability analysis or PVA) for projecting population-level uncertain outcomes under alternative management strategies (Lee and Rieman 1997; Beissinger 2002; Jager 2006). If integrated appropriately into valuation models, indices of population viability or ecosystem integrity

provide a basis to distinguish utility gained from specific elements of the ecological system (e.g., populations of specific species) from utility associated with certain elements of overall system condition (e.g., the long term viability of species in given ecosystems).

Primary elements distinguishing ISPV from traditional stated preference valuation hence include:

1. clear distinction between ecological impacts (or assessment endpoints) and indicators (measurement endpoints), as well as between management approaches (what is done) and outcomes (what results);
2. explicit recognition of the potential divergence between actual ecological production functions and those that might be assumed by uninformed survey respondents;
3. explicit recognition of the potential relationship between higher- and lower-level indicators[5] in respondents preference functions, and the possibility that the latter may be used to infer the former;
4. related clear criteria for both indicators and information incorporated in SP surveys, designed to (a) impose an unambiguous link between survey scenarios and well-defined, measurable ecological outcomes and/or ecological model predictions (b) prevent divergence between ecological relationships assumed by respondents and those modeled by ecologists or policy experts (see discussion below), and;

[5] In the context of fish passage restoration, a "higher-level indicator" might include an index of biological integrity, designed to characterize the overall condition and integrity of the aquatic ecological system in question (Karr 1991). Lower-level indicator might include such measures as the number of diadromous fish passing upstream, the percentage of total fish biomass comprised of diadromous species, or the presence of certain indicator species such as freshwater mussels. Respondents may have preferences (and values) over both higher- and lower-level indicators, and may (appropriately or inappropriately) assume relationships between the two.

5. associated improvements in the ability to model linkages between policy impacts and economic values, and hence derive more defensible welfare estimates for ecological policy changes.

To be sure, there are numerous examples of stated preference survey instruments addressing aquatic habitat valuation that incorporate ecological indicators—either simple or complex—as elements in survey scenarios (Morrison et al. 2002; Holmes et al. 2004; Morrison and Bennett 2004; Hanley et al. 2006). *Moreover, it is entirely possible for non-ISPV surveys to derive valid welfare estimates in some cases.* However, the lack of a clear framework conceptualizing distinctions between ecological impacts and ecological indicators—as well as the potential linkages among different types of impacts and indicators—can lead to ambiguity regarding the interpretation of model results, skepticism among natural scientists and potential bias in welfare guidance to policymakers.

Relationships Between Indicators and Survey Attributes

ISPV clearly identifies most attributes presented in stated preference survey scenarios as *indicators* that proxy for the underlying, largely unobservable attributes that influence utility. This recognition also highlights the fact that survey respondents may not be aware of the specific linkages between indicators and underlying conditions assumed by natural scientists. *Hence, unless provided with sufficient information and a set of indicators that map to attributes in their underlying utility functions, individuals may draw erroneous conclusions based on the specific indicator set provided*. This recognition is largely absent from the existing stated preference literature, which typically assumes that respondents will answer questions based solely on the attributes presented in survey scenarios (i.e., that the attributes that are presented are the

attributes that directly enter the utility function).

For example, focus group respondents often reflect general concerns for the overall ecological condition of water bodies affected by policy changes (e.g., Besedin and Ranson 2004). In reaction, surveys such as those of Morrison and Bennett (2004) and Morrison et al. (2002) incorporate a wide range of individual ecological indicators, for example providing information on the presence/absence (or number) of various species and the percentage of river miles supporting "healthy vegetation and wetlands" (Morrison and Bennett 2004). Such surveys, while clearly well-designed according to the standards of the literature, can lead to potential bias and ambiguity when viewed within a more formal ecological-economic framework. For example, figure 1 makes it clear that attributes such as the number of native fish species present can serve as *indicators* for at least two types of underlying assessment endpoints that influence utility: 1) those dealing directly with the size and composition of fish populations, and 2) those dealing with overall system health or integrity. Which of these elements is being valued by respondents who react to the presented "native fish species" attribute is unclear. Moreover, the functional relationship used by respondents to derive information on (2) from the presented survey attributes is also unclear, and almost certainly diverges from that presented in the ecological indicators literature.

For the case of fish restoration, ISPV addresses these challenges through the inclusion of specific "lower-level" indicators for each of the resource-specific attributes that focus groups suggest influence individuals' utility. For example, one might include indicators representing the number of fish passing upstream to spawning areas, the size of spawning areas available to diadromous fish, changes in rare or threatened species that are ecologically dependent on migratory fish, and similar indicators. Following the structure of figure 1, however, one would

also include at least one representation of the change in overall ecosystem condition or population viability resulting from the provision of fish passage. Finally, one would include specific information providing at least some detail regarding the relationship between various indicators presented in survey scenarios (i.e., ecological production functions)—to avoid the potential for erroneous assumptions among respondents.

All ecological indicators, including those representing population and ecosystem status, are also scaled relative to clear benchmarks, providing respondents with unambiguous referents. For example, Population Viability Analysis provides an example of 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 likelihood that a population will persist for a specified period of time under different scenarios (Boyce 1992). Indices of Biotic Integrity provide comparable benchmarks on a system-wide level; they are conventionally scaled relative to least-impacted, relatively pristine reference sites (Jackson et al. 2000).[6] Other lower-level indicators such as fish-counter data may be scaled relative to either historical measures or comparisons to established regional reference sites. The use of well-defined reference points for both baseline conditions and potential ecological change both provides a clear link between survey attributes and established ecological models and practice, and also provides information critical to enable respondents to understand the scope of changes under consideration.

Finally, recognizing the importance of linking estimated preference functions to measurable policy outcomes, ISPV imposes a requirement that indicators used in surveys are both unambiguous from the perspective of natural scientists and are grounded either in

[6] Both PVA and IBI analyses are now commonly used to inform species conservation efforts and/or resource management decisions, and their use in ISPV will therefore strengthen the integration of ecological science and resource management with valuation methods.

potential/actual predictions of established ecological models or in ecological field data that is either currently available or could be made available using standard methods. Moreover, to the extent possible given respondents' cognitive limitations, the basis in measurable data or ecological models should be communicated, at least at a basic level.

While the informational and developmental requirements of ISPV are somewhat greater than those of standard stated preference approaches, the result is a clearer understanding of the attributes driving survey responses, a reduced likelihood that responses are influenced by inappropriate ecological assumptions, and more valid information about public values regarding ecosystem protection. Moreover, the information presentation required for ISPV can be accomplished using established methods—such as the use of video and graphical devices during survey implementation (e.g., Opaluch et al. 1993, Johnston et al. 1999, Johnston et al. 2002).

Criteria for Indicators and Survey Attributes

The proposed valuation approach is designed to generate welfare measures that can be *unambiguously linked to models of ecosystem function*, are based on *measurable outcomes*, and can be more *easily incorporated into benefit cost analysis*. In addition, the approach provides clear mechanisms to distinguish the role of attributes as valued components in and of themselves, versus as indicators of broader ecological condition. As noted above, these goals lead to a set of systematic criteria for survey attributes that are unique to ISPV. These criteria are in addition to the well-known guidance for survey attributes provided by the stated preference literature (e.g., Mitchell and Carson 1989; Bateman et al. 2002; Arrow et al. 1993), and include the following:

1. Ecological survey attributes must have a *systematic linkage to measurable*

indicators representing data that is either commonly available to resource managers, or that could be reasonably obtained using established field methods.[7]

2. Ecological survey attributes must have a *clear, quantitative basis* that is both understood by survey respondents and is meaningful to natural scientists.
3. Ecological survey attributes must be viewed as largely *consistent and meaningful* by natural scientists (subject, of course, to controversies regarding the appropriateness of certain indicators that are a long-standing characteristic of the ecological literature).
4. Ecological survey attributes must be *grounded in identifiable reference conditions*, both in terms of the best and worse outcomes within the realm of practical possibility.
5. Ecological survey attributes must be clearly *associated with specific ecological services* within potential respondents' utility functions.
6. Ecological survey attributes must cover the *full range of primary ecological services* that are both affected by proposed policies (in a significant manner) and valued by the public.
7. To the extent warranted by preliminary research on public preferences (e.g., focus groups), *attributes must communicate both policy approaches (what is done) and ecological outcomes (what results)*.

Violation of any of the seven above-noted criteria can lead to situations in which

[7] Ad hoc or after the fact linkages are insufficient. For example, one might attempt to link a "high-medium-low" scale to measurable bioindicators using a variety of assumptions. However, unless clearly specified to survey respondents prior to completing survey questions, such linkages are ad hoc and non-unique, and will likely lead to biases related to divergence between respondents' perceptions and actual ecological relationships.

associated WTP estimates are either biased or cannot be securely linked to measurable policy outcomes. Hence, while their use may present an additional burden to SP survey designers, these criteria also help ensure that resulting welfare measures are both valid and viewed as appropriate by natural scientists. For example, violation of criterion #1 leads to a potential disconnect between attributes valued by survey respondents and policy outcomes that are either measured or predicted by ecologists. Violation of criterion #6, in contrast, might lead respondents to draw unintended conclusions regarding valued ecosystem services that are nonetheless omitted from survey scenarios (see Johnston and Duke (2007) for a closely related discussion of technical issues in discrete choice preference modeling). As described above, these conclusions will likely be based on incorrect inferences drawn from those ecological attributes that *are* presented in survey scenarios. Similar causal links to welfare estimation biases or limitations may be shown for each of the seven above-noted criteria. Review of SP survey instruments in the published literature, however, shows widespread violation of these criteria. While this does not imply that associated welfare measures are *necessarily* biased, it does lead to questions regarding the interpretation of results for policy guidance.

Within this framework, multimetric indicators of ecological integrity present a particular challenge, given that (1) a variety of such indicators are available, with relatively standardized formats and scaling, (2) the development of such indicators can be highly site-specific, (3) such indicators have not been developed for most aquatic areas, and (4) these indicators can be controversial. Hence, when specifying such indicators in survey scenarios, it may be necessary to generalize at least somewhat, such that subsequently-developed multimetric indicators can be "plugged-in" to estimated preference functions. For example, one might specify survey scenarios based on a multimetric Index of Biotic Integrity (IBI) modeled closely after those

reported in the literature (Jorgensen et al. 2004; Karr 1991), even though a specific IBI has not been developed for the water body in question. This IBI would be specified with the same standardized scales present in published IBIs, such that site-specific IBI results could be later combined with estimated preference functions to provide more exact welfare estimates.[8]

Implications for Benefit Transfer

The proposed ISPV framework and attribute criteria are also designed to promote SP valuation functions that are better suited to benefit transfer. Function-based benefit transfer (including that based on meta-analysis) requires reconciliation of valued policy changes (e.g., changes in valued ecological services) at study site(s) and policy site(s) (Smith et al. 2002). This reconciliation is complicated by the wide array of different measurements and units often applied in SP research to communicate essentially similar ecological changes. Moreover, as noted above, many of these metrics have unspecified or ambiguous linkages to accepted ecological models or data collected by management agencies. The proposed methods, in contrast, are developed around established, well-defined indicators drawn from data commonly collected by management agencies (e.g., the number of diadromous fish passing upsteam, as commonly measured using fish counters installed on fish ladders or other fish passage devices). Hence, the resulting valuation functions are more likely to be directly commensurate with data available for policy sites, allowing more defensible benefit transfer. The proposed methods will also allow more defensible *tests* of benefit transfer performance, as the unambiguous definition

[8] In such cases, respondents would be provided with information regarding the general components of the IBI, but would not be given detailed information regarding the specific mathematical composition of the index. Even if site-specific IBI results were available, it is unlikely that respondents would have the experience or education to understand the complex mathematics that form the basis of many multimetric indicators. Hence, the challenge is to provide sufficient information such that respondents and experts share the same fundamental interpretation of the index, without the requirement that respondents be provided with complex mathematical details.

and common availability of incorporated indicators will ensure that transfer tests are drawn from an appropriate "apples to apples" comparison of ecological change.

Preliminary Survey Scenarios

The proposed approach is currently under the process of empirical development and testing for the valuation of fish passage in New England water bodies. As a result, quantitative results are not yet available. However, numerous expert interviews, four focus groups and significant ecological background research have been conducted over the past 18 months. In addition, the project has drawn from prior results from one dozen transcribed focus groups conducted during the design of a prior stated preference survey for US EPA, addressing similar issues related policy impacts on fish and associated effects on aquatic ecosystem health and integrity (US EPA 2005), as well as evidence gained by the authors in numerous prior stated preference studies addressing WTP for ecological improvements (e.g., Johnston et al. 1999; 2002).[9] This combined research has led to the development of preliminary templates for survey choice questions.

Figures 2a and 2b illustrate an example of the current format of choice experiment questions currently under further development and testing. As shown by figure 2b, survey attributes represent effects on six primary vectors over which focus groups indicated respondents maintain preferences. These vectors characterize (1) the size of aquatic habitat areas restored, or made newly available to diadromous fish, (2) the number of diadromous fish that use the provided fish passage devices to reach the restored upstream river areas, (3) effects on recreational fishing for non-diadromous fishes, (4) effects on rare and threatened aquatic species,

[9] This project addressed use and nonuse values associated with the reduction of entrainment an impingement losses of fish in US water bodies.

(5) effects on other fish dependent wildlife, and (6) effects on overall ecological integrity. Survey attributes provide data both on what is done by managers (e.g., how many acres of habitat are restored) and the ecological outcomes that result (e.g., how many fish pass upstream, effects on freshwater mussels, etc.). Attributes also represent both lower- (e.g., acres restored, fish passing upstream) and higher-level (e.g., ecological integrity) ecological indicators, and survey information materials provide at least rudimentary information on the relationships between the two indicator types.

Each attribute is represented in relative terms, with regard to upper and lower reference conditions defined in survey informational materials. The provided scores represent the percent progress towards the upper reference condition (100%), starting from the lower reference condition (0%). The scenario also presents the cardinal basis for this relative score (e.g., number of acres, number of fish). As noted above, all attributes are grounded in measurable indicators. Prior survey informational materials clarify the specific interpretation of each attribute, the associated reference conditions, and the reason for their inclusion in survey scenarios. These descriptions balance a need to provide sufficient ecological detail with a need to provide concise information that will not exceed the cognitive capacity of respondents.

For example, the Migratory Fish score is described in the survey using the simple text, "the Migratory Fish Score shows the number of fish that swim upstream each year. The score shows the number of fish, as a percent of the maximum possible in the river." The lower (0) and upper (1.2 million) are clearly identified, and are drawn from established models used by the Rhode Island Department of Environmental Management (RIDEM 2006; O'Brien 2006). Prior informational materials also clearly describe the effect of fish passage restoration on migratory fish, along with mechanisms whereby fish passage restoration impacts the other ecological

attributes characterized by the survey question.

The Freshwater Mussels score, in contrast, is motivated by focus group respondents' clear concerns for rare and threatened species, where freshwater mussels are the primary rare species that would be significantly affected by the restoration of fish passage in Rhode Island rivers (Raithel and Hartenstine 2006; O'Brien 2006). The survey first describes the ecological basis for this effect ("…some animals require migratory fish to survive, such as certain freshwater mussels whose larvae need to attach to river herring.") The survey attribute is then characterized: "[t]he Freshwater Mussels Score is based on the presence of rare mussels. These mussels require fish to complete their life cycle. They are indicators of ecosystem health and are endangered in many states. The score shows the percentage of these species that are common in restored areas." Finally, the survey scenario characterizes the effect on these species, relative to the lower (0 species) and upper (5 species) reference baselines (cf. Raithel and Hartenstine 2006), and the current level in the non-restored area (0 species). Similar information and baselines are provided for each survey attribute, as based on preliminary ecological research, interviews with policy and ecological experts, and focus groups with Rhode Island residents. Initial pretests with these questions and attributes have been promising, but further testing and development is required prior to survey implementation.

In addition to the specific attributes present in draft choice questions, the survey booklet also provides information to assist respondents in understanding the general policy and ecological context. For example, respondents in early focus groups requested clarification regarding "whether this issue was important" from a broader ecological perspective, and the extent to which fish passage had been diminished relative to its historical maximum. For this reason, the survey provides numerous maps and other graphics to illustrate and quantify the

broader effects of fish passage on ecological systems, as well as the historical and present extent of habitat available to these species. For example, Figures 3 and 4 represents maps and accompanying text included in the survey materials to illustrate the present and historical extent of fish passage in Rhode Island and the Pawtuxet Watershed; these maps were designed explicitly for the survey based on ecological data from RIDEM and RIGIS map templates (Erkan 2002).

As noted above, these survey materials are under development, with a number of focus groups and expert interviews planned prior to survey implementation. It is likely that numerous changes will be made to survey materials, based on input from these sources. The current illustrations, however, show up-to-date progress towards a survey instrument that complies with the guidelines and criteria of ISPV developed during the initial phases of the research, as summarized above. The goal of these developmental activities is to design a survey instrument that is viewed as equally appropriate by (1) potential respondents, (2) policymakers, (3) ecologists and other natural scientists, and (4) valuation practitioners and other economists.

Conclusion

Current stated preference valuation methods may in many cases provide insufficient information to enable meaningful, unbiased WTP estimation of the value of ecological changes. The research presented here, although still in its early stages, is designed to extend the frontier of valuation research at the intersection of ecology and welfare economics, with a specific emphasis on the critical nature of measurement metrics and structural linkages between ecological and economic models. Unlike prior work, here we emphasize the role of ecological indicators not only as simple and interchangeable means to communicate “well-known” ecological information

(the common economic perspective)—but as a fundamental means to bridge the gap between ecological and economic treatments of policy outcomes. Despite numerous prior efforts to merge ecological and economic information for neoclassical valuation purposes, divergent disciplinary perspectives often lead to an ad hoc treatment of ecological attributes—with little reference to the extensive (and sometimes controversial) work by ecologists to model and measure ecosystem health and condition. The purpose of this research is to ground SP welfare evaluation—both theoretically and empirically—in exactly the types of ecological indicators and associated models developed and tested within the ecological sciences, as well as measurable data available to policymakers. This research also formalizes this "grounding" using a set of derived guidelines and criteria for the development of SP surveys and incorporated ecological indicators. This formal grounding offers many advantages, but also significant challenges, which we will be addressing as we further develop survey materials over the coming months. The goal is a more defensible structure for ecological valuation, as well as applied welfare evaluation viewed as more appropriate by natural scientists.

References

Bateman, I. J., and coauthors. 2002. Economic Valuation with Stated Preference Surveys: A Manual. Edward Elgar, Northampton MA.

Bateman, I. J., and K. G. Willis. 1999. Introduction and Overview. I. J. Bateman, and K. G. Willis, editors. Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries. Oxford University Press, Oxford UK.

Beissinger, S. R. 2002. Population viability analysis: past, present, future. S. R. Beissinger, and D. R. McCullough, editors. Population Viability Analysis. University of Chicago Press, Chicago.

Bergstrom, J.C. and J.R. Stoll. 1989. Application of Experimental Economics Concepts and Precepts to CVM Field Survey Procedures, Western Journal of Agricultural Economics 14 : 98-109.

Bergstrom, J.D., J.R. Stoll and A. Randall. 1989. Information Effects in Contingent Markets, American Journal of Agricultural Economics 71: 321-323.

Besedin, E. Y., and Ranson. 2004. Findings from Focus Groups. Memo to U.S. EPA/OW. Appendices I-III. Focus Group Transcripts. Abt Associates, Cambridge, MA.

Bingham, G., and coauthors. 1995. Issues in ecosystem valuation: improving information for decision making. Ecological Economics 14:73-90.

Boardman, A. E., D. H. Greenberg, A. R. Vining, and D. L. Weimer. 2001. Cost-Benefit Analysis: Concepts and Practice. Prentice Hall, Upper Saddle River NJ.

Bortone, S. A., editor. 2005. Estuarine Indicators. CRC Press, Boca Raton, FL.

Boyce, M. S. 1992. Population viability analysis. Annual Review of Ecology and Systematics

23:481-506.

Boyer, T., and S. Polasky. 2004. Valuing Urban Wetlands: A Review of Non-Market Valuation Studies. Wetlands 24:744-755.

Bryant, B. C., and D. Huppert. 2006. Why "Separating Science and Management" Confuses the Debate over Management Reform in U.S. Fisheries. Fisheries 31:127-129.

Cameron, T.A. and J. Englin. 1997. Respondent Experience and Contingent Valuation of Environmental Goods. Journal of Environmental Economics and Management 33(3): 296-313.

Carson, R.T. 1998. Valuation of Tropical Rainforests: Philosophical and Practical Issues in the Use of Contingent Valuation. Ecological Economics 24(1): 15-29.

Champ, P. A., K. J. Boyle, and T. C. Brown, editors. 2003. A Primer on Non-Market Valuation. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Davis, W. S., and T. P. Simon, editors. 1995. Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making. Lewis Publishers, Boca Raton FL.

Ebert, U., and H. Welsch. 2004. Meaningful Environmental Indices: A Social Choice Approach. Journal of Environmental Economics and Management 47:270-283.

Erkan, D.E. 2002. Strategic Plan for the Restoration of Anadromous Fishes to Rhode Island Coastal Streams. Wakefield, RI: Rhode Island Department of Environmental Management , Division of Fish and Wildlife.

Flores, N. E., and A. Shafran. 2006. A Review of Studies Valuing Ecosystem Improvements and Restoration. Annual USDA W-1133 Meetings, San Antonio, TX.

Freeman, A. M., III. 2003. The Measurement of Environmental and Resource Values: Theory and Methods. Resources for the Future, Washington DC.

Haab, T. C., and K. E. McConnell. 2002. Valuing Environmental and Natural Resources: The Econometrics of Non-Market Valuation. Edward Elgar Publishers, Cheltenham UK.

Hanemann, W. M. 1994. Valuing the Environment Through Contingent Valuation. Journal of Environmental Perspectives 8:19-43.

Hanley, N., R. E. Wright, and B. Alvarez-Farizo. 2006. Estimating the economic value of improvements in river ecology using choice experiments: an application to the water framework directive. Journal of Environmental Management 78:183-193.

Heberlein, T.A., M.A. Wilson, R.C. Bishop, and N.C. Schaeffer. 2005. Rethinking the Scope Test as a Criterion in Contingent Valuation. Journal of Environmental Economics and Management 50(1): 1-22.

Hoehn, J.P. and A. Randall. 2002. The Effect of Resource Quality Information on Resource Injury Perceptions and Contingent Values, Resource and Energy Economics 24 (1) 13-31.

Holmes, T. P., J. C. Bergstrom, E. Huszar, S. B. Kask, and F. Orr, III. 2004. Contingent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. Ecological Economics 49(1):19-30.

Huang, J. C., T. C. Haab, and J. C. Whitehead. 1997. Willingness to pay for quality improvements: Should revealed and stated preference data be combined? Journal of Environmental Economics and Management 34:240-255.

Jackson, L. E., J. C. Kurtz, and W. S. Fisher, editors. 2000. Evaluation Guidelines for Ecological Indicators. EPA/620/R-99/005. US EPA, Washington, DC.

Jager, H. I. 2006. Chutes and ladders and other games we play with rivers. 1. Simulated effects of upstream passage on white sturgeon. Canadian Journal of Fisheries and Aquatic Sciences

63:165-175.

Johnston, R. J., E. Y. Besedin, and M. J. Mazzotta. 2005. Combining Economic and Ecological Modeling to Guide Coastal Ecosystem Restoration: Assessing Tradeoffs in Restoration Options and Economic Values. Coastal Zone 05, New Orleans LA.

Johnston, R.J. and J.M. Duke. 2007. Willingness to Pay for Agricultural Land Preservation and Policy Process Attributes: Does the Method Matter? American Journal of Agricultural Economics, in press.

Johnston, R. J., T. A. Grigalunas, J. J. Opaluch, J. Diamantedes, and M. Mazzotta. 2002. Valuing Estuarine Resource Services Using Economic and Ecological Models: The Peconic Estuary System Study. Coastal Management 30:47-66.

Johnston, R. J., G. Magnusson, M. Mazzotta, and J.J.Opaluch. 2002. Combining Economic and Ecological Indicators to Prioritize Salt Marsh Restoration Actions. American Journal of Agricultural Economics 84:1362-1370.

Johnston, R.J., E.T. Schultz, K. Segerson and E.Y. Besedin. 2006. Development and Testing of Bioindicator-Based Stated Preference Valuation of Estuarine Resources. Annual Meetings of the American Fisheries Society. Lake Placid, NY. September 10-14.

Johnston, R.J., E.T. Schultz, K. Segerson and E.Y. Besedin. 2006(b). Focus Group Recorded Transcripts and Notes. Department of Agricultural and Resource Economics, University of Connecticut.

Johnston, R. J., S. K. Swallow, and T. F. Weaver. 1999. Estimating Willingness to Pay and Resource Trade-offs With Different Payment Mechanisms: An Evaluation of a Funding Guarantee for Watershed Management. Journal of Environmental Economics and Management 38:97-120.

Jorgensen, S. E., R. Costanza, and F. L. Xu. 2004. Handbook of Ecological Indicators for Assessment of Ecosystem Health. CRC Press, Boca Raton FL.

Karr, J.R. 1991. Biological Integrity: A Long-Neglected Aspect of Water Resource Management. *Ecological Applications* 1: 66-84.

Lackey, R. T. 2006. Axioms of Ecological Policy. Fisheries 21:286-290.

Lee, D. C., and B. E. Rieman. 1997. Population Viability Assessment of Salmonids by Using Probabilistic Networks. North American Journal of Fisheries Management 17:1144-1157.

Loomis, J., P. Kent, L. Strange, K. Fausch, and A. Covich. 2000. Measuring the Total Economic Value of Restoring Ecosystem Services in an Impaired River Basin: Results from a Contingent Valuation Survey. Ecological Economics 33:103-117.

Mitchell, R. C., and R. T. Carson. 1989. Using Surveys to Value Public Goods: The Contingent Valuation Method. Resources for the Future, Washington D.C.

Morrison, M., and J. Bennett. 2004. Valuing New South Wales rivers for use in benefit transfer. Australian Journal Of Agricultural And Resource Economics 48:591-611.

Morrison, M., J. Bennett, R. Blamey, and J. Louviere. 2002. Choice Modeling and Tests of Benefit Transfer. American Journal of Agricultural Economics 84:161-170.

O'Brien, J. 2006. Personal Communication 6/29/06. Freshwater & Anadromous Fisheries Management, Division of Fish and Wildlife, Rhode Island Department of Environmental Management.

Opaluch, J. J., K. S. Stephen, W. Thomas, W. W. Christopher, and W. Dennis. 1993. Evaluating Impacts from Noxious Facilities: Including Public Preferences in Current Siting Mechanisms. Journal of Environmental Economics and Management 24:41-59.

Raithel, C.J. and Hartenstine, R.H. 2006. The Status of Freshwater Mussels in Rhode Island.

Northeastern Naturalist 13(1):103–116.

Rhode Island Department of Environmental Management (RIDEM). 2006. Department of Environmental Management Summary of New Initiatives for Anadromous Fisheries Habitat Restoration, Revised 3/27/2006. Internal RIDEM document. Wakefield, RI: Rhode Island Department of Environmental Management, Division of Fish and Wildlife.

Schiller, A., and coauthors. 2001. Communicating Ecological Indicators to Decision Makers and the Public. Conservation Ecology 5:19.

Simpson, R. D. 1998. Economic Analysis and Ecosystems: Some Concepts and Issues. Ecological Applications 8:342-349.

Spash, C.L. and N. Hanley. 2005. Preferences, Information and Biodiversity Preservation. Ecological Economics 12: 191-208.

Turner, R. K. 1999. The Place of Economic Values in Environmental Valuation. I. J. Bateman and K. G. Willis, editors. Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries. Oxford University Press, Oxford UK.

US EPA. 1998. Guidelines for Ecological Risk Assessments, EPA/630/R-95/002F, Washington DC.

US EPA. 2005. Agency Information Collection Activities: Proposed Collection; Comment Request; Willingness To Pay Survey for Section 316(b) Phase III Cooling Water Intake Structures: Instrument, Pre-Test, and Implementation, EPA ICR Number 2155.02 *Federal Register* 70(110), June 9.

Vatn, A., and D. W. Bromley. 1994. Choices without Prices without Apologies. Journal of Environmental Economics and Management 26:129-148

Vaughn, W. J., and C. S. Russell. 1982. Freshwater Recreational Fishing: The National Benefits of Water Pollution Control. Resources for the Future, Washington DC.

Whitehead, J. C. 1992. Economic Valuation of Wetland Resources: A Review of the Value Estimates. Journal of Environmental Systems 22:151-161.

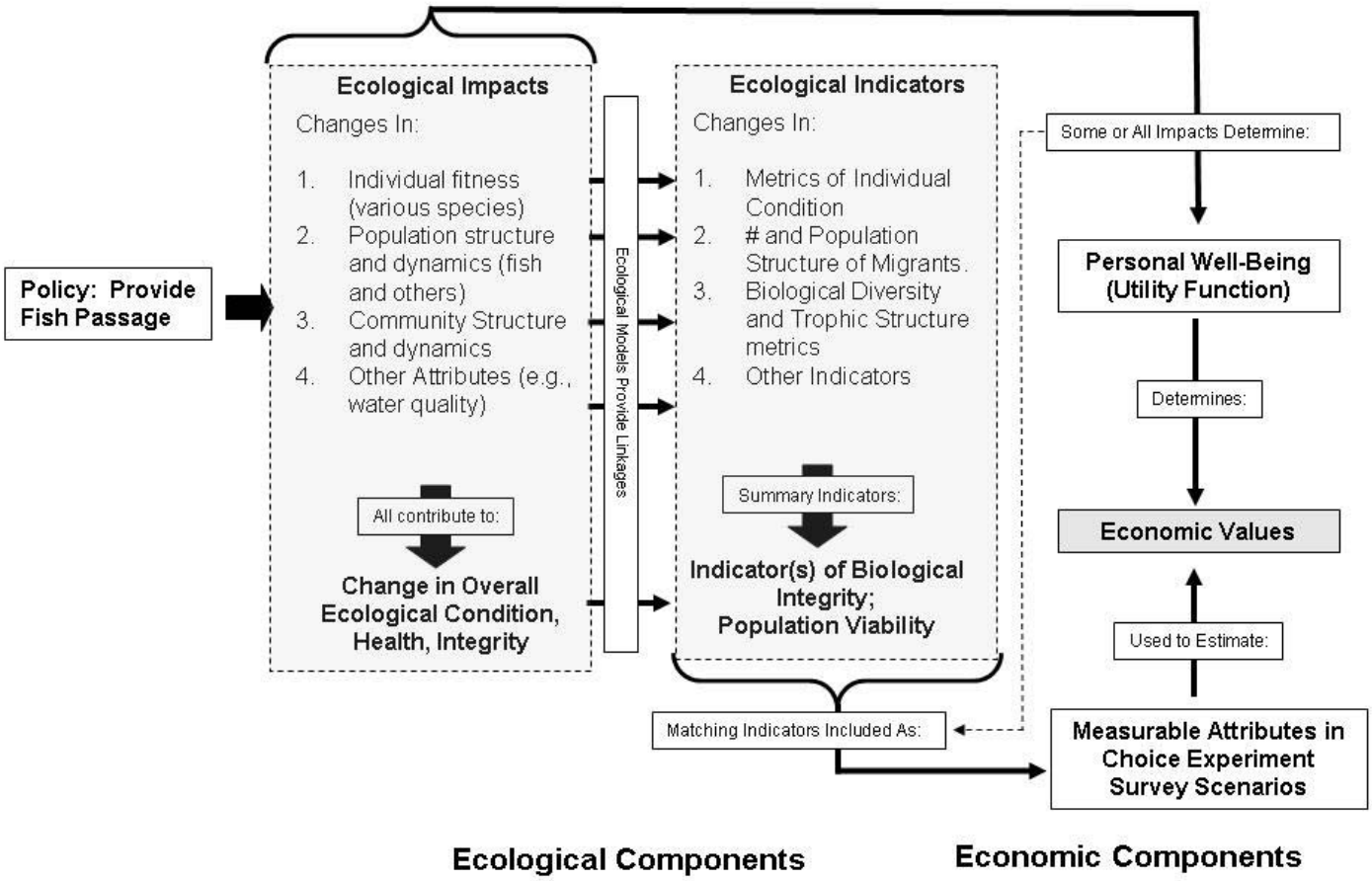


Figure. 1. Conceptual Framework of Indicator-Based Stated Preference Valuation

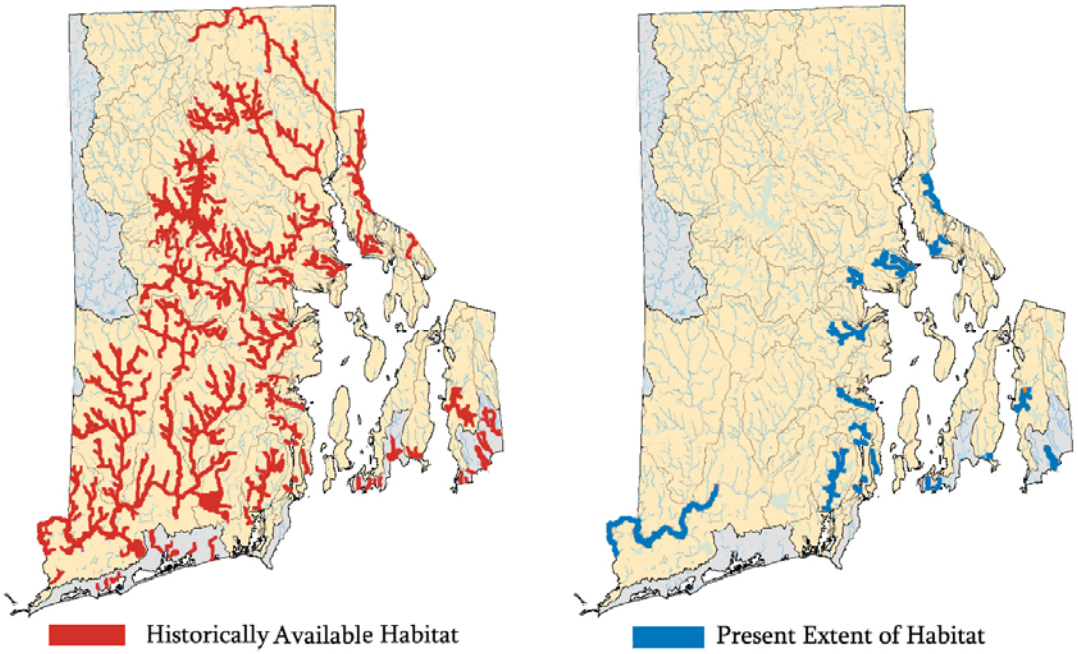
Question 3. Assume **Options A** and **B** are different Fish Passage Restoration Options for the Pawtuxet Watershed, and that the **Current Situation** is the status quo with no restoration. Unless otherwise indicated, scores show effects in restored area of the river only. **How would you vote?**

Figure 2a. Preliminary Choice Experiment Question Text

Effect of Restoration	Current Situation (no fish ladder)	Fish Ladder Option A	Fish Ladder Option B
	SCORE OUT OF 100	SCORE OUT OF 100	SCORE OUT OF 100
Fish Habitat	0 0 of 4347 watershed acres restored	3 125 of 4347 watershed acres restored	4 160 of 4347 watershed acres restored
Migratory Fish	0 0 fish out of 1.2 million possible in watershed	9 100,000 out of 1.2 million possible in watershed	10 125,000 out of 1.2 million possible in watershed
Angler Catch	50 Anglers catch the limit on 50% of trips	50 Anglers catch the limit on 50% of trips	50 Anglers catch the limit on 50% of trips
Freshwater Mussels	20 1 of 5 species native to RI are common	20 1 of 5 species native to RI are common	20 1 of 5 species native to RI are common
Fish-Dependent Wildlife	50 4 of 8 species native to RI are common	62 5 of 8 species native to RI are common	62 5 of 8 species native to RI are common
Ecological Condition	20 Ecological Health Index	25 Ecological Health Index	25 Ecological Health Index
$ Cost to your Household per Year	$0 Increase in Annual Taxes and Fees	$3 Increase in Annual Taxes and Fees	$7 Increase in Annual Taxes and Fees
HOW WOULD YOU VOTE? (CHOOSE ONE ONLY)	“ I would vote for **NO FISH LADDERS**	“ I would vote for **OPTION A**	“ I would vote for **OPTION B**

Figure 2. Preliminary Choice Experiment Question Template

DAMS AND MIGRATORY FISH IN RHODE ISLAND



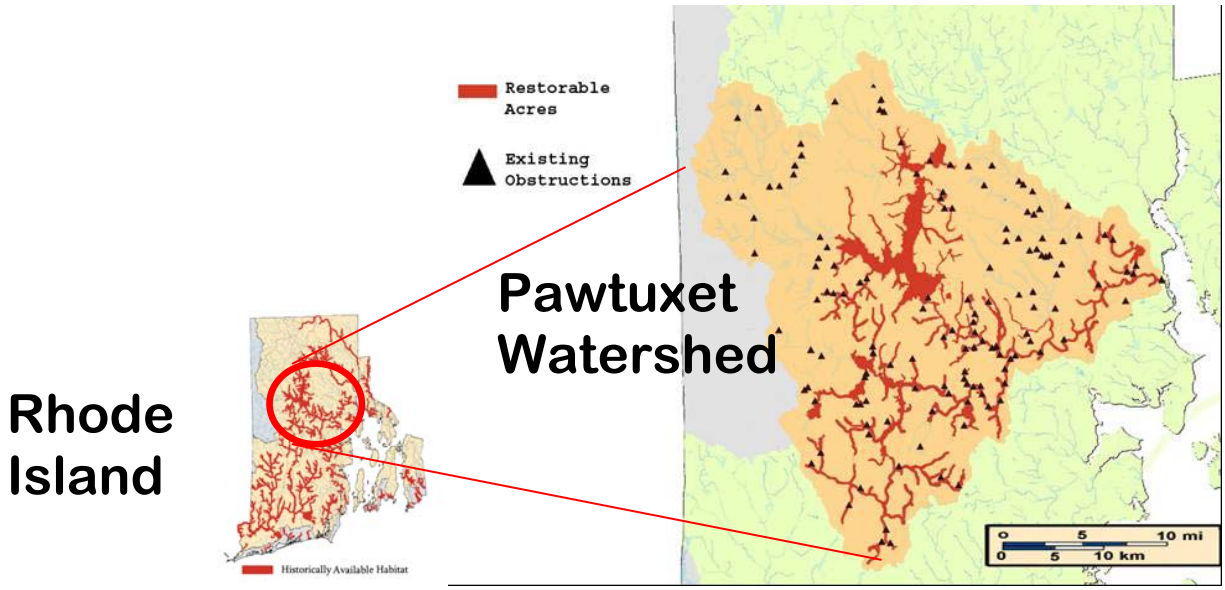
The two maps above show Rhode Island rivers where migratory fish used to live before humans arrived (left side), and rivers where these fish live today (right side).

Many Rhode Island rivers used to support annual runs in excess of 300,000 fish, now the largest measured run is only 50,000 fish. Most are much smaller.

Figure 3. Excerpt from Survey Information Booklet: Historical and Present Fish Passage (graphics edited from RIGIS data).

THE PAWTUXET WATERSHED

The Pawtuxet Watershed provides 38% of all restorable migratory fish habitat in Rhode Island, and <u>all</u> of this habitat is blocked by 22 obstructions (mostly dams). As a result, the Pawtuxet provides no upstream habitat.



If fish ladders were built over <u>all</u> 22 dams, it would provide 4347 new acres of fish habitat. Most realistic projects, however, are <u>much smaller</u>.

Figure 4. Excerpt from Survey Information Booklet: Restoration of Fish Passage in the Pawtuxet Watershed (graphics edited from RIGIS data).

Valuing environmental changes in managed fisheries

Steve Newbold & Matt Massey

U.S. EPA,

National Center for Environmental Economics

April 2007

The views expressed here are those of the authors and do not necessarily reflect those of the U.S. EPA. No official agency endorsement should be inferred.

General motivation

- In previous work, we estimated the value of water quality improvements in a recreational fishery using results from a RUM model of recreational site choices and a model of fish population dynamics (Massey *et al*. 2006).

- Additional ongoing work is attempting to formally define the potential welfare bias resulting from using reduced form models

- A quick example to illustrate …

Is integration necessary?

- If q is an exogenous environmental attribute:

$$I_i = \ln\left(e^{\theta_0} + e^{\theta_1 q - \theta_4 c_i}\right)$$

$$WTP = \frac{O}{\theta_4}\sum_i \frac{\partial I_i}{\partial q} = T\frac{\theta_1}{\theta_4}$$

→A recreation demand model alone is sufficient.

- If y is expected catch and is endogenous:

$$I_i = \ln\left(e^{\theta_0} + e^{\theta_1 y - \theta_4 c_i}\right)$$

$$WTP = \frac{O}{\theta_4}\sum_i \frac{\partial I_i}{\partial \alpha}\frac{\partial \alpha}{\partial q} = \frac{O}{\theta_4}\left[\frac{\sum_i p_i}{\beta/(\eta\theta_1) + \eta O\sum_i p_i(1-p_i)}\right]\frac{\partial \alpha}{\partial q}$$

→ An integrated model is needed.

General motivation

- Previously, however, we have ignored the management regime in the fishery.

- It is well known that the benefits generated by a renewable natural resource are strongly influenced by how the resource is managed.

General motivation

- Consider the standard model of a commercial fishery:
  - If open access, all rents are dissipated.
  - If landings (really fish mortality) is taxed at the proper level, social benefits are maximized.

- Natural resource economists have explored these issues in great detail for commercial fisheries.

General motivation

- Should we expect the same in a recreational fishery?
- If unregulated, will all benefits from water quality improvements be dissipated by new participants?
- Environmental economists have done some preliminary work on valuing different management regimes, e.g., by adding indicator variables to site choice models.

General motivation

- Reduced form recreation demand models on their own may not be sufficient for comparing the welfare effects of different fishery management strategies or environmental improvements.

- Ecological modeling on its own may not be sufficient for comparing the biological impacts.

- In many cases these processes need to be modeled jointly.

General motivation

- In this project we address both sides of this issue—management and valuation—using an integrated model of angler behavior and fish population dynamics, including fish growth.

Harvest regulations

- A variety of approaches are available to recreational fishery managers for restricting harvests:
  - Bag limits, size limits, trip fees, landing taxes, season licenses, season closures, ITQ's, …

- The relative performance of these options will depend on the anglers' preferences, the biological characteristics of the fish stock, and implementation costs.

Harvest regulations

- In general, the welfare effects of each management option will depend on the relative values of each trip attribute—total catch, take home catch, and average length of catch—and how each trip attribute is changed by the management option.

- In equilibrium, these trip attributes will depend on both the anglers' preference parameters and the reproduction, growth, and survival parameters of the fish stock.

Valuing water quality changes

- The value of a change in water quality also will depend on the nature of the management regime in the fishery.
- For example, suppose a water quality improvement causes the average catch rate to increase from 1 to 2 fish per trip.
- What is the value of this improvement to the anglers if there is a 1 fish bag limit in place in the fishery?
- What if there were a minimum size limit in the fishery instead of a bag limit?

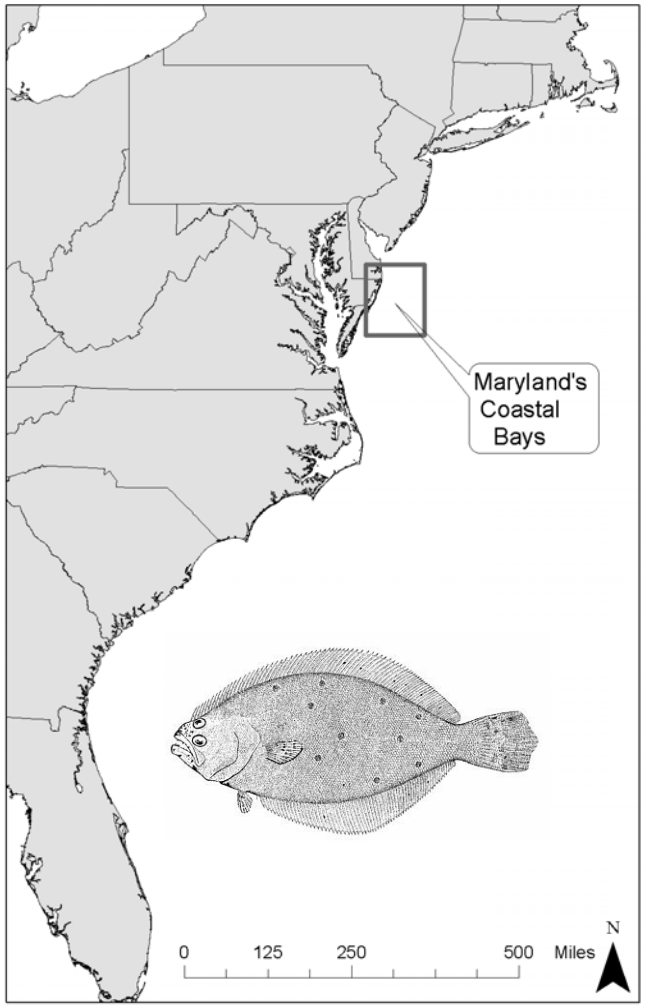
Questions to be addressed

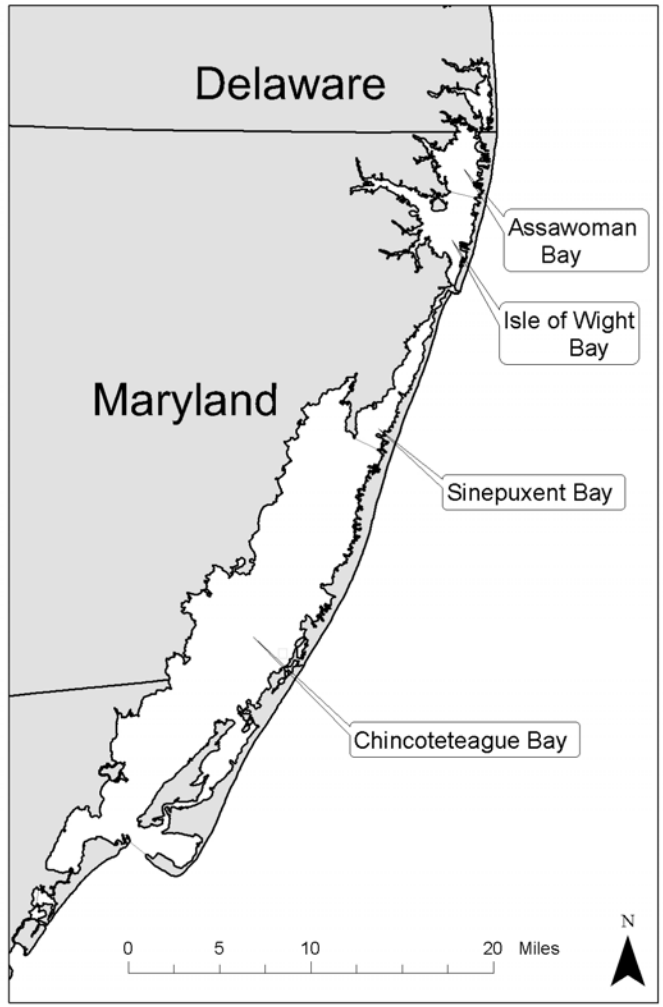
1. What is the optimal management regime for a particular recreational fishery (i.e., for a given set of angler preferences and biological parameters)?

2. How does the nature of the management regime—whether optimal or not—affect anglers' valuation of water quality changes?

Previous work

Summer flounder fishing in Maryland's coastal bays





Framework: Angler preferences and behavior

Indirect utility of fishing or doing something else:

$$V_i = \theta_1 y + \theta_2 y_h + \theta_3 l + \theta_4 \left(m_i / O - c_i \right)$$
$$V_{0i} = \theta_0 + \theta_4 \left(m_i / O \right)$$

Probability of taking a fishing trip:

$$p_i = \frac{e^{V_i}}{e^{V_{0i}} + e^{V_i}}$$

Expected max utility per choice occasion:

$$u_i^{\max} = \ln\left(e^{V_{0i}} + e^{V_i} \right)$$

Total number of fishing trips:

$$T = O \times \sum_i p_i$$

Framework:
Fish growth and population dynamics

Age structure (constant mortality):

$$n(a) = \begin{cases} n_R e^{-m(a-a_R)} & a_R \le a < \hat{a} \\ n_R e^{-m(\hat{a}-a_R)-(m+f)(a-\hat{a})} & \hat{a} \le a < \infty \end{cases}$$

Growth (von Bertalanffy):

$$l(a) = l_\infty - (l_\infty - l_R) e^{-\mu(a-a_R)}$$

Reproduction (logistic):

$$n_R = \int_{a_R}^{\infty} r(a) n(a)\, da = \int_{a_R}^{\infty} (\alpha - \beta N) n(a)\, da = \alpha N - \beta N^2$$

Equilibrium stock size:

$$N = \frac{1}{\beta}\left[\alpha - \frac{m}{1-\left(f/(m+f)\right)e^{-m(\hat{a}-a_R)}}\right]$$

Framework: Bio-economic linkages

Expected catch:

$$y = \eta N$$

Expected take-home catch (catch is a Poisson process):

$$y_h = \sum_{y=0}^{\hat{y}} y e^{-\eta N_h} \left(\eta N_h\right)^y / y! + \hat{y}\left(1 - \sum_{y=0}^{\hat{y}} e^{-\eta N_h} \left(\eta N_h\right)^y / y!\right)$$

Average length:

$$len = \frac{n_R}{N}\left[\frac{l_\infty e^{-m(\hat{a}-a_R)}}{m+f} - \frac{\left(l_\infty - l_R\right)e^{-(\mu+m)(\hat{a}-a_R)}}{\mu+m+f}\right]$$

Total harvest:

$$H = fN_h = y_h T$$

Recreation Demand Model:

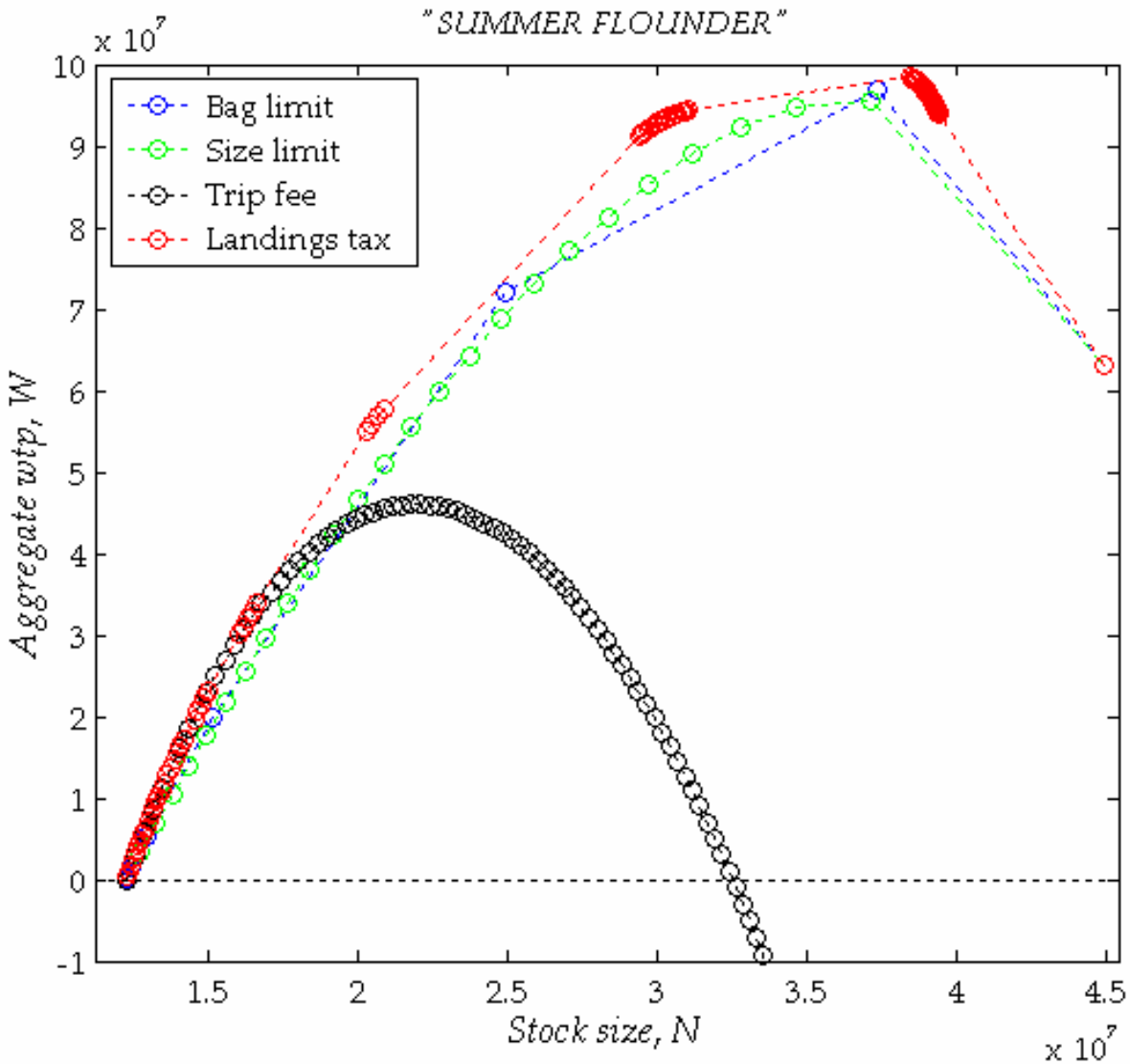
Variable	Estimate	t-stat
No Trip Dummy (NTD)	1.2616	7.555
Participant characteristics:		
NTD*Boat Owner	-0.0143	-0.122
NTD*Non-White	0.3496	1.564
NTD*Male	-0.3728	-1.402
NTD*Attended College	0.1616	1.346
NTD*Work Fulltime	-0.4701	-4.040
NTD*Daysfished	-0.0080	-1.492
Trip characteristics:		
Travel Cost	-0.0132	-13.419
Expected Total Catch	0.6183	18.216
Bag Limit	0.4770	18.322
Size Limit	0.2685	24.645
Other fishing Good	0.3148	7.416
Other fishing Bad	-0.4374	-10.913
Number of Respondents	2392	
Number of Observations	9568	
Mean Log-likelihood	-0.829243	

Example 1: summer flounder

- While there is certainly skill in hooking flounder, they are not a “fighting” species.
- Bigger is better, but fishermen do not normally target trophy flounder.
- Flounder fishermen are generally most concerned with take home catch.

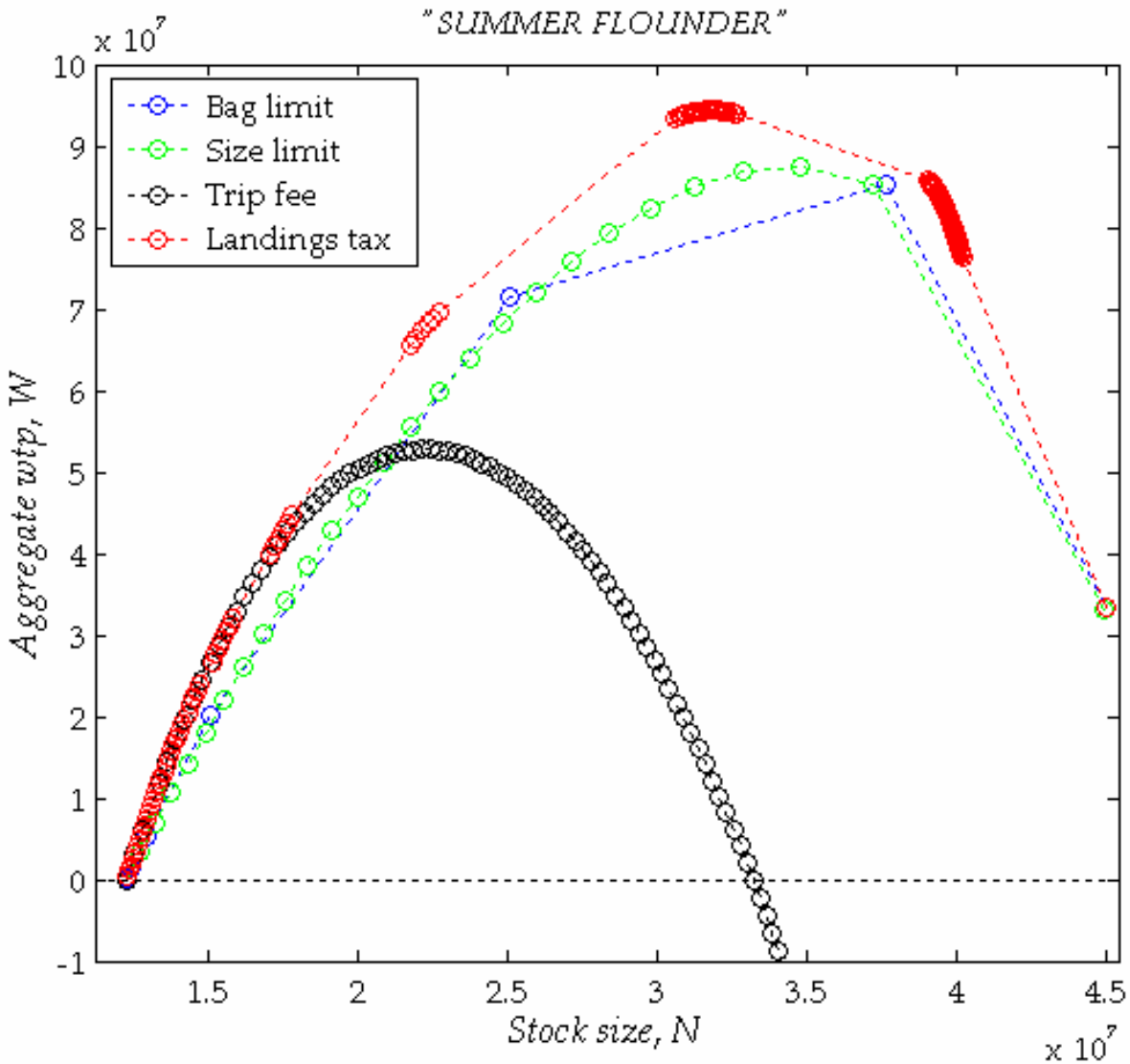
Expected take home catch

= 0.4770*.75



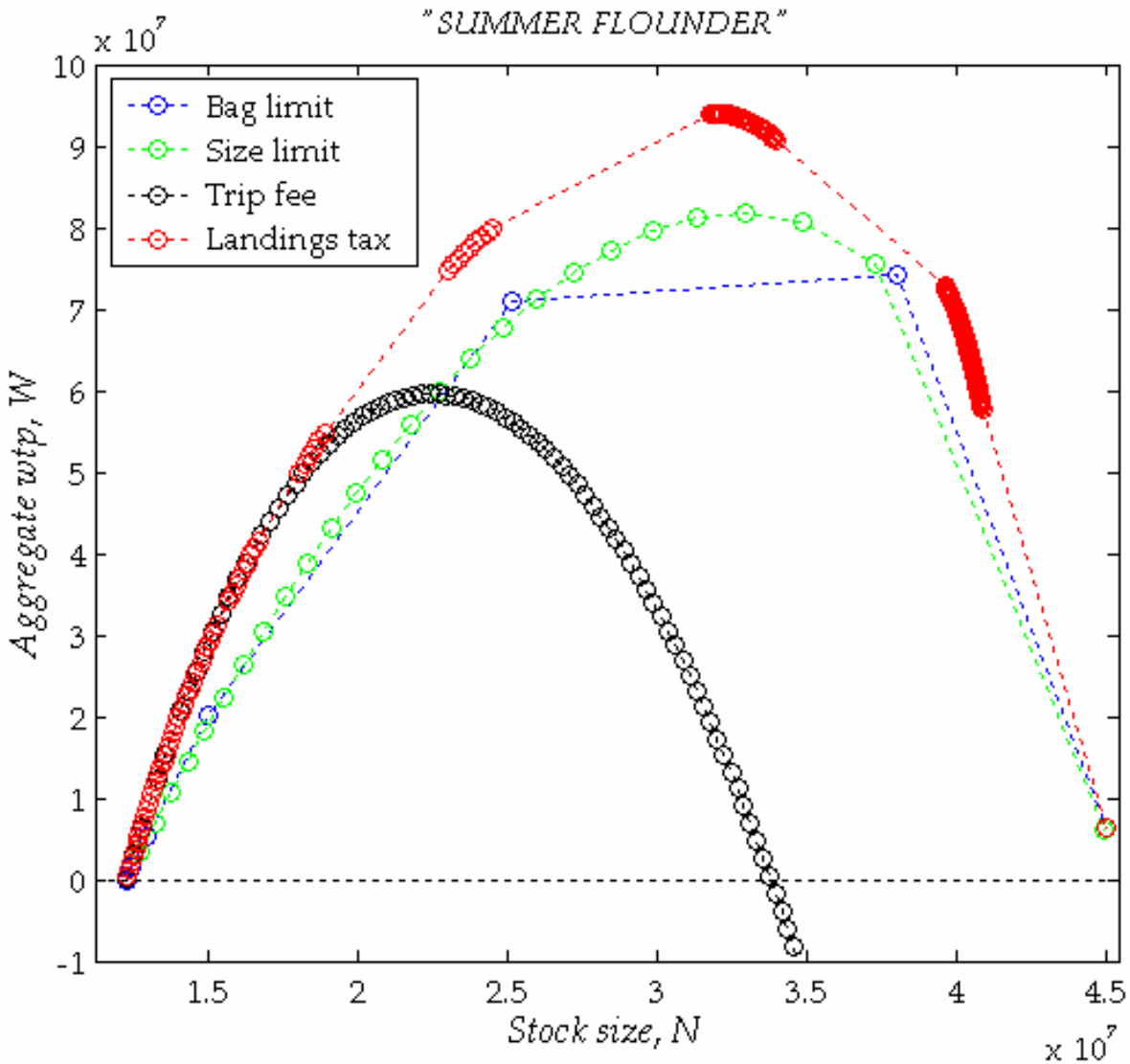
Expected take home catch

= 0.4470 (Baseline)



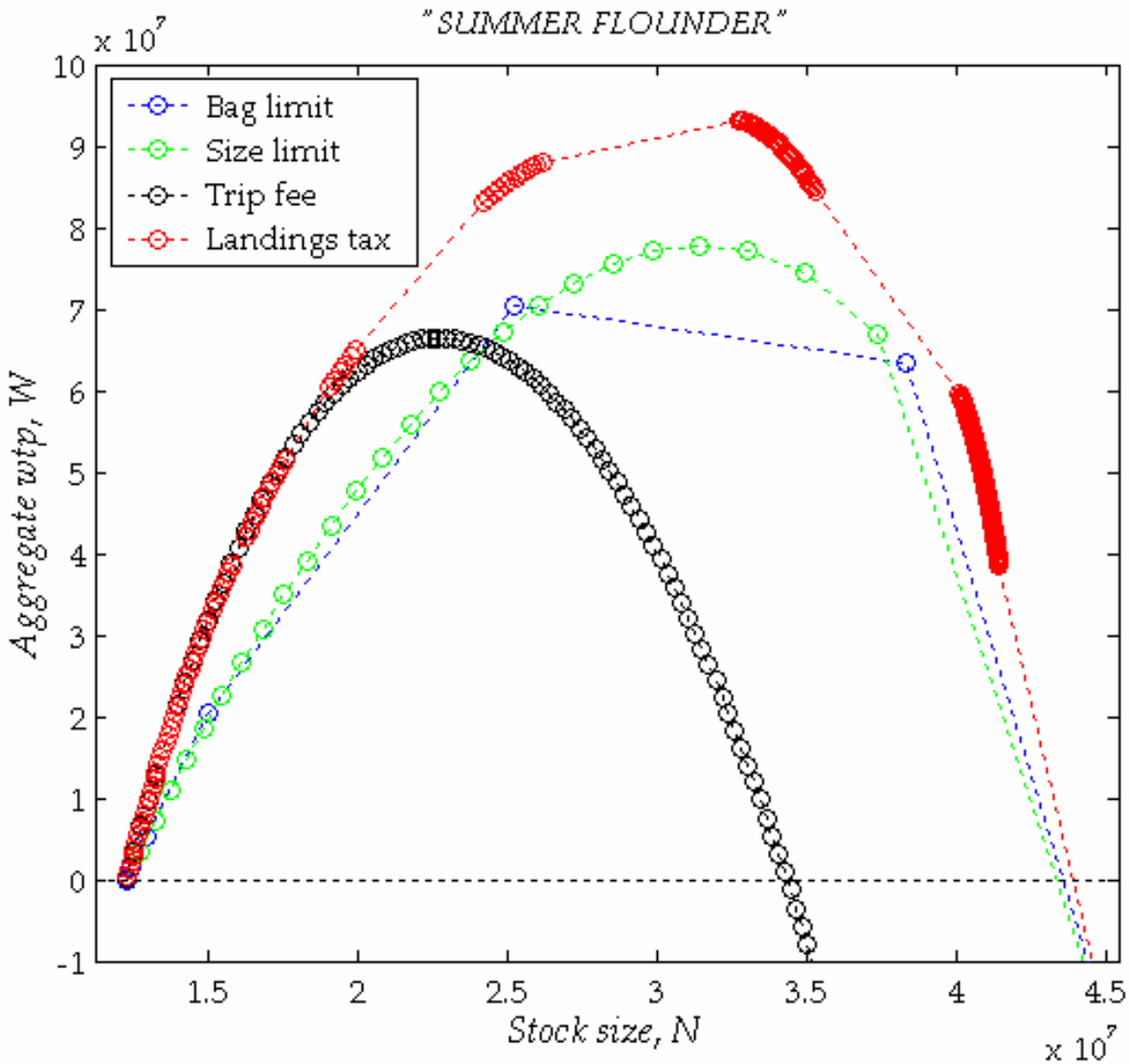
Expected take home catch

= 0.4770*1.25



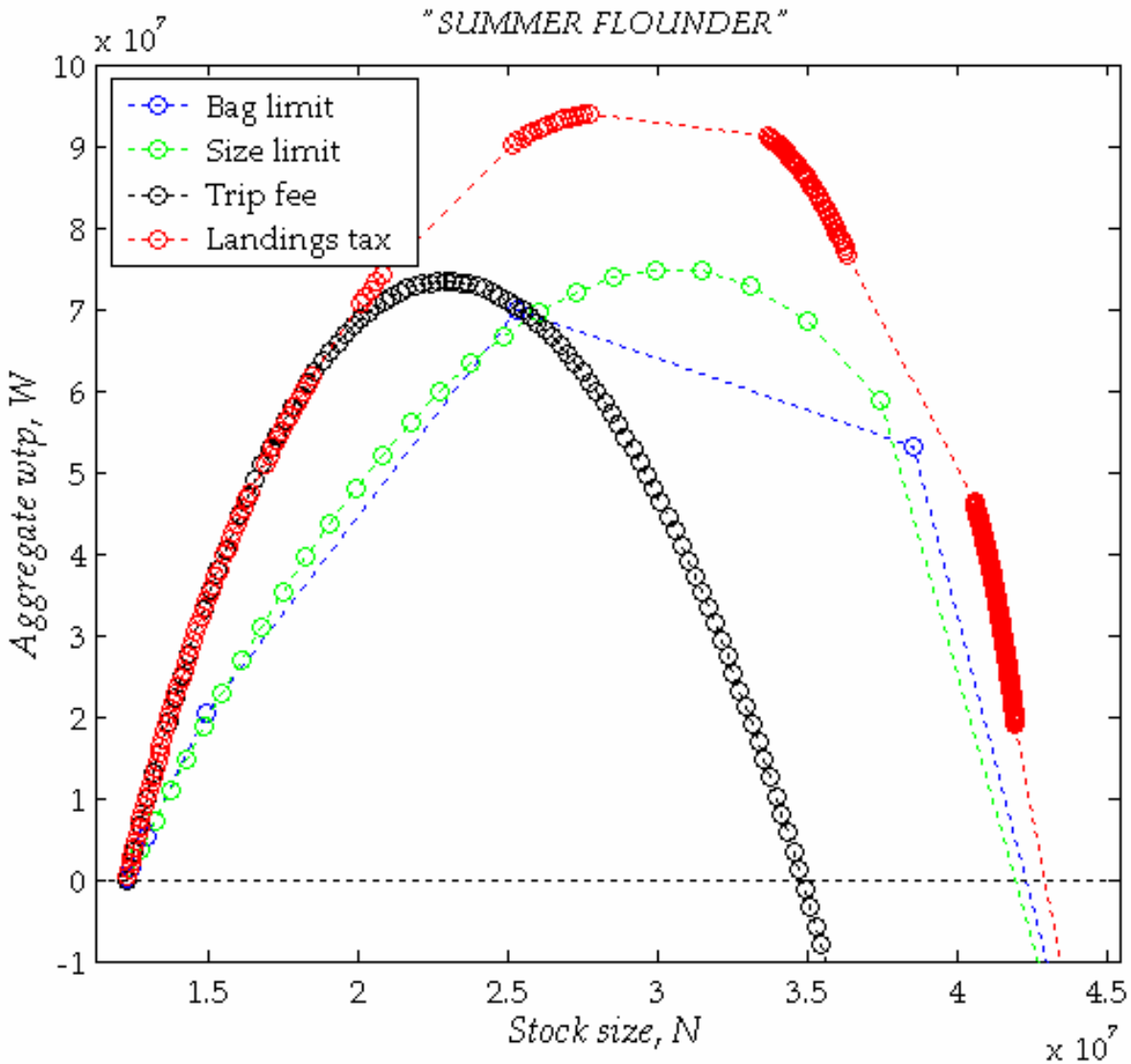
Expected take home catch

= 0.4770*1.5



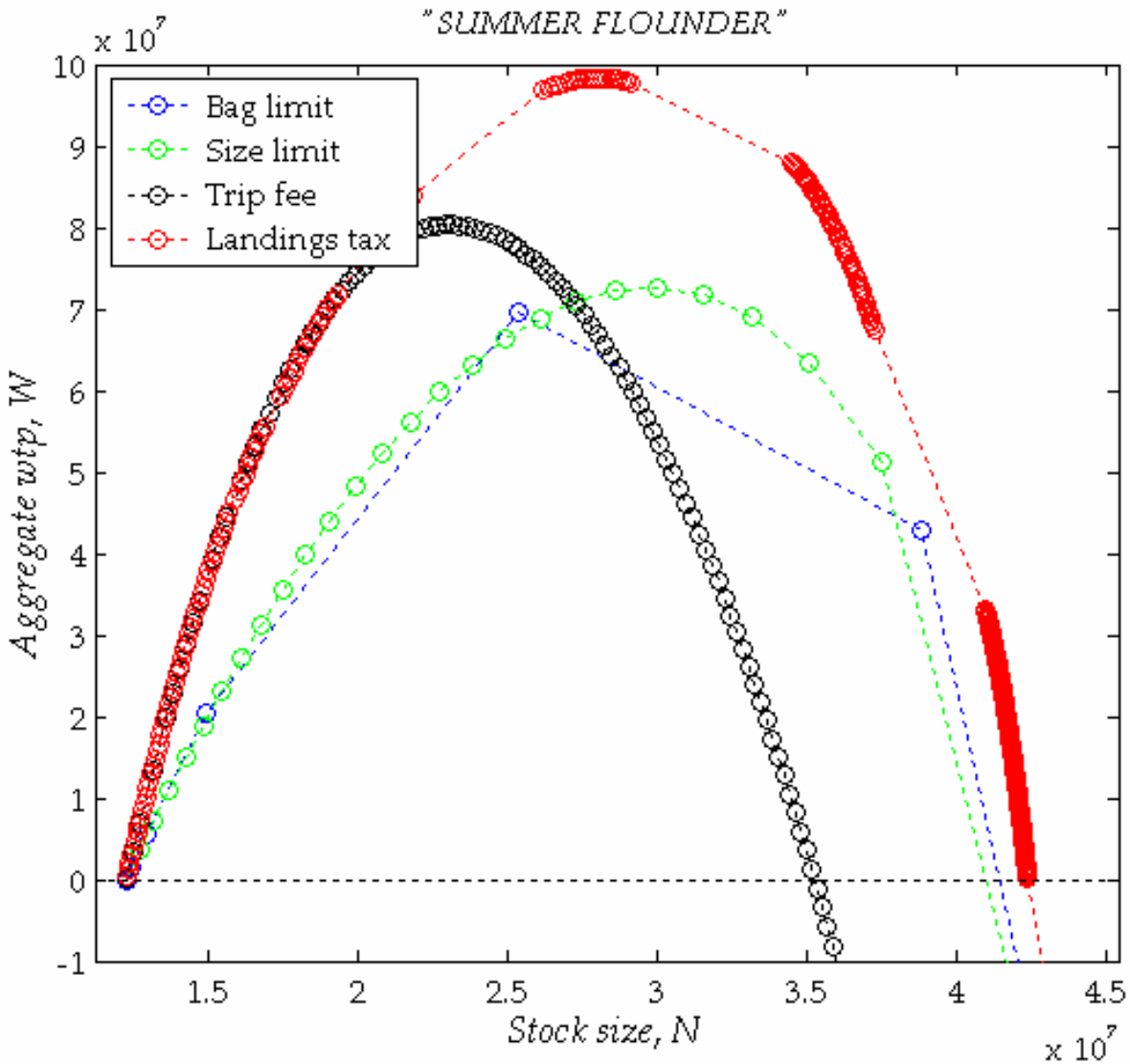
Expected take home catch

= 0.4770*1.75



Expected take home catch

= 0.4770*2



As the value of take home catch increases:

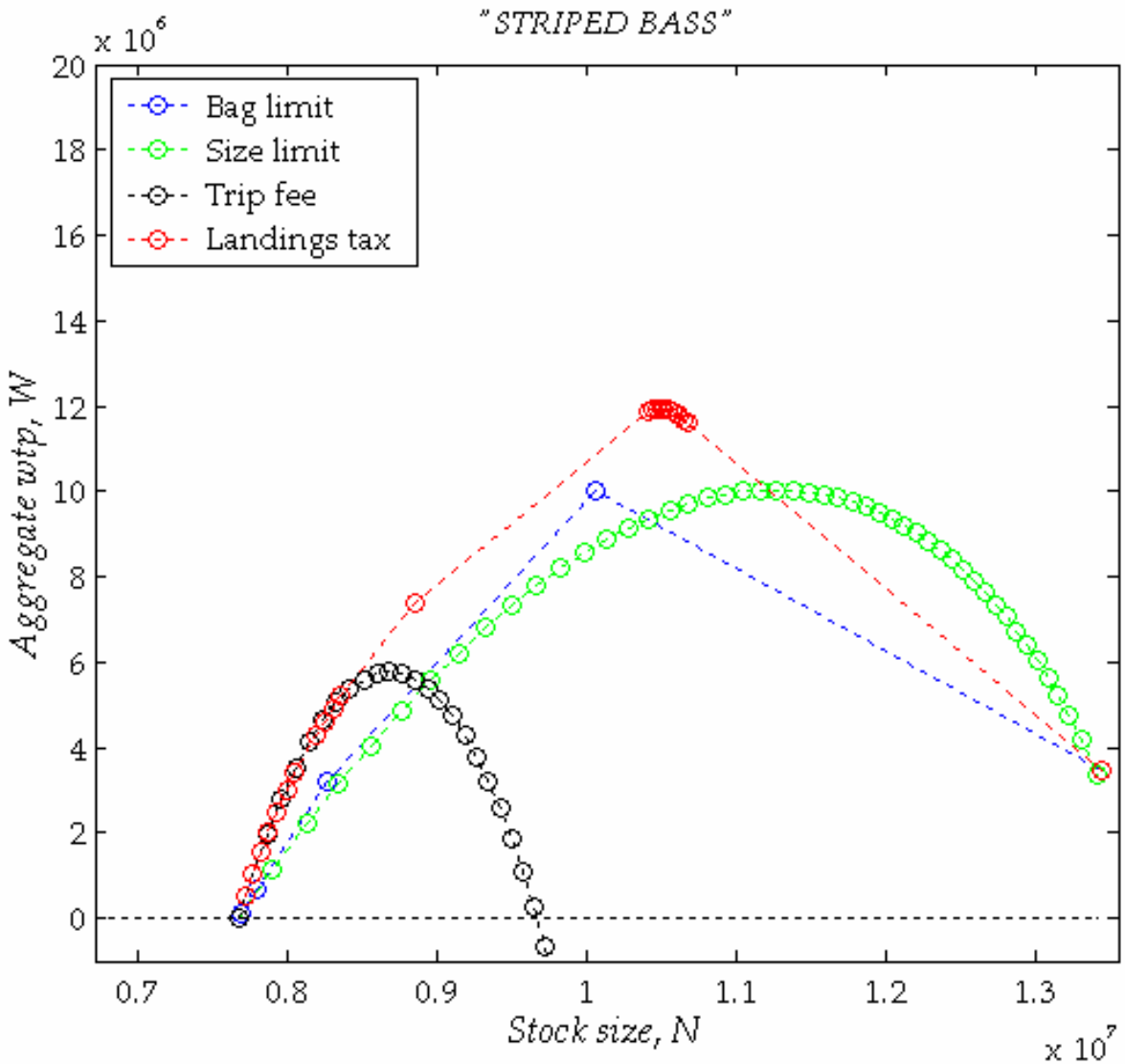
- Optimal stock size decreases
- A landings tax is always the optimal management option
- The biological methods (bag and size limits) are very similar
- Biological methods become less optimal because they restrict the number of fish you take home
- Optimal trip fee increases

Example 2: striped bass

- Striped bass are a very charismatic species and are well known for being a “fighting” fish.
- Many fishermen would value catching one large striped bass more than catching several smaller ones.
- Because we are aware of no studies that estimate angler preferences for striped bass management options, we adjust the estimated flounder parameters to increase the value for large fish and decrease the value for take home catch.

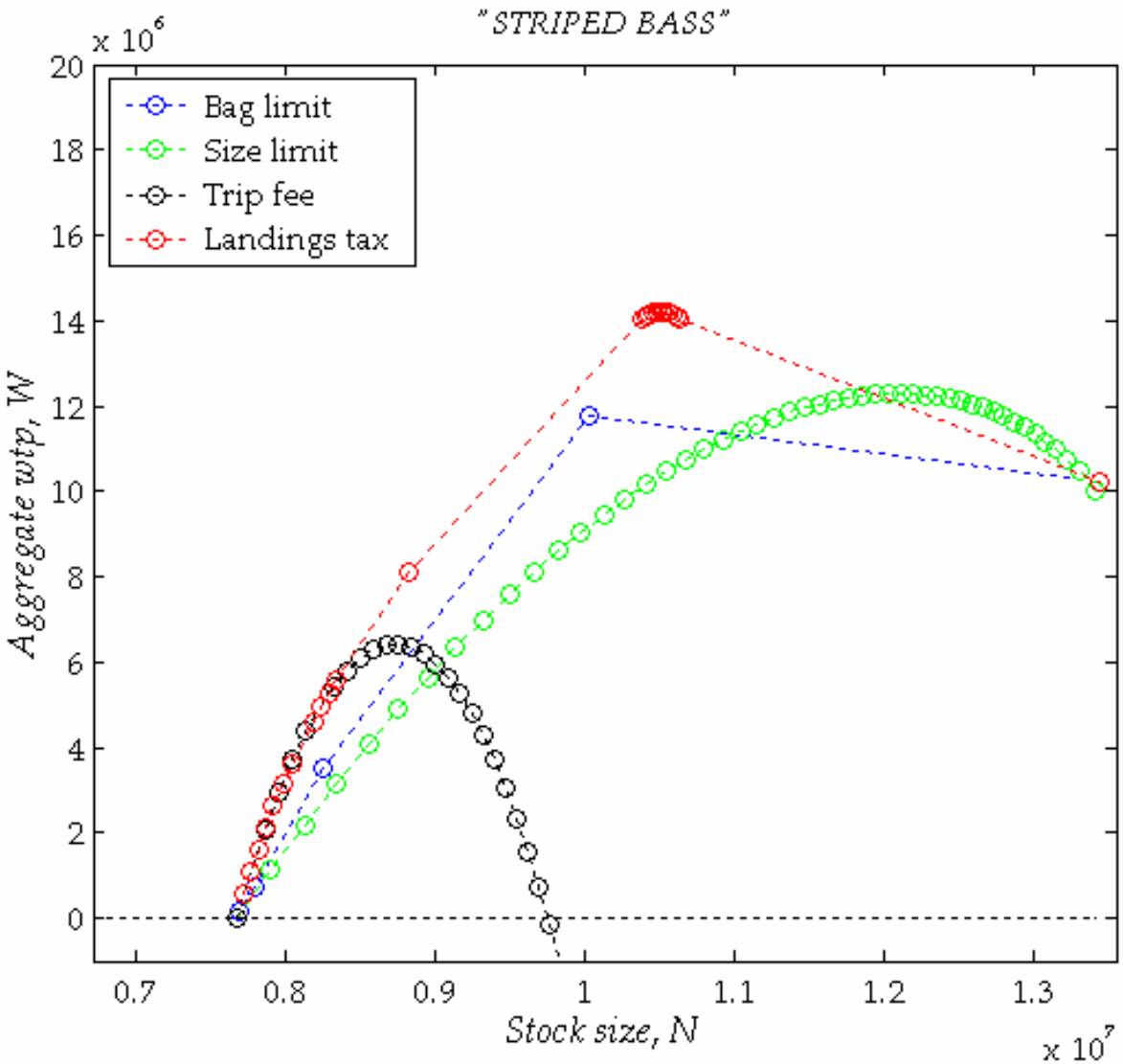
Average length of catch

= 0.2685*.5



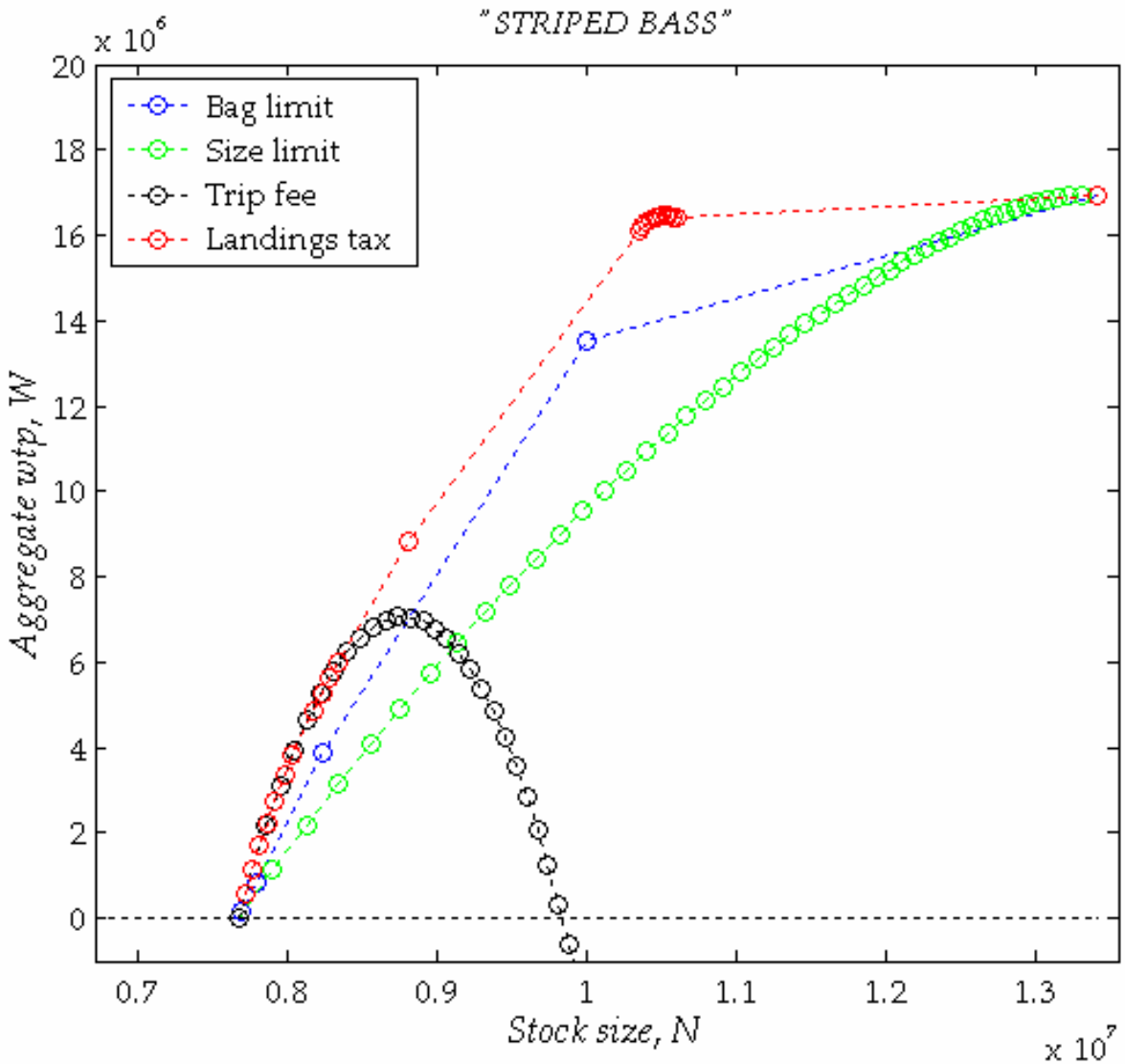
Average length of catch

= 0.2685 (Flounder Baseline)



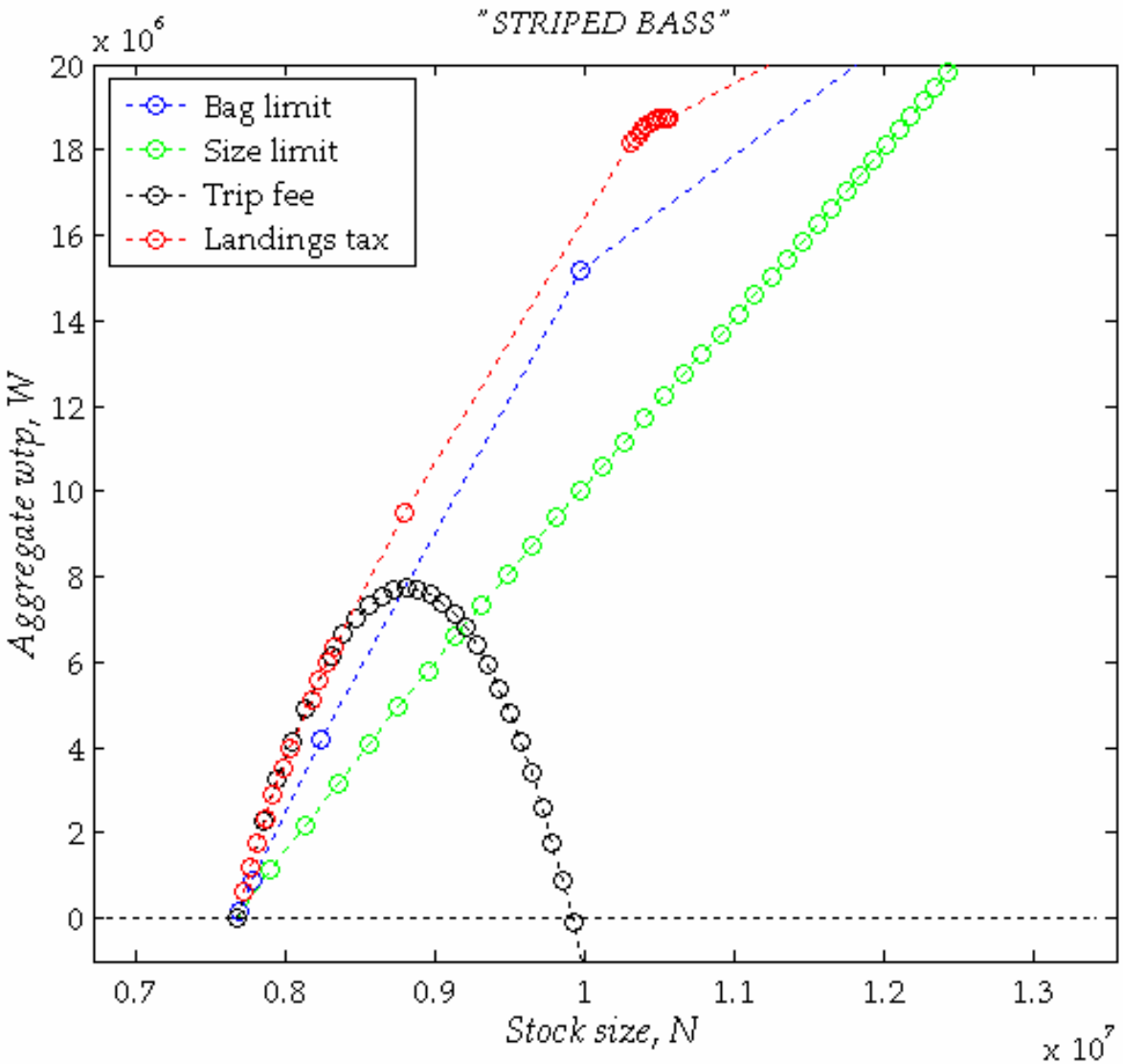
Average length of catch

= 0.2685*1.5 (striped bass baseline)



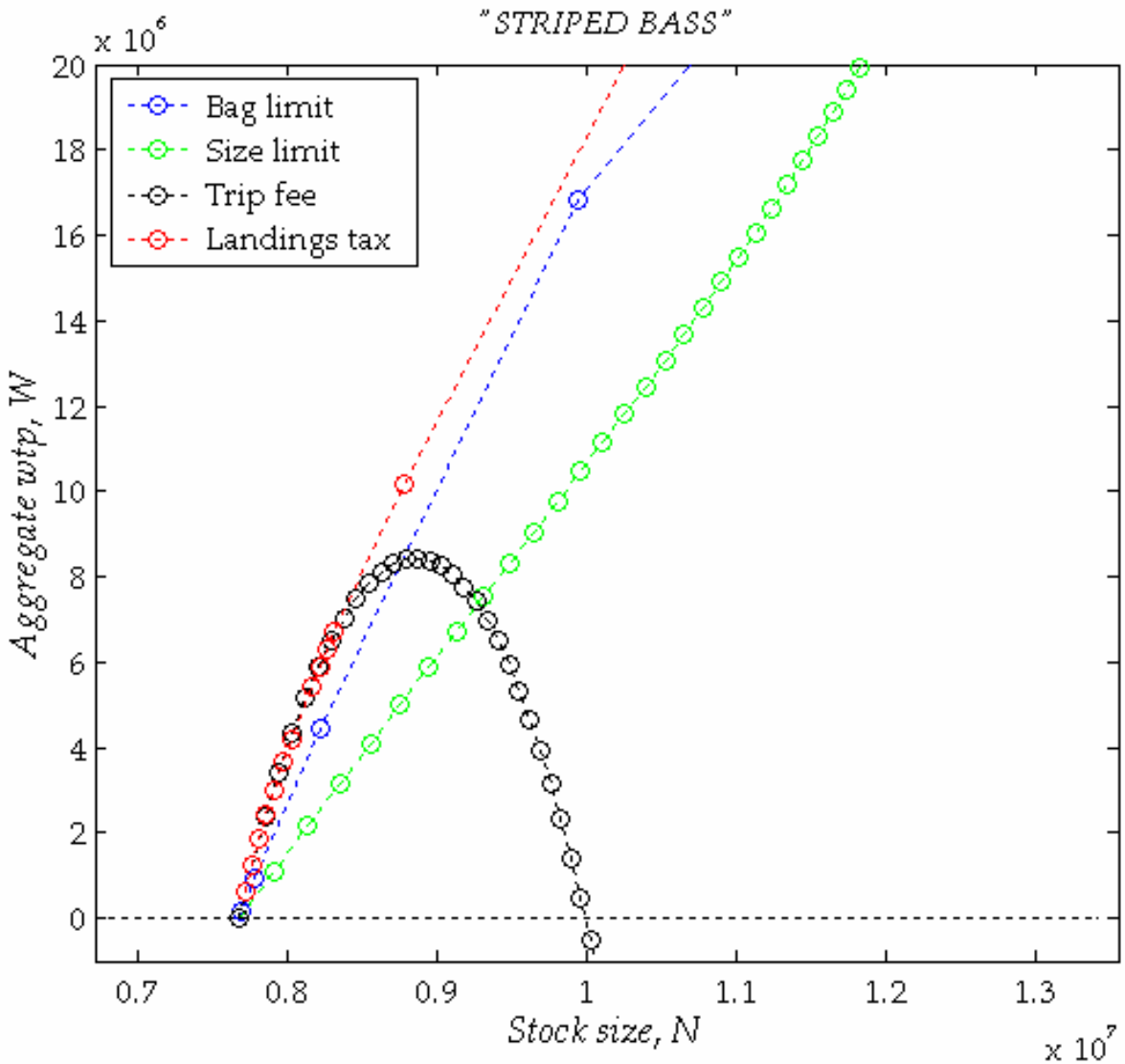
Average length of catch

= 0.2685*2



Average length of catch

= 0.2685*2.5



As the value of length of catch increases…

- The model predicts a shift to a catch and release fishery.
- A landings tax is always the optimal management option.
- Bag limits are preferred to size limits.
- Bag limits are relatively close to the optimal tax.
- Optimal trip fee increases.

Valuing environmental improvements

Management regime	Summer flounder	Striped bass
	$W(1.01\alpha) - W(\alpha)$	$W(1.01\alpha) - W(\alpha)$
Unregulated	2.57E+06	1.27E+06
Optimal bag limit	2.54E+06	8.98E+05
Optimal size limit	2.99E+06	9.18E+05
Optimal trip fee	3.09E+06	1.29E+06
Optimal landings tax	3.49E+06	8.98E+05

Summary

- Optimal management in a recreational fishery will depend on the preferences of the anglers and the biological characteristics of the fish stock.
- In some cases harvest restrictions via bag or size limits can perform nearly as well as economic incentives.
- Early results suggest that the management method can affect the magnitude of welfare estimates

Next steps

- Add discard mortality and commercial fishing to the model
- Evaluate season licenses
- What is the optimal mix of management methods?
- Apply to several managed fisheries using harvest data and calibration
- How does the adjustment path of each management techniques to the long run equilibrium differ?
- Jointly estimate biological and economic parameters

Valuation for Environmental Policy: Ecological Benefits

Water Resources Session

April 24, 2007

Julie Hewitt

Outline

- General comments
- Iowa Lakes study
- Riparian Buffer Zones study
- Indicator-based Stated Preference Valuation study
- General Conclusion
  - The Value of Non-use Value
  - Other comments

General Comments

- All papers received on schedule
- All papers well-written

- All leave me a little wanting

Iowa Lakes study

- Careful collection of environmental data
  - Allows choice of specification
- Split sample for model development, estimation and prediction
- Provides (half of) information that decision makers need

Iowa Lakes study, cont.

- Not ecological, per se
  - Focus is on recreational trips
- Only single day trips
- Sociodemographic parameters aren't random because this info doesn't vary across the sites
  - Vajjhala et al. is suggestive that it does
  - Could key to local sociodemographic info

Riparian Buffer Zones

- Motivated by paper that ignores upstream spawning but uses CEA
- Hard for me to judge whether modeling assumptions are appropriate
- Simulations show interesting differences
  - Optimal solution depends on downstream spillovers

Riparian Buffer Zones, cont.

- Based on exhaustive combinatorial search
  - Won’t work on real-world problems
- Analysts determine reach definition
- Not valuation, per se

IbSPV

- Do preferences take form $U_i(E(F))$ or $U_i(F)$?
  - Formal incorporation of focus group process
- List of 7 criteria for applying IbSPV
  - Note that existing studies don't generally meet
  - Suggest validity still possible under "certain conditions" (begs the question)
- Relationships among indicators and impacts need to be described
  - Span the range, science can follow

IbSPV, cont.

- No results, per se
- Concern about cognitive burden
- Can we learn anything about order of presenting impacts/indicators?
- Is this really new?
  - Does not mean it's not a useful articulation
- Promontory Point
  - Ideal of analysis vs. real-world

General Conclusion

- Session is appropriately titled
- Is the workshop?
  - STAR grant results and progress
  - Ecological benefits

The Value of Non-use Value

- We have to ask the question, “have we incorporated non-use values?”
- Do we have to report non-use values to answer that question?
  - Recreational use is easy to measure
  - Ecological use is not so easily measured

Other Thoughts

- Incentive compatibility of choice experiments
- Multi-purpose or multi-day trips

Comments on Session V: Water Resources
George Parsons

1. Valuing WQ (Kevin Egan)

- Omitted Variable Bias?
  - WQ Correlated w/ Missing Amenities
- Why not estimate dispersion terms for WQ?
- Recreation Types
- Repeat this Analysis w/ same data
- Small dispersion terms on no-trip constant
- Dropping heavy users?
- Policy Implications

2. Indicator Based SP (Rob Johnston)

- Are ‘full information’ values correct for policy?
  - Pathways
  - Value of Information

3. Riparian Buffer Zones (Heidi Albers)

- When will the model be ready for prime time?