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Key Points:
- Demand was modeled for recreational fishing to rivers and streams
- Fishing site qualities were measured by biomass of five game fish species
- Fishing sites were defined by hydrological boundaries and stream type

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Valuing recreational fishing quality at rivers and streams

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Abstract This paper describes an economic model that links the demand for recreational stream fishing to fish biomass. Useful measures of fishing quality are often difficult to obtain. In the past, economists have linked the demand for fishing sites to species presence-absence indicators or average self-reported catch rates. The demand model presented here takes advantage of a unique data set of statewide biomass estimates for several popular game fish species in Michigan, including trout, bass and walleye. These data are combined with fishing trip information from a 2008–2010 survey of Michigan anglers in order to estimate a demand model. Fishing sites are defined by hydrologic unit boundaries and information on fish assemblages so that each site corresponds to the area of a small subwatershed, about 100–200 square miles in size. The random utility model choice set includes nearly all fishable streams in the state. The results indicate a significant relationship between the site choice behavior of anglers and the biomass of certain species. Anglers are more likely to visit streams in watersheds high in fish abundance, particularly for brook trout and walleye. The paper includes estimates of the economic value of several quality change and site loss scenarios.

1. Introduction

Fishing at rivers and streams is a major recreational activity in the United States, with nearly 12 million participants in 2011 [*U.S. Fish and Wildlife Service (USFWS)*, 2012]. Rivers also support swimming, paddling and boating activities, provide ecosystem services such as spawning habitat for marine fishes, and are a source of substantial nonuse value [*Sanders et al.*, 1990; *Loomis*, 2003; *Debnath et al.*, 2014]. However, rivers and streams are susceptible to landscape and climate change, and the value of these resources is frequently impaired by human activity [*Allan*, 2004; *Suplee et al.*, 2012; *Ficklin et al.*, 2013]. A comparison of water quality indicators in the United States over the past decade indicates a significant decline in stream condition, predominantly in the Midwest and Plains regions [*U.S. Environmental Protection Agency (USEPA)*, 2013]. Stream anglers in particular will be sensitive to these changes, which directly affect valuable stream characteristics such as the biomass of game fish species.

The economic effects of watershed changes on stream anglers and other users can be measured with nonmarket valuation techniques. The random utility maximization (RUM) model is now a common means of estimating values for the recreational use of natural resources. In a recreational angling context, the RUM model explains the choice of fishing trip to a site among a set of many possible alternatives. By describing choice as a function of site characteristics, a RUM model is capable of predicting the monetary benefits or damages that will arise from changes in the environmental quality of sites [*Haab and McConnell*, 2003]. Below, we describe a RUM model of recreational fishing that can be used to value detailed changes in fish abundance and stream quality.

Identifying the influence of fishing quality on site choice can be challenging. Data on appropriate measures (e.g., fish abundance, catch and harvest) are often not available or are difficult to obtain for most sites. Several prior studies of stream fishing have addressed this problem by using proxies for fishing quality [*Jones and Lupi*, 2000] and presence-absence indicators [*Hunt et al.*, 2007]. Many others have elected to use anglers' self-reported catch rates, averaged by site (Table 1). These methods are less than ideal: proxies provide few insights into fishing quality, presence-absence indicators only capture discrete changes and some types of catch rate measures are prone to measurement error and estimation bias in the demand model

Table 1. Stream Angling RUM Model Studies

Authors and Year	Study Area	Site Definition	Selected Site Quality Variables
Hunt et al. [2007]	Ontario lakes and rivers	Known access points	Species-specific presence-absence indicator, walleye and trout catch rates (from observed trips)
Ji et al. [2014]	Iowa rivers	River segments	Fish presence index, water quality index, land use measures
Jakus et al. [1998]	Tennessee reservoirs	Reservoirs	Total catch rate (from observed trips) fish advisory indicator
Jones and Lupi [2000]	Michigan lakes and rivers	Counties	Species-specific catch rates at Great Lakes (from creel data), stream type indicators, landscape characteristics
Lin et al. [1996]	Willamette River basin	Four river segments	Fishing quality index, congestion
MacNair and Cox [2000]	Montana lakes and rivers	River segments and lakes	Total species biomass, restricted species, site size
Morey et al. [1993]	North Atlantic salmon rivers	Maine rivers and Canadian provinces	Total catch rate (from observed trips)
Morey and Waldman [1998]	Montana rivers	River segments	Total catch rate (from observed trips)
Morey et al. [2002]	Clark Fork River basin	River segments	Total catch rate (from observed trips), site size
Murdock [2006]	Wisconsin lakes and rivers	Rivers grouped by quadrangles and lakes	Species-specific catch rates (from observed trips), boating facilities, landscape characteristics
Parsons and Hauber [1998]	Maine lakes and rivers	River segments and lakes	Salmon presence-absence indicator, water toxicity
Peters et al. [1995]	Alberta lakes and rivers	River segments and lakes	Total and trout-specific catch rates (from observed trips), water quality index, site size
Phaneuf [2002]	North Carolina lakes and rivers	Subbasin watersheds	Phosphorous, dissolved oxygen, ammonia, acidity indexes
Train [1998]	Montana rivers	River segments	Total species biomass, restrictedspecies, site size
Von Haefen [2003]	Susquehanna River basin	Sub-subbasin watersheds	Trophic state index, dissolved oxygen index

[*Morey and Waldman*, 1998; *Train and McFadden*, 2000]. Furthermore, many of the catch rate measures employed in the literature are not designed to distinguish between fish species, although there is evidence that the impact of fishing quality on site choice is species-specific [*Peters et al.*, 1995; *Murdock*, 2006].

Another challenge in modeling the demand for stream fishing is determining what constitutes a fishing site. There is not yet a consensus in the literature on the site definition for stream fishing (Table 1), although it is generally recognized that large individual sites tend to be heterogeneous in site quality, suggesting that using small sites will result in a better model [*Lupi and Feather*, 1998]. Indeed, there does appear to be a trend toward more refined site definitions. For example, *Morey et al.* [1993] used rivers, *Parsons and Hauber* [1998] used river segments and *Hunt et al.* [2007] used river access points as sites. Several papers have also used hydrological boundaries to assist in defining sites [*Phaneuf*, 2002; *Von Haefen*, 2003].

This paper presents a site choice model of stream fishing using species-specific biomasses as measures of fishing quality. The biomass data come from biological stream surveys, i.e., a form of fisheries-independent data, which are generally preferred to self-reported angler catch rates, a form of fisheries-dependent data which can vary based on angler skill and gear [*Maunder and Punt*, 2004]. Fishery-independent biomass estimates are well suited to capturing relative differences in abundance across freshwater streams [*Hayes et al.*, 2007]. To date, biomass measures are rarely employed in models of stream fishing (*Train* [1998] is an exception), even though catch rates directly relate to biomass [*Clark*, 1990]. Our data are also unique in that they include several different species-specific measures of biomass rather than a single composite measure.

Valuation of recreational fishing is a key component in the science of river restoration. By including species-specific biomass, the site choice model can be used to value detailed and diverse changes in fishing quality—e.g., abundance increases for some species but decreases for others, as might be expected under a climate change scenario, under management changes that alter hydrology, or as a consequence of ecosystem restoration [*Meyer et al.*, 1999; *Bond and Lake*, 2003; *Palmer and Bernhardt*, 2006]. Communicating the role that restored ecosystem services have on individual and social benefits can have a significant impact on ecosystem management decisions, especially when there is conflict over which services a river system or watershed should support [*Wohl et al.*, 2005]. Valuation is especially useful if benefits can be measurably related to riparian landscape and habitat conditions that drive fishing quality, which is a major motivation for the fish biomass data used in the angler model below.

10.1002/2014WR016152

Table 2. Fishing Trip Characteristics

Characteristic	Mean
Restricted license[a]	0.397
Fished in spring[b]	0.292
Fished in summer	0.321
Fished in fall	0.341
Targeted trout[c]	0.395
Targeted bass	0.314
Targeted panfishes	0.272
Targeted walleye	0.176
Targeted other fishes	0.267
Did not target particular species	0.179

[a]Anglers have about a dozen different fishing license options in Michigan but there are two basic types: restricted licenses and all-species licenses. Restricted licenses permit fishing for all species except trout, salmon, lake sturgeon, lake herring, amphibians, reptiles and crustaceans. Typical sales consist of about 60% restricted and 40% all-species licenses.

[b]Spring: March–May. Summer: June–August. Fall: September–November. Approximately 4% of sample trips were taken in an unspecified month.

[c]A trip could have targeted more than one species group.

Our model makes several further contributions to the literature. Hydrological boundaries are used to construct the choice set in which fishing sites are classified at the subwatershed level. Additionally, many of the largest subwatersheds are broken down into two sites using information on site characteristics that relate to fish assemblages. This advances the trend in the literature to further refine fishing site definitions. To account for the role of latent fishing site characteristics, the variant of the model presented here includes site fixed effects (sometimes referred to as alternative specific constants). The model is applied to stream fishing in Michigan and the results are used to estimate the economic benefits of several hypothetical improvements in fishing quality.

2. Methods

2.1. Fishing Trip Data

We use data from the Michigan Recreational Angler Survey (MRAS), a mail survey that has been administered monthly to a random sample of Michigan fishing license holders since July 2008. The survey questionnaire inquires about the two most recent fishing trips and the household characteristics of anglers. The response rate is approximately 47%. Details of the MRAS survey instrument can be found in *Simoes* [2009]. The questionnaire includes the usual questions about demographics and economic status, including household income. Data from the MRAS available for our analysis include the responses from 2008 through the 2010 survey period. We focus on the subsample of day trips that respondents reported were for the purpose of fishing a river or stream and were within 200 miles of an angler's home. We dropped trips taken in December–February because these would have visited a distinct subgroup of sites, e.g., frozen impoundments. These refinements yielded a total of 2064 trips taken by 1591 anglers (some anglers reported only their most recent trip or a second trip that did not fall into the defined subsample). Relevant descriptive statistics of this sample are consistent with our expectations (Table 2), in that the most popular months for fishing are in the summer and fall. Approximately 40% of the stream trips are taken by anglers with a restricted license (which means they are not allowed to fish for trout). About 60% of the licenses sold in the state are restricted, so the data for stream fishing trips reflect the increased emphasis stream anglers place on trout.

We use hydrologic units to define the set of possible fishing destinations. A hydrologic unit defines an area of land with a common drainage outlet point (e.g., a river mouth). The U.S. Geological Survey and U.S. Department of Agriculture has divided the United States into nested hydrologic units that are classified within a six-level hierarchy, where each unit is identified by a "HUC" code consisting of two to twelve digits based on the position of a unit within the system [*U.S. Geological Survey (USGS) and U.S. Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS)*, 2012]. At the top level of classification are 2-digit HUCs representing the major national river drainage regions, such as the Great Lakes. Each region then consists of several subregions (HUC4) that nest perfectly within them, with additional 6, 8, 10, and 12-digit nested units defined at progressively finer spatial resolutions. We initially distinguished fishing destinations at the level of the 10-digit HUC, which produced a tentative choice set of 258 watershed units (Figure 1). Fishable river reaches were defined within these units so that reach-level summaries of fisheries biomass and other covariates (described below) could be summarized without accounting for unfished headwater streams. A fishable reach was defined as a stream segment in the 1:100,000-scale National Hydrography Dataset (NHD) [*US Environmental Protection Agency (USEPA) and US Geological Survey (USGS)*, 2005] with an upstream catchment area greater than or equal to 50 km^2. Ninety seven percent of reported river fishing sites in the MRAS that could be matched to a specific reach fall within this cutoff.

A further refinement of the site definition was made to reduce heterogeneity of stream types within a watershed. Distinctive fish assemblages are associated with warm water and cold water habitats in Michigan on the basis of the fisheries they support [*Wehrly et al.*, 2006; *Zorn et al.*, 2011]. The NHD stream reaches

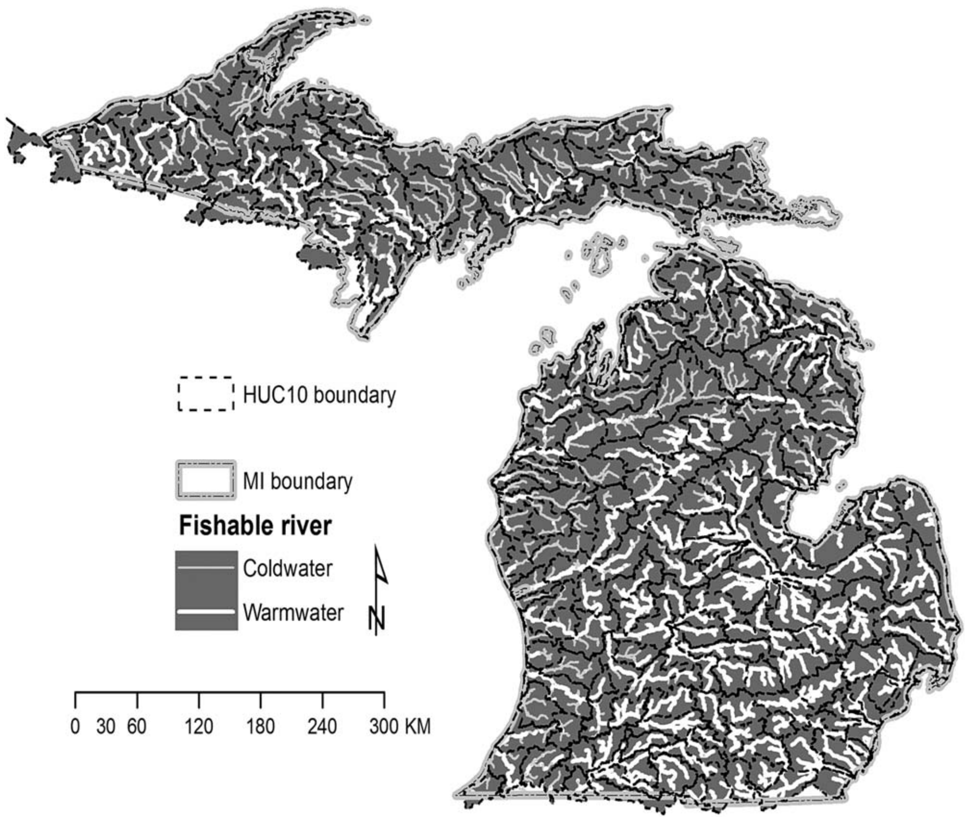

Figure 1. Fishable rivers and streams in Michigan with hydrological boundaries.

within each watershed were classified as cold water or warm water using a July mean temperature of 19.5° C as a cutoff [*Zorn et al*., 2009]. A small watershed might only contain warm water streams and would therefore consist of a single, warm water fishable alternative, while a large watershed might contain both warm water (e.g., downstream) and cold water (e.g., upstream) reaches and would consist of two alternatives in each stream class. This refinement resulted in 408 fishable alternatives—the sites in the RUM model. Trip destinations were matched to sites based on the stream name, county and/or nearest city reported by the angler.

Although our river fishing model has a very large choice set with a broad range of fishing options, it does not include fishing at other waters such as the Great Lakes and inland lakes. This decision was made to maintain a tractable model but also because we know that for certain anglers (e.g., brook trout anglers) there are no feasible alternative water body types in Michigan. Moreover, prior research with models covering a statewide scale has demonstrated that for many changes in quality of site access there is a relatively small degree of substitution between fishing in different water body types in Michigan [*Jones and Lupi*, 2000; *Kotchen et al*., 2006]. Thus, the insights from our model are likely to be accurate as long as they are interpreted in the context of the relevant population (in this case, only stream anglers) [*Jones and Lupi*, 2000; *Parsons et al*., 2000].

Travel costs were calculated from travel distances, angler characteristics and gasoline prices. Travel distances from the centroid of an angler's home zip code to the centroid of each fishable alternative were estimated using the PC*Miler program [*ALK*]. The midpoint of an angler's income category from one of six possible categories on the questionnaire or, for anglers who omitted a response, the census-reported zip code median income was used as a measure of income. We then used one third of an angler's income divided by 2000 to proxy for the opportunity cost of travel time [*Parsons*, 2003]. Per-mile driving costs were computed from Michigan monthly retail gasoline prices (per gallon) [see *US Energy Information Administration (EIA)*, 2012] divided by the average per-gallon fuel economy for light vehicles in the year of the trip, plus per-mile maintenance and depreciation costs gathered from AAA reports. For undated trips we used the 2007–2010 average gasoline price and fuel economy. This yielded an average per-mile cost of fuel,

Table 3. Fish Biomass Estimates Across Fishable Alternatives[a]

Species	Min	Median	Mean	Max	% Occupied[b]
Brook trout	0.00	0.00	0.48	6.78	39
Brown trout	0.00	0.05	2.38	32.42	54
Smallmouth bass	0.00	0.00	0.62	9.60	49
Panfishes	0.00	1.98	3.67	292.66	83
Walleye	0.00	0.00	0.09	1.59	46

[a]These refer to the untargeted biomass estimated from *Esselman et al*. [2014].
[b]This is the percentage of sites that are predicted to have a positive amount of biomass for each species.

maintenance and depreciation of \$0.40. Finally, travel costs were calculated as round-trip distance in miles times per mile fuel, maintenance and depreciation costs plus the opportunity cost of travel assuming an average driving speed of 45 miles per hour.

2.2. Fish Biomass Data

The fish biomass estimates for each HUC come from a series of models developed by *Esselman et al*. [2014]. To summarize, fish biomass for commonly targeted sport fisheries was modeled using biomass measures compiled in the Michigan Rivers Inventory (MRI) [*Seelbach and Wiley*, 1997]. The MRI data set contains biomass (kg ha^{-1}) by species measurements for 675 sites in Michigan cold and warm water rivers. Modeled fish species include brook trout, brown trout, walleye, smallmouth bass, and a combined group of panfishes that are targeted more generally with hook and line (including black crappie, white crappie, bluegill, green sunfish, hybrid sunfish, pumpkin seed sunfish, redear sunfish and rockbass). For each species or species group, a boosted regression tree model was trained and optimized on the MRI data. Predictors in the models were drawn from databases developed in the Great Lakes Aquatic Gap Analysis Program [*US Geological Survey Great Lakes Science Center (GLSC)*, 2006] and the Classification and Impairment Assessment of Upper Midwest Rivers project [*Brenden et al*., 2006; *University of Michigan (UM)*, 2006]. The regression tree models predicted fish biomass to all confluence-to-confluence river reaches in fishable rivers predicted to be occupied by each species based on *Steen et al*. [2008]. Fish biomass was then summarized to the angler choice set as the length-weighted mean value of all warm or cold water reaches in each fishable alternative (Table 3). The predictions indicate that fish biomass for a particular species or species group is characterized by little-or-no abundance at most of our sites and high abundance at some sites. Among the species, brook trout are least likely to be found at our sites, which is not surprising given their habitat requirements. On the other hand, some kind of panfish can be expected at most sites, which is consistent with the variety of fish included in this species group.

The biomass measures enter the RUM model as individual (angler)-specific variables. The MRAS database includes information on the particular species targeted, if any, by anglers during a fishing trip. Five indicators classify anglers as targeting some combination of trout, bass, panfishes, walleye or other species, while a sixth indicator accounts for anglers who did not target a particular species on a trip (Table 4). Interacting these indicators with the species biomass predictions from Esselman et al. creates targeted biomass variables. This adjustment allows us to focus on the desirability of biomass for site choice taking fish preferences as given [*Scrogin et al*., 2004]. The resulting biomass variables are used as individual-specific explanatory variables in the recreational fishing site choice model.

2.3. Site Choice Model

We use a RUM model to test and measure the importance of the site characteristics travel cost and fish biomass on stream choice. In general, recreational demand RUM models explain observed trip patterns in terms of the characteristics a trip-taker would experience at different alternatives. Each angler i has the choice to visit N_i sites, while each site $j \in 1,\ldots,N_i$ is associated with a utility level of U_{ij}. The indirect utility level measures the benefits an angler enjoys on a trip occasion to alternative j and is expressible as:

$$U_{ij}=U\left(y_i-p_{ij}, b_{ij}, q_j, \varepsilon_{ij}\right) \tag{1}$$

where y_i is the angler's income, p_{ij} is the travel cost, b_{ij} is the targeted species biomass, q_j is a vector of site-specific quality measures and ε_{ij} is the part of utility determined by factors unobserved by the researcher.

Table 4. Stream Fishing RUM Model Results[a]

Parameter	Coefficient	Clustered Standard Error
Targeted Fish Biomass		
Brook trout	0.400	0.180
Brown trout	0.132	0.057
Smallmouth bass	0.098	0.031
Panfishes	0.109	0.023
Walleye	0.364	0.084
Price Measure		
Travel cost	−0.031	0.003
Landscape		
NWSR[b]	0.563	0.236
Forest[b]	−1.540	0.491
Agriculture[b]	−2.325	0.745
Urban[b]	−3.165	1.279
Length[b]	0.181	0.073
Group Class		
Cold water[b]	−0.495	0.194
Dissimilarity, θ	0.582	0.065
Trips		2064
Rows of data		273,378

[a]All reported estimates are significant at the 5% level. The results for the site fixed effects are withheld for brevity.
[b]Identified via a regression of the site fixed effects on these variables (N=232; R^2=0.552), which included 12 basin-level fixed effects withheld for brevity.

Assuming utility is linear and additively separable in the observed and unobserved components, we can rewrite equation (1) as

$$U_{ij}=V_{ij}+\varepsilon_{ij} \text{ where } V_{ij}=\alpha(y_i-p_{ij})+\beta b_{ij}+\gamma q_j. \quad (2)$$

Trips are taken to the alternative that yields the highest utility among all possible choices, implying that site j is chosen when $U_{ij}>U_{ik}$, although the researcher only observes the portion V_{ij} and cannot predict with certainty the preferred fishing alternative for any given trip. However, by specifying a distribution for ε_{ij} the probability that the site visited is best can be formed:

$$\begin{aligned} prob_i(choose\ j)&=prob(U_{ij}>U_{ik})\ \forall\ j\neq k \\ &=prob(V_{ij}+\varepsilon_{ij}>V_{ik}+\varepsilon_{ik})\ \forall\ j\neq k \\ &=prob(V_{ij}-V_{ik}>\varepsilon_{ik}-\varepsilon_{ij})\ \forall\ j\neq k \end{aligned} \quad (3)$$

Note that in the probability only differences in utility matter so that with equation (2) angler-specific characteristics such as income are differenced away and have no role in the model. Following one common approach in the recreation demand literature, we assume ε_{ij} is distributed generalized extreme value. This yields the nested logit site choice model, which allows alternatives to be placed in groups to account for unobserved similarities between grouped alternatives. Within a group the alternatives are assumed to share common but unobserved characteristics that drive correlation between choices. We adopt a two-level model, where the upper level consists of the choice of group and the lower level consists of the choice of alternatives within the preferred group. We distinguish the alternatives by their cold water and warm water classification so the nested model consists of two groups. The probability of visiting a particular site j is therefore

$$prob_i(choose\ j)=e^{V_{ij}/\theta}\times\left[\sum_{k=1}^{N_g}e^{V_{ik}/\theta}\right]^{\theta-1}\Bigg/\sum_{g=1}^{G}\left[\sum_{k=1}^{N_g}e^{V_{ik}/\theta}\right]^{\theta} \quad (4)$$

where N_g is the number of sites in group g (in our particular case g = cold water, warm water) and θ is a "dissimilarity" parameter that captures the degree of correlation between alternatives within a group.

There are several types of variables used in the RUM model. Of primary interest are the targeted biomass variables, *brook trout*, *brown trout*, *smallmouth bass*, *panfishes* and *walleye*, measuring the fishing quality at each site. Although one could argue that biomass does not directly enter into an angler's utility function, using it as a measure of fishing quality has several advantages over catch rates: First, catch data gathered from surveys where the anglers are sampled rather than the sites tend to produce expected catch rates with measurement error, particularly for the least visited sites, and therefore biased demand model parameters [*Morey and Waldman*, 1998; *Train and McFadden*, 2000]. Second, catch is a function of biomass and fishing effort, which is endogenous [*Clark*, 1990; *Harley et al.*, 2001], so using biomass can be viewed as a sort of reduced-form approach to measuring site quality independent of effort. Third, fisheries managers in Michigan tend to stock streams based on added fish per unit area, which is akin to our biomass formulation [*Dexter and O'Neal*, 2004]. Of course, anglers might care about other factors such as fish sizes, but size-specific measures are generally unavailable for both biomass and catch rates.

Next, we include the variable *travel cost* to account for the individual-specific price of taking a fishing trip. The coefficient on this variable reflects the change in utility from a small increase in the cost of visiting a site.

The final set of variables controls for the influence of site-specific features on site choice, including landscape characteristics and built amenities. In the version of the model reported here we use site fixed effects

– that is, a full set of alternative specific constants – to avoid problems with omitted variables bias in the biomass and travel cost parameter estimates [*Moeltner and von Haefen*, 2011; *Weber et al*., 2012]. We considered alternative specifications that combined observable landscape variables with more aggregated fixed effects but the results suggested that controlling for site-specific omitted characteristics was critical. To identify the importance of observed site-specific factors on site choice the estimated fixed effects were regressed on several landscape variables [*Murdock*, 2006], including: *NWSR*, a proxy for the remote and scenic setting around stream segments protected under the National Wild and Scenic Rivers Act of 1968 [*Sanders et al*., 1990]; *forest*, *agriculture*, and *urban*, the percentages of the riparian landscape in different land uses (the omitted category is composed of scrub/shrub, grass and bare land); *length*, the natural logarithm of the aggregate stream lengths in the site (in km); and, finally, *cold water*, an indicator to capture the share of trips taken to cold water sites relative to warm water sites that remains unexplained by the other variables.

The RUM model is parameterized on the Michigan stream angler and biomass data. We use equation (4) to create a likelihood function across the possible choice alternatives for all trips and estimate the parameters by maximum likelihood. To control for monthly changes in MRAS surveying intensity, each trip is weighted by the inverse of the probability that it was collected from a survey in a particular month. Trips are also clustered by angler to account for individuals who have multiple trips in the sample. Due to the use of site fixed effects, the 176 of 408 fishable alternatives that did not receive any visits in the sample could not be included in the final RUM model choice set. We estimated variations of the model without site fixed effects that did and did not include unvisited sites and found few significant changes between the variants, suggesting that this decision has little bearing on the results.

2.4. Value Measurement

Changes in the characteristics and quality of the choice alternatives can be valued using the estimates of the RUM model. Monetary values are computed as anglers' willingness to pay (WTP) to forgo a quality change on a choice occasion [*Haab and McConnell*, 2003]. Following a quality change, WTP is the amount that leaves the angler no better or worse off than before the quality change. Let V_j and V_j^* refer to measurable utility before and after a quality change, respectively. In the context of the RUM model estimated as a nested logit it can be shown that

$$\mathrm{WTP}_i = \frac{-1}{\rho}\left[ln\sum_{g=1}^{G}\left[\sum_{j=1}^{N_g} e^{V_{ij}/\theta}\right]^{\theta} - ln\sum_{g=1}^{G}\left[\sum_{j=1}^{N_g} e^{V_{ij}^*/\theta}\right]^{\theta}\right] \quad (5)$$

per choice occasion. Equation (5) can also be used to estimate the monetary damage of site loss, where the affected alternative is removed from the summation of V_j^* in the right hand side of the equation.

Following estimation of the RUM model, WTP is computed using the estimated parameters and the observed quality measures for V_j and quality measures for V_j^*. In our applications, we report WTP for several quality change scenarios. The first set of scenarios measure the benefits arising from a 50% increase in biomass for each species at all sites. The second set of scenarios evaluates the benefits arising from a 1 kg per ha increase in biomass at all sites. Each of these WTP estimates is a type of per-trip gain, and should be interpreted as the expected monetary benefit across day trips to every fishing site in the model.

We also examine the monetary damages from closing some of the fishable alternatives. These damages are calculated as loss-to-trip ratios by evaluating equation (5) and dividing by the average probability that a trip was taken to the affected (closed) site [*Parsons et al*., 2009]. Loss-to-trip ratios are interpreted as the monetary damage to those fishing trips taken specifically to the lost site. Whether expressed as values across all trips in the choice set as in equation (5) or as loss-to-trip ratios, the measures are highly nonlinear in the estimated parameters. Thus, confidence intervals were computed by bootstapping the estimation of the model parameters 200 times.

3. Results and Discussion

3.1. RUM Model Estimates

Table 4 presents the estimated parameters of the RUM model. The travel cost parameter has the expected negative sign and is statistically significant at the 0.01 level, indicating that the probability of a trip to a site is decreasing in the trip price. Overall, the RUM model predicts a strong targeted biomass effect. The biomass parameters are positive and significant at traditional confidence levels for all five species. These estimates show that Michigan stream anglers respond to differences in fish abundance between sites and, specifically, that the probability of visiting a site increases with targeted biomass.

The estimates demonstrate that anglers do not react equivalently to changes in fish biomass across species. The hypothesis that the effect of targeted biomass on site choice is the same for all species is rejected at a high confidence level. Of the biomass parameters, the point estimates are greatest for brook trout and walleye, implying that anglers' site preferences are particularly sensitive to the biomass of these two species.

The fixed effects add significantly to the model based on the Akaike information criterion goodness-of-fit measure. For brevity the 232 estimates for these parameters are not reported but, in general, the fixed effects suggest that unmeasured site attributes enjoyed by all anglers tend to be important components of utility. The role of observed site attributes on site choice can be gauged through an auxiliary regression of the estimated fixed effects on site-specific variables [*Murdock*, 2006]. The results of this procedure in the present case are reported in Table 4 (that auxiliary regression also included 12 basin fixed effects which are omitted for brevity). The landscape variable estimates indicate that anglers tend to fish at sites with the National Wild and Scenic Rivers designation but avoid sites with a high proportion of urban or agricultural development in the riparian area, other things being equal.

The results further suggest that there are unobserved characteristics that are correlated within the nested groups. The dissimilarity parameter, which was constrained to be equal across groups, is significantly different from 1, suggesting that alternatives within the cold water or warm water group exhibit more similarities with alternatives in their own group than with alternatives in the other group. Though not reported here, we also considered a specification with different dissimilarity parameters by nests; we found this had a negligible effect on the estimated effects, although it did suggest that cold water alternatives were less correlated with one another than warm water alternatives.

3.2. Benefit Estimates

Welfare estimates are calculated for a 50% and for a one kilogram per hectare increase in biomass at all sites for each species (Table 5). Although these scenarios are for illustration of the model, in practice managers do adopt stocking strategies based on the added weight of a particular species per unit area [*Dexter and O'Neal*, 2004]. As discussed above, in these scenarios WTP is expressed in terms of a trip taken to any river or stream in Michigan.

WTP varies between the two welfare scenarios largely due to differences in the estimated parameters on the targeted biomass levels and differences in the mean targeted biomasses (see last column, Table 5). For example, the value of changing walleye biomass is less than that for panfishes for an equivalent percentage increase in in situ biomass, though walleye is more valuable per unit biomass. A 50% increase in walleye is worth about \$1.1/trip while a 50% increase in panfishes is worth about \$3.7/trip, but this equi-proportional increase in targeted biomass leads to a much greater total increase in panfishes (about 0.897 kg ha^{-1}) than in walleye (about 0.028 kg ha^{-1}). The WTP for walleye is greater for an equal increase in biomass: a 1 kg ha^{-1} increase is worth about \$4.0/trip for walleye versus \$1.5/trip for panfishes.

Overall, these estimates imply that increasing brook trout and walleye abundance would return the most value to Michigan's stream fisheries. These two game fish species also happen to have the least in situ biomass of the species considered in the model (Table 3 and last column of Table 5).

Comparing the WTP estimates from our quality change scenarios with those reported in the literature is difficult because our measures of fishing quality are distinct from prior studies. The ranking of values we identify is similar to *Murdock*'s [2006] results for a RUM model of Wisconsin fishing; both indicate that anglers are willing to pay significantly for increases in walleye and trout abundance. *Melstrom and Lupi* [2013] find that on average Great Lakes anglers are willing to pay \$4–6 per trip to avoid a 50% decline in walleye catch rates, which is more than our own willingness to pay estimate of about \$1 to obtain a 50% increase in walleye biomass in rivers (that could be expected to have a proportional impact on walleye catch); however, this difference may be attributable to the larger share of anglers who target walleye in the Great Lakes.

The average loss-to-trip ratio ranges from about \$19–23 depending on the closed site. For example, we find that, on average, trips to the warm water portion (i.e., the main stem) of the Muskegon River below Hardy

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Table 5. Average per Trip WTP ($) for Increase in Targeted Biomass[a]

Species	50% Increase	1 kg ha^{-1} Increase	RUM Model Mean Targeted Biomass (kg ha^{-1})
Brook trout	2.372 (0.773–6.971)	7.104 (2.655–14.670)	0.249
Brown trout	3.370 (1.605–7.931)	2.346 (1.275–4.523)	1.198
Smallmouth bass	1.707 (0.836–2.571)	1.567 (0.836–2.216)	0.531
Panfishes	3.692 (2.531–4.836)	1.549 (1.073–1.984)	1.793
Walleye	1.149 (0.704–1.771)	4.032 (2.649–5.792)	0.055

[a]WTP 95% confidence intervals in parentheses below estimates computed by bootstrapping the model 200 times.

Dam, the most popular fishing alternative in the sample (receiving about 6% of sample trips), are worth $23 (95% confidence interval: $21–$24). For trips to the southern watershed of the Au Sable River, a more typical sportfishing site (receiving less than 1% of sample trips), we estimate an average value of $19 ($17–$21). Jointly closing groups of alternatives or entire river systems will produce higher damages on a per-trip basis: we find that access to the Au Sable River system subbasin has a mean estimated value of $26 ($23–$28). Furthermore, we estimate that access to the northwestern Lake Huron basin, which includes the Au Sable River subbasin, has a mean value of $45 ($40–$51).

The damages of lost access that we estimate, about $20 per trip, are somewhat smaller than those reported in the literature due in part to the comparatively fine scale of our site definitions. Not surprisingly, our damage estimates grow closer to these other estimates after conditioning on the scale of lost access. For example, *Train* [1998] estimates that the Madison River in Montana is worth around $40 per trip and *Von Haefen* [2003] estimates that the lower Susquehanna River is worth about $30 per trip, after adjusting for inflation. Both of these sites are on the scale of a subbasin, which makes the Train and the von Haefen estimates very similar to our own for access to subbasins in Michigan.

4. Conclusion

This paper developed a site choice model capable of valuing recreational fishing quality at Michigan rivers and streams. The objective was to identify angler preferences for various fish—trout, bass, panfishes and walleye—using species-specific biomass as an exogenous measure of fish abundance. Prior research has largely relied on presence-absence indicators or average catch rates to characterize fishing quality and was not designed to value a variety of individual fish species or biomass. Our model took advantage of species-specific biomass measures in order to derive anglers' willingness-to-pay for improvements in the quality of fishing for individual species. Our estimates indicate that anglers, conditional on the species or species groups they are targeting, tend to visit sites that are high in fish biomass. In particular, we found that brook trout followed by walleye had the most valuable biomasses for stream fishes in Michigan.

The set of fishable alternatives used in the model was characterized by watershed boundaries. These boundaries resulted in watershed areas that were generally 100–200 square miles (260–520 km^2) in size with the site containing a short river reach and its fishable tributaries. This site definition is useful because, first, it allows the researcher to value changes in the quality at a variety of watershed levels and, second, it was based on both stream temperatures and USGS hydrologic units (10-digit HUC), so the classification could be applied to any US state or region.

There are some caveats to this analysis that could be addressed by future research. The model only included single-day trips, and thus may not capture values and substitution in the same manner as a model that incorporates the behavior of anglers who take multiple-day trips. Furthermore, while angler heterogeneity was partially embedded into the model via targeted-species preferences, further insight may be gained by exploring the influence of other observable and unobservable angler characteristics on site choice. In terms of the species-specific biomass measures, our sites include tributaries considered fishable, but smallmouth bass were predicted by *Esselman et al.* [2014] to be limited to larger rivers, which might be taken into account in future site definitions focused on bass angling. Finally, angler welfare may be influenced by both the rate and size of catch, which biomass cannot distinguish between. The willingness of anglers to tradeoff catch rate for catch size needs further study.

Managing aquatic ecosystem services requires knowledge about the benefits that users gain from the resource. This paper provided benefit estimates that can be easily used in cost-benefit analysis. Although the model was applied to stream anglers taking single-day trips in Michigan, we expect that the reported WTP estimates are suitable for benefits transfer to streams around the Midwest and the Great Lakes region.

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