Enhanced Geospatial Data for Meta-Analysis and Environmental Benefit Transfer: An Application to Water Quality Improvements

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Abstract

Meta-regression models (MRMs) are commonly used within benefit transfer to approximate mean willingness to pay (WTP) for environmental improvements. With rare exceptions, theory suggests that these estimates should be sensitive to core economic factors including geospatial scale (the geographical size of affected environmental resources or areas), market extent (the size of the market area over which WTP is estimated) and substitute availability (the availability of proximate, unaffected substitutes). No MRMs in the current valuation literature enable simultaneous adjustments for these factors, leading to benefit transfers that fail key tests of content validity. This paper reports on a novel meta-analysis for US water quality benefit transfer that incorporates quantitative measures of these and other spatially explicit factors predicted by theory to influence WTP. The metadata combine primary study information with extensive geospatial data from geographic information system (GIS) data layers and other external sources. The result is the first meta-analytic benefit function able to test and adjust for combined value surfaces associated with quantitative measures of geospatial scale, market extent and spatially differentiated substitutes. Scenario analyses demonstrate that these adjustments can be crucial to transfer accuracy.

Introduction

Meta-analyses in environmental economics are commonly used to evaluate systematic influences of study, economic, resource and population attributes on measures of nonmarket willingness to pay (WTP), and to generate parameterized functions for use in benefit transfer (Bergstrom and Taylor 2006; Boyle et al. 2013; Johnston and Rosenberger 2010; Nelson and Kennedy 2009; Smith and Pattanayak 2002). Within meta-regression models (MRMs) used for such purposes, the dependent variable is most often a comparable mean or median welfare (e.g., WTP) measure drawn from existing primary valuation studies.[1] Independent moderator variables represent observable factors hypothesized to explain variation in this measure across observations. MRMs of this type have been used to estimate benefit functions for many different types of environmental policy effects, such as changes to water quality, air quality, wetlands, fisheries, coral reefs, outdoor recreation sites, endangered species, and others (Boyle et al. 2013; Johnston and Rosenberger 2010; Nelson and Kennedy 2009). Benefit transfers from these functions—typically predicting mean per household WTP—have been used to support multiple types of policy analysis (e.g., US EPA 2010, 2011, 2012).

With rare exceptions, theory suggests that these transferred welfare estimates should be sensitive to core economic factors including geospatial scale (the geographical size of affected environmental resources or areas), market extent (the size of the market area over which WTP is estimated) and substitute availability (the availability of proximate, unaffected substitutes). Yet despite significant advances in meta-analytic benefit transfer over the past decade, no MRMs in

[1] Consistency or comparability of these welfare measures is required across multiple dimensions. Commodity consistency requires that the nonmarket commodity being valued is approximately the same across studies included in the metadata. Welfare consistency requires that these welfare measures represent comparable theoretical constructs. Only observations that satisfy a minimum degree of welfare and commodity consistency should be pooled within metadata (Bergstrom and Taylor 2006; Johnston and Moeltner 2014; Nelson and Kennedy 2009; Smith and Pattanayak 2002).

the current literature enable simultaneous adjustments for these core factors. The resulting benefit transfers may hence fail important tests of content validity.[2] Specifically, these benefit transfers do not exhibit sensitivity to core geospatial variables that should—according to theory—exert important influences on WTP.

In the context of water quality valuation, for example, the *geospatial scale* of a water quality change might reflect the size of the water body or area affected. The sampled *market area* would reflect the location of populations for which values were estimated by the primary study (Loomis 2000; Loomis and Rosenberger 2006). For example, were values measured for residents of a community, watershed, state, region or nation? Past work has shown such differences to have important implications for welfare estimates, as predicted by theory (Bateman et al. 2006; Johnston and Duke 2009). *Availability of substitutes* would reflect the spatially-variable quantity or quality of substitute resources in the surrounding geographical area (Loomis and Rosenberger 2006; Schaafsma et al. 2012). For example, we might expect that households' WTP to improve water quality in a single lake might depend on the existence and size of other, substitute lakes in the surrounding region.

All of these factors are potentially relevant to the expected welfare gain from environmental improvements, and at least in principle should be incorporated within benefit transfers. Yet while there has been significant attention to measures used to reconcile environmental *quality* measures across primary studies (e.g., Johnston et al. 2005; Loomis and Rosenberger 2006; Van Houtven et al. 2007), there has been less attention to core aspects of geospatial context. Among the reasons for this lack of attention is the tendency of primary

[2] Content validity relates to whether a measure (here, a WTP estimate provided by a benefit function) adequately reflects the construct's domain (here, the core factors suggested by theory to influence WTP). In this context, invariance of a transferred WTP estimate to the core economic factors expected to influence welfare would constitute a lack of content validity.

studies to omit information on geospatial aspects of studied resources, market areas and populations (Loomis and Rosenberger 2006). Inclusion of these data in MRMs hence requires the meta-analyst to reconstruct these variables, typically by combining information in primary studies with external geospatial data such as that available from geographic information system (GIS) data layers. Metadata supplementation of this type has been used to incorporate various types of information within MRMs (e.g., Londoño and Johnston 2012; Ghermandi et al. 2010; Ghermandi and Nunes 2013), but never to accommodate the set of core economic factors addressed here.

This paper reports on a novel meta-analysis for US water quality benefit transfer that incorporates quantitative measures of core, geospatial factors predicted by theory to influence WTP. These include geospatial scale, market extent and spatial substitute availability. The metadata combine information reported by primary studies with extensive geospatial data derived from external, spatially-explicit databases.[3] The result is the first meta-analytic benefit function able to explicitly test and adjust for related value surfaces. Results illustrate theoretically anticipated scale and substitution effects that have not previously been identified by MRMs in the valuation literature. The resulting functions allow heretofore unavailable adjustments for these core economic variables, enhancing the content validity and potential accuracy of resulting benefit transfers. Results also suggest a much greater capacity for MRMs to incorporate theoretically-supported geospatial factors than is reflected in the current literature, and the errors that can result when associated value surfaces are ignored.

[3] Examples include the National Hydrography Dataset (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php), the Hydrologic Unit Code Watershed Boundary Dataset (http://water.usgs.gov/GIS/huc.html), the National Land Cover Database (http://www.mrlc.gov), and boundary shape files from the US Census (http://www.census.gov/geo/maps-data/data/tiger.html), among others.

Geospatial Scale in Meta-Analysis and Benefit Transfer

As noted above, the validity and accuracy of benefit transfers depend upon the ability of transfer methods to account for geospatial factors that exert systematic influences on WTP. Yet, despite a spatial dimension being implicit in all benefit transfers, MRMs almost universally apply simplistic approaches to characterize associated value patterns. A common example is the use of dummy variables to distinguish broad size categories of affected resources (e.g., large versus small wetlands; Brouwer et al. 1999). Although some MRMs have supplemented primary study metadata using external spatial data sources (e.g., Brander et al. 2007; Ghermandi and Nunes 2013; Londoño and Johnston 2012), none of these enable transfers that account for the simultaneous effects of geospatial scale, market area and spatial substitute availability.[4]

Among the core geospatial factors with potentially important influences on WTP is the size of the area over which environmental changes occur, or geospatial scale. For example, one might expect WTP for an improvement in lake water quality to be positively related to the size or number of affected lakes, *ceteris paribus*. Despite this expectation, most valuation MRMs omit variables characterizing the geospatial scale of affected resources. Others include only low-resolution qualitative variables. Examples include categorical variables identifying (1) whether quality changes affect single or multiple areas, (2) relative size categories such as "large," "medium" and "small," or (3) the type of geographic areas addressed by the analysis, e.g., improvements at a national, regional or local level (Brouwer et al. 1999; Johnston et al. 2003, 2005; Lindhjem and Navrud 2008; Rosenberger and Loomis 2000; Santos 2007; Van Houtven et al. 2007). A few MRMs incorporate more explicit measures of site area, for example as

[4] For example, Ghermandi and Nunes (2013) develop a set of spatial variables that characterize socioeconomic and ecological characteristics within a 20km buffer of wetland sites in the metadata. However, these variables do not characterize wetland size, the size of market area sampled by each primary study, or the availability of substitutes.

variables explaining WTP for outdoor recreation (Brander et al. 2007; Londoño and Johnston 2012) or values for wetland ecosystem services (Ghermandi et al. 2010). However, to the knowledge of the authors, no MRMs in the published literature incorporate explicit, quantitative measures of both scope[5] (i.e., the magnitude of a resource quality change) and geographic scale (the size of the area over which the quality change occurs).

It is also well-established that mean WTP often declines as one moves further from an affected resource—this is typically denoted distance decay (Bateman et al. 2006; Jørgensen et al. 2013; Schaafsma et al. 2012). In such cases, accurate benefit transfers require one to account for the expected distance of individuals from policy effects (Bateman et al. 2006; Johnston and Ramachandran 2013; Loomis and Rosenberger 2006). These effects are naturally related to the size of the market area or jurisdiction over which values are estimated (Loomis 2000; Loomis and Rosenberger 2006). Patterns such as these have been empirically evaluated in both primary studies and benefit transfers (e.g., Bateman et al. 2006; Johnston and Duke 2009; Jørgensen et al. 2013; Loomis 2000; Schaafsma et al. 2012).[6] Because quantitative measures of resource distance or market area are almost universally omitted within valuation MRMs, transfers applying MRM estimates must use ad hoc assumptions to account for these theoretically-expected welfare patterns.[7] Alternatively, these transfers assume that WTP is invariant to distance (and market area)—an assumption often contradicted by the primary study literature.

[5] For example, the scope or magnitude of a water quality change can be measured using a standard water quality ladder or index (Johnston et al. 2003, 2005).
[6] For example, Johnston and Duke (2009) demonstrate that failure to account for the size of the sampled market area in benefit function transfers can lead to significant transfer errors, with mean per household WTP for a given quantity of farmland preservation approximately two orders of magnitude greater when considered at the community scale than at a comparable state scale.
[7] For example, MRMs that omit measures of the sampled market area cannot account for the fact that per household WTP often declines with distance to an affected resource. Hence, these MRMs will forecast the same WTP value for households regardless of distance. To avoid unrealistically high WTP estimates from these models, analysts may arbitrarily truncate the area over which benefits are estimated, or assume an ad hoc distance decay factor.

Finally, the expected economic value of environmental goods and services is related to the geospatial availability of substitutes (Schaafsma et al. 2012). For example, WTP for improvements to particular water bodies likely depends on the availability of substitute water bodies in the surrounding area (Jørgensen et al. 2013). However, while some authors have speculated that MRM results may be related to differential spatial availability of substitutes (e.g., Brander et al. 2006), and others have included quantitative proxies for substitute availability (e.g., Ghermandi et al. 2010), the authors are aware of no published valuation MRMs that incorporate high-resolution, quantitative measures of unaffected substitutes.

Without an explicit mechanism to adjust WTP estimates for geospatial scale and substitute availability, even MRMs with superior statistical performance may lack the properties necessary to support valid and accurate benefit transfer. These shortcomings are generally overlooked by a valuation meta-analysis literature that too often focuses on statistical performance and systematic value patterns associated with individual moderator variables, with little attention to whether and how model results support applied benefit transfer. Yet without a capacity to adjust for such factors, WTP estimates provided by these models will be invariant to changes in factors such as resource size and market area that should—according to theory and intuition—have important implications for welfare.

Data and Empirical Model

The illustrated MRM was designed to support benefit estimation for policies that improve water quality in US water bodies such as rivers, lakes and estuaries. Among other factors, model design was motivated by the need for benefit transfers to account for theoretically-anticipated WTP variations associated with differences geospatial scale, market area and substitute availability, along with quantifiable changes in scope (the magnitude of water quality change). The metadata are drawn from primary stated preference valuation studies that estimate per household (use and nonuse) WTP for water quality changes in US water bodies that affect ecosystem services including aquatic life support, recreational uses (such as fishing, boating, and swimming), and nonuse values.

To develop model data, we began with the metadata of Johnston et al. (2005), one of the most heavily-cited meta-analyses in the benefit transfer literature.[8] This original metadata was updated and expanded to enable the illustrated modeling. Primary changes included the deletion of studies conducted prior to 1980[9] and others that did not meet updated screening criteria for the present analysis (see details on screening criteria below); the addition of 21 studies not previously included in Johnston et al. (2005), including 8 studies conducted since 2005[10]; and the development of new, spatially-explicit moderator variables. In addition, observations from two papers that were unpublished as of 2005 (Azevedo et al. 2001; Whitehead 2002) were replaced with observations from subsequently published versions of the same studies (Corrigan et al. 2009; Whitehead 2006).

Observations were identified and added to the metadata following the guidelines of Stanley et al. (2013) for research identification and coding. This included documentation of protocols used to identify potential new studies, including (a) the exact databases and other sources searched[11], (b) the precise combination of keywords, and (c) the date completed.

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[8] Google Scholar listed 97 citations to this article as of August 4, 2014.
[9] This was done based on the significant advances in stated preference methods that took place during the 1980s.
[10] Some but not all of these additions were included in a previous update to the metadata (US EPA 2009).
[11] Databases and other sources searched included: (1) general literature databases and search engines (EBSCO, Google Scholar, Google), (2) online reference and abstract databases (Environmental Valuation Resource Inventory (EVRI), Benefits Use Valuation Database (BUVD), AgEcon Search, RePEc/IDEAs), (3) webpages of authors and university program known to publish stated preference studies and/or water quality valuation research, (4) web sites of organizations and agencies known to environmental and resource economics valuation research (e.g., Resource

Specific details on keywords and dates are suppressed here for conciseness, but are available from the authors upon request. Following recommendations of Stanley et al. (2013), study review, identification and coding were completed and verified by multiple individuals, with all variables and coding documented clearly.

In addition to the required study characteristics identified above, studies were further screened according to a number of criteria to ensure validity, consistency and applicability. To ensure welfare consistency, observations were limited to US studies that estimate total (use & nonuse) value, use generally accepted stated preference methods and models, and report theoretically comparable Hicksian welfare measures (Boyle et al. 2013). Studies omitting information needed to quantify WTP, evaluate methods, or document key resource, context, and study attributes were excluded. Necessary data included information identifying affected water bodies, the extent of water quality change, and sampled market areas, along with core methodological attributes. Finally, studies were limited to those for which per household WTP estimates could be readily linked to water quality changes measured on the standard 100-point Water Quality Index (WQI); this index is a simple linear transformation of the 10-point water quality ladder used by Johnston et al. (2005).[12] This screening led to the exclusion of studies for which WTP for water quality could not be disentangled from WTP for other ecosystem changes (e.g., riparian land restoration). Additional details on the reconciliation of water quality measures across studies are provided below.

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for the Future, National Center for Environmental Economics), (5) websites of key resource economics journals for the years 2005-2013 (*Land Economics*, *Environmental and Resource Economics*, *Marine Resource Economics*, *Journal of Environmental Economics and Management*, *Water Resources Research*, and *Ecological Economics*).

[12] Additional details on the WQI and the use of the WQI in survey instruments are provided by McClelland (1974), Mitchell and Carson (1989, p. 342), and Vaughan (1986). This index is linked to specific pollutant levels, which in turn are linked to presence of aquatic species and suitability for particular recreational uses. The WQI allows the use of objective water quality parameters (e.g., dissolved oxygen concentrations) to characterize ecosystem services or uses provided by a given water body. The water quality ladder of Vaughn (1986) is expressed on a scale of 0 to 10 and can be mapped to the WQI by multiplying by 10 (USEPA, 2009)

The resulting metadata include 143 observations from 52 stated preference studies conducted between 1981 and 2011. Multiple WTP estimates from a single study are available due to in-study variations in such factors as the extent of amenity change, elicitation methods applied, market area sampled, water body type and number, and recreational uses affected. The inclusion of multiple observations per study is standard valuation metadata (Nelson and Kennedy 2009). All monetary values are adjusted to 2007 US dollars. The dependent variable for all estimated MRMs is the natural log of household WTP for water quality improvements measured on the 100-point WQI. Table 1 summarizes principal study characteristics for studies included in the metadata.

Independent variables in the metadata characterize (1) study methodology and year, (2) region and surveyed populations, (3) sampled market areas and study site, (4) affected water bodies, and (5) water quality improvement. *Study methodology and year* variables characterize such features as the year in which a study was conducted, payment vehicle and elicitation formats, and WTP estimation methods. *Region and surveyed populations* variables characterize such features as the US region in which the study was conducted, the average income of respondent households and the representation of users and nonusers within the survey sample. *Sampled market area and study site* variables characterize features such as the size of the market area over which populations were sampled, as well as land cover and the quantity of substitute water bodies. Characteristics of *affected resources* include hydrological features (i.e., water body type), recreational uses affected by the proposed water quality changes, and measures of geospatial scale (e.g., shoreline length, river flow). Finally, *water quality baseline and change* variables characterize baseline conditions and the extent of the water quality change. Following standard econometric practice and the Weak Structural Utility Theoretic (WSUT) approach to meta-analysis (Bergstrom and Taylor 2006), these variables were selected and specified based on guidance from theory and prior literature.

When specifying the model, emphasis was given to core economic and resource variables directly relevant to benefit transfer. Methodological and study variables are included to capture related value surface patterns identified previously in the literature, and particularly where methodological variations imply concomitant theoretical or empirical variations in the type of welfare estimate that is reported (e.g., mean versus median WTP). Variables were also included to test for systematic value surfaces associated with different publication types. At the same time, care was taken to avoid over-parameterizing the model with methodological variables, as this can lead to models with seemingly good statistical fit that nonetheless have poor transfer performance (Bateman et al. 2011). Table 2 summarizes the set of independent variables included in the meta-analysis.

Reconciling Measures of Water Quality Change

A critical component metadata development is the reconciliation of variables across observations (Johnston et al. 2005, Smith and Pattanayak 2002, Smith et al., 2002). Although the calculation and reconciliation of most independent variables requires little explanation, there are some variables for which additional detail is warranted. These include variables characterizing surface water quality and its measurement. To reconcile measures of water quality across studies we apply the prior approach of Johnston et al. (2005), mapping all water quality changes to the standard 100-point Water Quality Index (WQI) as noted above.[13]

A large number of the original studies in the metadata (30% of observations) include

[13] See Van Houtven, Powers and Pattanayak et al., (2007) for an alternative means of reconciling water quality measures.

WQI or related 10-point water quality ladder measures as a native component of the original primary studies. For these studies, no additional transformations were required. In most other cases the descriptions of water quality rendered mapping of water quality measures to the WQI straightforward. In cases where baseline and improved (or declined) water quality was not defined by suitability for recreational activities (e.g., boating, fishing, and swimming) or corresponding qualitative measures (e.g., poor, fair, good) that could be readily mapped to the WQI, we used descriptive information available from studies (e.g., amount/indication of the presence of specific pollutants; effects on sensitive aquatic species) to approximate the baseline level of water quality and the magnitude of the change. Preliminary models failed to identify any systematic variation in results associated with studies for which the WQI was a native component, versus those for which quality changes were mapped to the WQI.

Geospatial Analysis and Variables

Significant attention was also given to the development and testing of variables characterizing geospatial scale, market extent and regional substitutes. Variable development was guided by theory and information available from primary study reporting and external databases. The data necessary to derive these variables were extracted from databases including the National Hydrography Dataset (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php), Hydrologic Unit Code Watershed Boundary Dataset (http://water.usgs.gov/GIS/huc.html), National Land Cover Database (http://www.mrlc.gov), and US Census (http://www.census.gov/geo/maps-data/data/tiger.html). None of these variables could be calculated using data reported in primary studies alone.

We measure geospatial scale using the shoreline length of each affected water body.

Shoreline length (*shoreline*) is calculated in kilometers using data layers from the GIS databases identified above, and accounts for the fact that affected river reaches have both a left and right bank. Among other advantages, this provides a measure of geospatial scale (the scale of affected water bodies) that is quantifiable and directly comparable across all affected water bodies, regardless of type. Measuring scale using shoreline length also improved model performance relative to alternative measures of scale that varied in measurability or interpretation across water body types (e.g., surface area).

Market area (*sa_area*), in contrast, is defined as the size of the geographic areas sampled by the stated preference survey, based on sampled areas identified by each primary study. This is also the geographic area over which population-mean (or median) WTP was calculated. This area is defined in square kilometers, and is also derived from external data.

Multiple specifications including geospatial scale and market area were tested in preliminary models. Within these models, performance was enhanced (e.g., in terms of model fit, significance of individual variables, and correspondence of results with theoretical expectations) when the effect of geospatial scale (*shoreline*) was modeled as a function of market area (*sa_area*). That is, MRM performance improves when we allow the marginal effect of water body scale on WTP to vary (declining) as size of the market area increases. This is an intuitive finding. The resulting composite variable is included in the model as a natural log *ln_rel_size = ln(shoreline* / *sa_area)*, and may be interpreted as an index of the relative size of the affected water body relative to the size of the sampled market area.[14] Table 2 provides additional details on this variable.

For a river, scale relates not only to the length of a river but the amount of water it

[14] Including separate scale variables for each water body type (e.g., rivers, lakes, bays) did not improve performance in preliminary models. Hence, modeling proceeds with the single, composite index discussed above.

carries. Hence, an additional scale variable is measured as the quantity of water flowing through affected rivers (*riv_flow*), in cubic feet per second, based on data from the National Hydrography Dataset (http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php). For water bodies with no identifiable flow (e.g., lakes), this variable is set to zero. This variable enables the model to account for the fact that rivers of similar lengths can nonetheless have different flow characteristics, and that these may be relevant to the value of water quality improvements.

Finally, the model includes a variable to characterize the proportional availability of substitutes within the surrounding area. We define this variable as the proportion of water bodies (of the same hydrological type) affected by the water quality change, within affected state(s). This variable is inversely related to the quantity of unaffected substitutes. For rivers, the proportion is measured as the length of the affected river reach(es) as a proportion of all reaches that are of the same river order or lower (*sub_frac_reach*).[15] For non-river inland water bodies (e.g., lakes and ponds), the proportion is defined as the area of the affected water body as a proportion of all water bodies of the same National Hydrography Dataset classification (*sub_frac_area*). For bays and estuaries, the proportion is defined as the shoreline length of the water body as a proportion of all analogous (e.g., coastal) shoreline lengths (*sub_frac_bay*). These are combined into a composite variable, *sub_frac*, defined as max(*sub_frac_reach*, *sub_frac_area*, *sub_frac_bay*) for each observation.[16] Model performance does not improve when including separate substitute variables for each water type (*sub_frac_reach*, *sub_frac_area*, *sub_frac_bay*); hence the final MRM includes only the composite variable *sub_frac* (Table 2).

[15] The concept if river order is used as a measure of relative size, with smaller-order streams flowing into larger order streams. For example, the convergence of two first-order streams forms a second order stream, etc.
[16] This specification provides an unambiguous measure of this variable for a few observations that include improvements to multiple water body types.

The Meta-Regression Model

We estimate the MRM as a multi-level model of the type applied commonly within the literature (Nelson and Kennedy 2009). The model allows for cross-sectional correlation among observations from the same study. If left unaddressed, such correlation can lead to heteroskedastic errors and inefficient, inconsistent parameter estimates (Rosenberger and Loomis 2000b). For each study in the metadata, a central tendency measure (mean or median) of WTP for the representative individual is given by $\bar{y}_{js}$, which is the measured effect size in the MRM:

$$\bar{y}_{js} = \bar{x}_{js}\beta + \varepsilon_{js}. \qquad (1)$$

Here, $\bar{y}_{js}$ is the welfare measure for observation *s* in study *j* (here the natural log of WTP), and $\bar{x}_{js}$ is the vector of independent moderator variables discussed above. The vector β represents a conforming vector of parameters to be estimated.

To allow for potential effects of study-specific unobservable factors, we partition ε_{js} into two components such that

$$\varepsilon_{js} = u_s + e_{js}. \qquad (2)$$

Here, u_s represents a systematic, normally distributed, study-level random effect with $E(u_s) = 0$ and $Var(u_s) = \sigma_u^2$, and e_{js} is a standard *iid* estimation level error, distributed with a zero mean and constant variance σ_e^2 (Shrestha and Loomis 2001). Clustering by study to account for within-study correlation is standard practice (Nelson and Kennedy 2009). Other aspects of the econometric model follow relatively standard conventions for valuation MRMs; we estimate the model using an unweighted GLS random-effects model with robust standard errors (Nelson and

Kennedy 2009).[17]

Two model specifications are estimated. The first is an unrestricted model including variables that characterize geospatial scale, market area and unaffected substitutes. The second is an otherwise identical restricted model that omits these three variables; this model is analogous MRM specifications common in the literature. An equivalent trans-log specification is used for both MRMs, based on input from preliminary models using alternative functional forms. This specification incorporates the natural log of the dependent variable (WTP per household) on the left hand side and the natural logs of household income (*lnincome*), water quality change (*lnquality_ch*), relative geospatial scale (*ln_rel_size*, for the unrestricted model only), and the proportion of agricultural land in the affected area (*ln_ar_agr*) on the right hand side (Table 2). All other dependent variables enter in linear form. General advantages of this functional form for meta-analysis are discussed by Johnston et al. (2005), and include an ability to capture curvature in the valuation function and multiplicative rather than additive effect of independent variables on WTP, and the implied constraints that WTP approaches zero when water quality change, income, and relative geospatial scale also approach zero.

Results and Discussion

Results are illustrated in Table 3 for both the unrestricted and restricted models. For the unrestricted model, a Wald χ^2 test indicates that parameter estimates associated with model variables are jointly significant at $p<0.0001$ ($\chi^2 = 8112.91$, df. 27), with a model R^2 of 0.63. Both

[17] Nearly identical results are obtained when using cluster-robust OLS estimation. It is standard practice in MRMs outside of the valuation literature to estimate models using weighted least-squares with inverse variances or standard errors from the primary studies as analytical weights (Nelson and Kennedy 2009). Such practices are rarely applied within meta-analyses of WTP because (1) variances or standard errors are often unreported by primary studies, (2) WTP variances and standard errors (as well as proxies such as sample sizes) cannot be directly and meaningfully compared across model types (e.g., linear versus discrete choice regressions; mixed versus conditional logit).

measures of model fit decline for the restricted model (χ^2 = 1075.88, df. 24; R^2 = 0.58). A Wald χ^2 test further indicates that the restrictions are jointly significant at $p<0.0001$ (χ^2 = 22.15, df. 3), suggesting that the omission of the three parameters (*ln_rel_size*, *sub_frac*, and *riv_flow*) has a statistically significant effect on the model. Of 27 non-intercept parameter estimates in the unrestricted model, 20 are statistically significant at $p<0.10$, with the majority statistically significant at $p<0.05$. Random-effects account for approximately 7.7% of total model variance. Statistical properties of the unrestricted MRM compare favorably to prior meta-analyses in the literature (Nelson and Kennedy 2009).[18]

Signs of statistically significant parameter estimates in both MRMs match those suggested by theory and intuition, where prior expectations exist. For example, within the unrestricted model, WTP is positively related to the magnitude of water quality change (*lnquality_ch*), household income (*lnincome*), and one-time payments (*lump_sum*), among other factors. These are all intuitive and expected findings. Also intuitively, nonuser samples (*nonusers*) are associated with systematically lower WTP estimates than user or general population samples, as are quality improvements in areas for which boating is a primary specified use (*boat_use*). These and other findings are parallel to those reported in prior meta-analyses, including Johnston et al. (2005).

The primary purpose of this analysis, however, is to evaluate the capacity of the MRM to support benefit transfers that account for variations in the geographic scale of affected resources, market area, and the availability of substitutes. As noted above, combined value surfaces

[18] A suite of robustness tests were also conducted on preliminary versions of the MRM. These included different horizontal and vertical robustness tests recommended by Boyle et al. (2013). While these tests find some evidence of horizontal and vertical fragility in certain dimensions, the weight of evidence suggests the meta-regression is relatively robust. This includes a high degree of robustness associated with the primary policy variables of interest (e.g., the effect of water quality change). Given the extensiveness of these tests and the fact that they did not suggest that specification changes were warranted, the results are omitted here for conciseness.

associated with such geospatial factors have not been estimated by prior MRMs in the published literature. Because of this emphasis, we omit extended discussion of other value surfaces and emphasize results associated with our core variables of interest. Those interested in a broader discussion of non-spatial value surfaces for water quality improvements are referred to Johnston et al. (2005), Johnston and Besedin (2009) and Van Houtven et al (2007), among others.

Implications of Geospatial Moderator Variables

Subsequent discussion focuses on the unrestricted model which includes the three core variables of focus here: *ln_rel_size* (geospatial scale relative to market extent), *sub_frac* (the proportion of similar-type water bodies affected, within affected states) and *riv_flow* (the flow of affected river reaches). Associated coefficient estimates are all statistically significant at the 5% level (Table 3). Signs of coefficient estimates match expectations in call cases; these reflect previously unexplored value surface dimensions associated with water quality improvements.

These dimensions, at least in principle, are directly relevant to benefit transfer. For example, the coefficient estimate for *ln_rel_size* ($p<0.05$) implies that per household WTP increases with the size of affected water bodies (measured by shoreline length in kilometers) relative to the size of the surveyed market area (measured in sq. kilometers). When viewed across (and within) different studies from the literature, improvements to larger water bodies are associated with larger WTP, *ceteris paribus*. Moreover, holding all else constant, stated preference surveys over larger market areas (e.g., over a state versus a small community) are associated with lower per household WTP. This is also intuitive, because larger sampled market areas imply greater distances between individual households and affected water bodies, *ceteris paribus*. Moreover, combining these effects within the single index variable *ln_rel_size* allows the marginal effect of geospatial scale to depend on market area size—another intuitive result. To the knowledge of the authors this is the first published valuation MRM to incorporate these joint effects in quantitative, continuous form. While some prior MRMs have included quantitative measures of affected site area (Brander et al. 2007; Londoño and Johnston 2012; Ghermandi et al. 2010), none of these have also included quantitative variables reflecting the extent of the sampled market.

The coefficient estimate for the variable *sub_frac* ($p<0.05$) further implies that per household WTP increases when a larger proportion of regional water bodies (of the same type) are affected by the proposed policy, again holding all else constant (see Table 2 for explicit definition). That is, when viewed across and within studies in the literature, WTP increases when there are fewer unaffected substitutes. For example, the model predicts larger per household WTP for a water quality improvement over 30% of a state's river kilometers, compared to an otherwise identical policy that only affects 10% of a state's river kilometers (of equivalent order; table 2). As noted above, this is the first valuation MRM to enable an explicit, quantitative adjustment for this type of value surface, and reflects another core economic variable absent from past MRMs.[19]

Finally, the coefficient estimate for the variable *riv_flow* ($p<0.05$) implies that per household WTP increases with the flow of water through affected river reaches, *ceteris paribus*. This finding is also intuitive, and suggests that the concept of geospatial scale is multifaceted. Specifically, information on the length of a river is likely insufficient alone to model welfare-relevant scale; data are also required on other aspects such as river flow, which allows one to

[19] The wetland values MRM of Ghermandi and Nunes (2013) includes a variable quantifying the total quantity of wetlands with a fixed 20km buffer of each valued wetland site, as a proxy for substitute wetlands. However, this variable does not quantify affected versus unaffected areas.

more fully differentiate between smaller and larger rivers.

The importance of these findings for benefit transfer depends not only on their statistical significance, but also on the relative magnitude of each effect (i.e., partial elasticities). In all cases, the partial elasticities associated with these effects appear to be relatively small, at least on an individual basis. The partial elasticity of WTP with respect to the log variable *ln_rel_size*, for example, is 0.05. Parallel elasticities (at mean values) for the linear variables *sub_frac* and *riv_flow* are 0.10 and 0.02, respectively. These individual elasticities viewed in isolation, however, can provide a misleading perspective on the practical policy relevance of these value surfaces, particularly given that these variables may vary in concert. When considered together, the joint impacts of these variables on benefit transfers can be substantial.

Implications for Benefit Transfer

To illustrate the implications of these findings for benefit transfer, we project per household WTP for illustrative water quality improvements, within policy sites that differ in geospatial scale, market extent and substitute availability. This parallels the process that would be used to conduct benefit transfer using MRM results. Other than assumed differences in these three core variables, sites and policies are assumed to be identical. Results are forecast using MRM results in Table 3.

Results are shown in Table 4. The table includes WTP forecasts from the unrestricted model for three illustrative scenarios, chosen to span the range of in-sample values for *sub_frac*, *riv_flow* and *ln_rel_size*. Scenario 1 forecasts per household, annual WTP assuming that *sub_frac*, *riv_flow* and *ln_rel_size* take on mean values from the metadata. Scenario 2 forecasts parallel a parallel WTP estimate assuming that these variables take on their minimum values from the metadata. Scenario 3 forecasts parallel a parallel WTP estimate assuming that these variables take on their maximum values from the metadata. The final row of the table illustrates parallel WTP estimates for each scenario from the restricted model. Because the restricted model omits *sub_frac*, *riv_flow* and *ln_rel_size*, the WTP estimate from this model is identical across all three scenarios.

Other than differences in the three core geospatial variables, the illustrative scenarios are identical, and are designed to represent a typical scenario for which WTP might be forecast. We assume a water quality improvement equal to the mean over the metadata (*lnquality_ch*=2.909); this is equivalent to a change of 18.335 = $e^{2.909}$ on the 100-point WQI, beginning from a mean baseline of *lnbase*=3.596 (36.440 on the WQI). We assume a forecast of annual mean WTP per household (*lump_sum*=0; *wtp_median*=0; *volunt*=0), over a general population sample (*nonusers*=0) in the mid-Atlantic region (*ma*=1; *mp*=*se*=*sw*=*mw*=0), for a water quality improvement in a single river (*river*=1; *mult_bod*=0). These assignments are made to assure interpretability and consistency of the resulting WTP estimates. All other variables are held at mean values from the metadata. Forecasts incorporate the standard intercept adjustment ($\sigma_e^2/_2$) prior to the exponential transformation to obtain an estimate of (linear) mean WTP.

Results suggest that researchers should exercise extreme caution when conducting benefit function transfers that do not account for variations in geospatial scale, the extent of the market, and spatial substitute availability. We begin by considering WTP forecasts from the unrestricted model alone. Note that all scenarios assume an identical change in average water quality. Compared to Scenario 1, which assumes mean values for *sub_frac*, *riv_flow* and *ln_rel_size*, WTP under Scenario 2 (with minimum in-sample values for these variables) declines by 46% ($50.76 versus $27.49 per household; Table 4). In contrast, Scenario 3 (with maximum in-sample values for these three variables) projects a WTP estimate of $234.50—a 362% increase over WTP in Scenario 1. Comparing WTP under Scenarios 2 and 3 reveals an even larger percentage difference of 753% ($27.49 versus $234.50; Table 4). Hence, variations in these three variables alone—over the range of values found in-sample—can lead to a nearly eight-fold difference in projected WTP within the unrestricted model.

In contrast, results of the restricted model generate a WTP estimate of $56.31 for all three scenarios, illustrating the invariance to geospatial factors common in published MRMs. As expected, the restricted model forecast ($56.31) is similar to the unrestricted model forecast that assumes mean values for all geospatial variables ($50.76). However, the unrestricted and restricted model WTP estimates diverge as geospatial variables depart from their mean values. WTP under Scenario 2 (with minimum in-sample values for these variables) differs by 51% between the restricted and unrestricted models ($56.31 versus $27.49 per household; Table 4). WTP under Scenario 3 (with maximum in-sample values for these variables) differs by 317% between the restricted and unrestricted models ($56.31 versus $234.50 per household). That is, even assuming that all other aspects of the transfer are completed without error, failure to account for these three core geospatial factors alone (as is nearly universal in published MRMs) leads to transfer errors of over 300%.

Parallel results are found for other illustrative scenarios (suppressed for conciseness). These results illustrate the substantial transfer errors which can result when relying upon common MRM specifications for benefit transfer, as these models almost universally fail to account for the systematic influence of geospatial scale, market extent and substitute availability on WTP.

Conclusion

The illustrated MRM results quantify systematic WTP value surfaces associated with a variety of theoretically motivated, geospatial factors. Results indicate that stated preferences are sensitive to measures of geospatial scale, market extent and substitute availability, even when these factors are not quantified by stated preference surveys or scenarios. In some cases, the joint influence of these commonly overlooked factors on WTP forecasts can be of greater magnitude than those associated with attributes traditionally included in stated preference benefit functions and associated MRMs. Comparison with a more traditional restricted MRM that omits these variables illustrates the transfer errors that can occur when these value surfaces are overlooked. Valuation MRMs in the published literature almost universally omit these core variables, leading to concerns related to the resulting validity and reliability of associated benefit transfers.

Results of the present analysis must be interpreted within the context of the present case study. The illustrated MRM specification was chosen after extensive preliminary modeling to evaluate alternative means to account for these patterns. Nonetheless, other specifications are possible. Additional research is required to evaluate whether similar MRM specifications and findings are applicable to other types of environmental changes and policy contexts. For example, it is unknown whether the present characterization of geospatial scale (using water body shoreline length)—while effective in the present MRM—is a widely applicable means to quantify this core economic variable other in valuation MRMs. Similar caveats apply to other geospatial variables included in the model. Moreover, it is well known that the influence of geospatial factors on WTP can vary across different types of resources and environmental improvements (Bateman et al. 2006; Schaafsma et al. 2012). Hence, additional work is required to evaluate whether and how results such as these apply to other resource types and valuation contexts.

These caveats aside, results of the present analysis demonstrate clear and intuitive geospatial scale, market extent, and substitution effects that have not been previously established by valuation MRMs. The estimated benefit function enables transfers to accommodate core welfare patterns predicted by theory (e.g., the importance of geospatial scale), yet almost universally overlooked by prior MRMs. These adjustments can be crucial to obtaining accurate benefit transfers. For example, in-sample variations in these variables alone can lead to a nearly eight-fold difference in WTP forecasts, for otherwise identical sites and policies. Such results highlight the potential hazards in meta-analysis and other comparative research that overlook the potential influence of (often study-invariant) geospatial factors. While such factors are often omitted from primary study publications, concomitant disregard of related welfare patterns risks significant transfer error.

While not a panacea, meta-analytic approaches such as those illustrated here can help ameliorate such concerns, and support accurate and defensible benefit function transfers. Beyond case study empirical results and implications, model results also suggest avenues for broader improvements in MRMs and benefit function transfer. These include the capacity to use theoretical expectations as a basis for guiding metadata supplementation—using externally-available geospatial data to characterize variables whose inclusion is supported by theory, but for which information is unreported by primary studies.

References

Aiken, R.A. 1985. Public Benefits of Environmental Protection in Colorado. Masters thesis, Colorado State University.

Anderson, G.D. and S.F. Edwards. 1986. Protecting Rhode Island's Coastal Salt Ponds: An Economic Assessment of Downzoning. Coastal Zone Management 14(½): 67-91.

Azevedo, C., J.A. Herriges, and C.L. Kling. 2001. Valuing Preservation and Improvements of Water Quality in Clear Lake. Staff Report 01-SR 94, Center for Agricultural and Rural Development (CARD), Iowa State University.

Banzhaf, H.S., D. Burtraw, S. Chung, D.A. Evans, A. Krupnik, and J. Siikamaki. 2011. Valuation of Ecosystem Services in the Southern Appalachian Mountains. Paper Presented at the Annual Meeting of the Association of Environmental and Resource Economists (AERE).

Banzhaf, Spencer H, Dallas Burtraw, David Evans, and Alan Krupnick. 2006. Valuation of Natural Resource Improvements in the Adirondacks. Land Economics 82(3): 445-464.

Bateman, I.J., B.H. Day, S. Georgiou and I. Lake. 2006. The Aggregation of Environmental Benefit Values: Welfare Measures, Distance Decay and Total WTP. Ecological Economics 60(2): 450-460.

Bateman, I.J., R. Brouwer, S. Ferrini, M. Schaafsma, D.N. Barton, A. Dubgaard, B. Haslet, S. Hime, I. Liekens, and S. Navrud. 2011. Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe. Environmental and Resource Economics 50(3): 365-387.

Bergstrom, J.C. and L.O. Taylor. 2006. Using Meta-Analysis for Benefits Transfer: Theory and Practice. Ecological Economics 60(2): 351-360.

Bergstrom, J.C. and L.O. Taylor. 2006. Using Meta-Analysis for Benefits Transfer: Theory and Practice. Ecological Economics 60(2): 351-360.

Bockstael, N.E., K.E. McConnell and I.E. Strand. 1989. Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay. Marine Resource Economics 6(1): 1-18.

Bockstael, N.E., McConnell, K.E., and Strand, I.E. 1988. Benefits from improvements in Chesapeake Bay Water Quality. Department of Agricultural and Resource Economics, University of Maryland.

Borisova, T., A. Collins, G. D'Souza, M. Benson, M.L. Wolfe, and B. Benham, 2008. A Benefit-Cost Analysis of Total Maximum Daily Load Implementation. Journal of the American Water Resources Association 44(4):1009-1023.

Boyle, K. J., C. F. Parmeter, B. B. Boehlert and R.W. Paterson. 2013. Due Diligence in Meta-analyses to Support Benefit Transfers. Environmental and Resource Economics 55(3): 357-386.

Brander, L., P.van Beukering, and H. Cesar. 2007. The Recreational Value of Coral Reefs: A Meta-analysis. Ecological Economics, 63(1), 209-218.

Brander, L.M., Florax, R.J.G.M. and J. Vermaat. 2006. The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature. Environmental and Resource Economics 33: 223-250.

Brouwer, R., I.H. Langford, I.J. Bateman and R.K. Turner. 1999. A meta-analysis of wetland contingent valuation studies. Regional Environmental Change 1(1): 47-57.

Cameron, T.A. and D.D. Huppert. 1989. OLS versus ML Estimation of Non-market Resource Values with Payment Card Interval Data. Journal of Environmental Economics and Management 17: 230-246.

Carson, R.T., W.M. Hanemann, R.J. Kopp, J.A. Krosnick, R.C. Mitchell, S. Presser, P.A. Ruud, and V.K. Smith. 1994. Prospective Interim Lost Use Value due to DDT and PCB Contamination in the Southern California Bight. Volume 2. Report to the National Oceanic and Atmospheric Administration, Produced by Natural Resources Damage Assessment Inc., LA Jolla, CA.

Clonts, H.A., and J.W. Malone. 1990. Preservation Attitudes and Consumer Surplus in Free Flowing Rivers. In J. Vining (ed.), Social Science and Natural Resource Recreation Management, Westview Press: Boulder, CO, pp. 301-317.

Collins, A.R., R.S. Rosenberger and J.J. Fletcher. 2009. Valuing the restoration of acidic streams in the Appalachian Region: A stated choice method. In H.W. Thurstone, M.T. Heberling and A. Schrecongost (eds.), Environmental Economics for Watershed Restoration. Boca Raton, FL: CRC/Taylor Francis. pp.29-52.

Collins, Alan R. and R.S. Rosenberger. 2007. Protest Adjustments in the Valuation of Watershed Restoration Using Payment Card Data. Agricultural and Resource Economics Review 36(2): 321-335.

Corrigan, J.R., C.L. Kling, and J. Zhao. 2009. Willingness to Pay and the Cost of Commitment: An Empirical Specification and Test. Environmental and Resource Economics 40: 285-298.

Croke, Kevin, R.G. Fabian, and G. Brenniman. 1986-87. Estimating the Value of Improved Water Quality in an Urban River System. Journal of Environmental Systems 16(1): 13-24.

De Zoysa, A.D.N. 1995. A Benefit Evaluation of Programs to Enhance Groundwater Quality, Surface Water Quality and Wetland Habitat in Northwest Ohio. Dissertation, Ohio State University.

Desvousges,W.H., V.K. Smith and A. Fisher. 1987. Option Price Estimates for Water Quality Improvements: A Contingent Valuation Study for the Monongahela River. Journal of Environmental Economics and Management 14: 248-267.

Downstream Strategies, LLC. 2008. An Economic Benefit Analysis for Abandoned Mine Drainage Remediation in the West Branch Susquehanna River Watershed, Pennsylvania.

Prepared for Trout Unlimited.

Farber, S., and Griner, B. (2000). Using Conjoint Analysis to Value Ecosystem Change. Environmental Science and Technology, 34(8): 1407-1412.

Ghermandi, A. and P.A.L.D. Nunes. 2013. A Global Map of Coastal Recreation Values: Results from a Spatially Explicit Meta-analysis. Ecological Economics 86(1): 1-15.

Ghermandi, A., J.C.J.M. van den Bergh, L.M. Brander, H.L.F. de Groot, and P.A.L.D. Nunes. 2010. Values of Natural and Human-made Wetlands: A Meta-analysis. Water Resources Research 46: W12516, doi:10.1029/2010WR009071.

Hayes, K.M., T.J. Tyrell and G. Anderson. 1992. Estimating the Benefits of Water Quality Improvements in the Upper Narragansett Bay. Marine Resource Economics 7: 75-85.

Herriges, J.A. and J.F. Shogren. 1996. Starting point bias in dichotomous choice valuation with followup questioning. Journal of Environmental Economics and Management 30(1):112-131.

Hite, D. 2002. Willingness to Pay for Water Quality Improvements: The Case of Precision Application Technology. Department of Agricultural Economics and Rural Sociology, Auburn University, Auburn, AL. August.

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(3): 240-255.

Irvin, S., T. Haab, and F.J. Hitzhusen. Estimating willingness to pay for additional protection of Ohio surface waters: contingent valuation of water quality. In F.J. Hitzhusen (ed.), Economic Valuation of River Systems, Cheltenham: Edward Elgar, pp. 35-51.

Johnston, R. J. & R.S. Rosenberger. 2010. Methods, Trends and Controversies in Contemporary Benefit Transfer. *Journal of Economic Surveys* 24: 479–510.

Johnston, R.J. and E.Y. Besedin. 2009. Benefits Transfer and Meta-Analysis: Estimating Willingness to Pay for Aquatic Resource Improvements. In H.W. Thurston, M.T. Heberling and A. Schrecongost, eds. Environmental Economics for Watershed Restoration. Boca Raton, FL: CRC Press.

Johnston, R.J. and J.M. Duke. 2009. Willingness to Pay for Land Preservation Across States and Jurisdictional Scale: Implications for Benefit Transfer. Land Economics 85(2): 217–237.

Johnston, R.J. and M. Ramachandran (2013, print version in press). Modeling Spatial Patchiness and Hot Spots in Stated Preference Willingness to Pay. Environmental and Resource Economics. DOI 10.1007/s10640-013-9731-2.

Johnston, R.J., and K. Moeltner. 2014. Meta-Modeling and Benefit Transfer: The Empirical Relevance of Source-Consistency in Welfare Measures. Environmental and Resource Economics, in press, online version available, DOI 10.1007/s10640-013-9730-3.

Johnston, R.J., E.Y. Besedin and R.F. Wardwell. 2003. Modeling Relationships Between Use and Nonuse Values for Surface Water Quality: A Meta-Analysis. Water Resources Research 39(12), p. 1363-1371.

Johnston, R.J., E.Y. Besedin, R. Iovanna, C. Miller, R. Wardwell, and M. Ranson. 2005. Systematic Variation in Willingness to Pay for Aquatic Resource Improvements and Implications for Benefit Transfer: A Meta-Analysis. Canadian Journal of Agricultural Economics 53(2-3): 221-248.

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

Jørgensen, S.L., S.B. Olsen, J. Ladenburg, L. Martinsen, S.R. Svenningsen and B. Hasler. 2013. Spatially Induced Disparities in Users' and Non-users' WTP for Water Quality Improvements – Testing the Effect of Multiple Substitutes and Distance Decay. Ecological Economics 92(1): 58-66.

Kaoru, Y. 1993. Differentiating Use and Nonuse Values for Coastal Pond Water Quality Improvements. Environmental and Resource Economics 3: 487-494.

Lant, C.L., and G.A. Tobin. 1989. The Economic Value of Riparian Corridors in Cornbelt Floodplains: A Research Framework. Professional Geographer (41): 337-349.

Lant, C.L., and R. S. Roberts. 1990. Greenbelts in the Cornbelt: Riparian Wetlands, Intrinsic Values, and Market Failure. Environment and Planning 22: 1375-1388.

Lichtkoppler, F.R. and T.W. Blaine. 1999. Environmental Awareness and Attitudes of Ashtabula County Voters Concerning the Ashtabula River Area of Concern: 1996-1997. Journal of Great Lakes Resources 25: 500-514.

Lindhjem, H. and S. Navrud. 2008. How Reliable are Meta-analyses for International Benefit Transfers? Ecological Economics 66(2-3): 425-435.

Lindsey, G. 1994. Market Models, Protest Bids, and Outliers in Contingent Valuation. Journal of Water Resources Planning and Management 12: 121-129.

Lipton, D. 2004. The Value of Improved Water Quality to Chesapeake Bay Boaters. Marine Resource Economics 19: 265-270

Londoño Cadavid, C. and A.W. Ando. 2013. Valuing Preferences over Stormwater Management Outcomes Including Improved Hydrologic Function. Water Resources Research 49: 4114-4125.

Londoño, L.M. and R.J. Johnston. 2012. Enhancing the Reliability of Benefit Transfer over Heterogeneous Sites: A Meta-Analysis of International Coral Reef Values. Ecological Economics 78(1): 80-89.

Loomis, J.B. 1996. How Large is the Extent of the Market for Public Goods: Evidence from a

Nation Wide Contingent Valuation Survey. Applied Economics 28(7): 779-782.

Loomis, J.B. 2000. Vertically Summing Public Good Demand Curves: An Empirical Comparison of Economic versus Political Jurisdictions. Land Economics 76: 312-321.

Loomis, J.B. and R.S. Rosenberger. 2006. Reducing barriers in future benefit transfers: needed improvements in primary study design and reporting. Ecological Economics 60: 343-350.

Lyke, A.J. 1993. Discrete Choice Models to Value Changes in Environmental Quality: A Great Lakes Case Study. Dissertation submitted to the Graduate School of the University of Wisconsin, Madison.

Matthews, L.G., F.R. Homans, and K.W. Easter. 1999. Reducing Phosphorous Pollution in the Minnesota River: How Much is it Worth? Staff Paper. Department of Applied Economics, University of Minnesota.

McClelland, N.I. 1974. Water Quality Index Application in the Kansas River Basin. EPA-907/9-74-001, US EPA Region VII, Kansas City, MO.

Nelson, J.P. and P.E. Kennedy. 2009. The Use (and Abuse) of Meta-Analysis in Environmental and Resource Economics: An Assessment. Environmental and Resource Economics 42(3): 345-377.

Olsen, D., J. Richards, and R.D. Scott. 1991. Existence and Sport Values for Doubling the Size of Columbia River Basin Salmon and Steelhead Runs. Rivers 2(1): 44-56.

Opaluch, J.J., T. Grigalunas, M.J. Mazzotta, J. Diamantides, and R. Johnston. 1998. Resource and Recreational Economic Values for the Peconic Estuary. Report prepared for Peconic Estuary Program, Suffolk County Department of Health Services, Riverhead, NY, by Economic Analysis,Inc., Peace Dale, Rhode Island.

Roberts, L.A., and J.A. Leitch. 1997. Economic Valuation of Some Wetland Outputs of Mud Lake. Agricultural Economics Report No. 381, Department of Agricultural Economics, North Dakota Agricultural Experiment Station, North Dakota State University.

Rosenberger, R.S. and J.B. Loomis. 2000b. Panel stratification in meta-analysis of economic studies: An investigation of its effects in the recreation valuation literature. Journal of Agricultural and Applied Economics 32(3): 459-470.

Rosenberger, R. S. and J. B. Loomis. 2000. Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreation database. Water Resources Research, 36, 1097-1107.

Rowe, R.D., W.D. Schulze, B. Hurd and D. Orr. 1985. Economic Assessment of Damage Related to the Eagle Mine Facility. Energy and Resource Consultants, Inc., Boulder, CO.

Sanders, L. B., R.G. Walsh, and J.B. Loomis. 1990. Toward Empirical Estimation of the Total Value of Protecting Rivers. Water Resources Research 26(7): 1345-1357.

Santos, J.M.L. 2007. Transferring landscape values: how and how accurately? In S. Navrud and R. Ready, eds. Environmental Value Transfer: Issues and Methods. Dordrecht, The Netherlands: Springer.

Schaafsma, M., R. Brouwer and J. Rose. 2012. Directional Heterogeneity in WTP models for Environmental Valuation. Ecological Economics 79(1): 21-31.

Schulze, W.D., R.D. Rowe, W.S. Breffle, R.R. Boyce, and G.H. McClelland. 1995. Contingent Valuation of Natural Resource Damages Due to Injuries to the Upper Clark Fork River Basin. State of Montana, Natural Resource Damage Litigation Program. Prepared by: RCG/Hagler Bailly, Boulder, CO.

Shrestha, R. K., J.R.R. Alavalapati. 2004. Valuing Environmental Benefits of Silvopasture Practice: A Case Study of the Lake Okeechobee Watershed in Florida. Ecological Economics 49: 349-359.

Shrestha, R.K. and J.B. Loomis. 2001. Testing a meta-analysis model for benefit transfer in international outdoor recreation. Ecological Economics 39(1): 67-83.

Smith, V.K. and S.K. Pattanayak. 2002. Is Meta-Analysis a Noah's Ark for Non-Market Valuation? Environmental and Resource Economics 22(1-2): 271-296.

Smith, V.K., G. Van Houtven and S.K. Pattanayak. 2002. Benefit Transfer via Preference Calibration: "Prudential Algebra" for Policy. Land Economics 78(1): 132-152.

Stanley, T.D., H. Doucouliagos, M. Giles, J.H. Heckemeyer, R.J. Johnston, P. Laroche, J.P. Nelson, M. Paldam, J. Poot, G. Pugh, R.S. Rosenberger, and K. Rost. 2013. Meta-Analysis of Economics Reporting Guidelines. Journal of Economic Surveys 27(2): 390–394.

Stumborg, B.E., K.A. Baerenklau and R.C. Bishop. 2001. Nonpoint Source Pollution and Present Values: A Contingent Valuation of Lake Mendota. Review of Agricultural Economics 23(1): 120-132.

Sutherland, R. J., and R.G. Walsh. 1985. "Effect of Distance on the Preservation Value of Water Quality." Land Economics 61(3): 282-29.

Takatsuka, Yuki. 2004. Comparison of the Contingent Valuation Method and the Stated Choice Model for Measuring Benefits of Ecosystem Management: A Case Study of the Clinch River Valley, Tennessee. PhD dissertation, University of Tennessee.

United States Environmental Protection Agency (U.S. EPA). 2009. Environmental Impact and Benefits Assessment for Final Effluent Guidelines and Standards for the Construction and Development Category. November.

United States Environmental Protection Agency (U.S. EPA). 2010. Economic Analysis of Final Water Quality Standards for Nutrients for Lakes and Flowing Waters in Florida. Office of Water, Office of Science and Technology.

United States Environmental Protection Agency (U.S. EPA). 2012. Economic Analysis of Proposed Water Quality Standards for the State of Florida's Estuaries, Coastal Waters, and South Florida Inland Flowing Waters. Office of Water, Economic Analysis Division.

United States Environmental Protection Agency (U.S. EPA). 2013. Benefit and Cost Analysis for the Proposed Effluent Limitations Guidelines and Standards for the Steam Electric Power Generating Point Source Category. May.

Van Houtven, G., J. Powers and S.K. Pattanayak. 2007. Valuing Water Quality Improvements in the United States Using Meta-analysis: Is the Glass Half-full or Half-empty for National Policy Analysis? Resource and Energy Economics 29(3): 206–228.

Vaughan, W.J. 1986. The RFF Water Quality Ladder. Appendix B in R.C. Mitchell and R.T. Carson. The Use of Contingent Valuation Data for Benefit/Cost Analysis in Water Pollution Control, Final Report. Washington: Resources for the Future.

Wattage, P. M. 1993. Measuring the benefits of water resource protection from agricultural contamination: Results from a contingent valuation study. PhD dissertation, Forestry, Iowa State University.

Welle, P.G. 1986. Potential Economic Impacts of Acid Deposition: A Contingent Valuation Study of Minnesota. Dissertation, University of Wisconsin-Madison.

Wey, K.A. 1990. Social Welfare Analysis of Congestion and Water Quality of Great Salt Pond, Block Island, Rhode Island. Dissertation, University of Rhode Island.

Whitehead, J.C. 2006. Improving Willingness to Pay Estimates for Quality Improvements through Joint Estimation with Quality Perceptions. Southern Economic Journal 73(1): 100-111.

Whitehead, J.C., and P.A. Groothuis. 1992. Economic Benefits of Improved Water Quality: a case study of North Carolina's Tar-Pamlico River. Rivers 3: 170-178.

Whitehead, J.C., G.C. Blomquist, T.J. Hoban and W.B. Clifford. 1995. Assessing the Validity and Reliability of Contingent Values: A Comparison of On-Site Users, Off-Site Users, and Nonusers. Journal of Environmental Economics and Management 29: 238-251.

Whitehead, J.C., T. Hoban and W. Clifford. 2002. Landowners' Willingness to Pay for Water Quality Improvements: Jointly Estimating Contingent Valuation and Behavior with Limited Information. White paper developed in part by U.S. EPA, NCDENR and the College of Agriculture and Life Sciences at NSCU.

Whitehead, J.C., T. Hoban and W. Clifford. 2002. Landowners' Willingness to Pay for Water Quality Improvements: Jointly Estimating Contingent Valuation and Behavior with Limited Information. White paper developed in part by U.S. EPA, NCDENR and the College of Agriculture and Life Sciences at NSCU.

Whittington, D., G. Cassidy, D. Amaral, E. McClelland, H. Wang and C. Poulos. 1994. The Economic Value of Improving the Environmental Quality of Galveston Bay. Department of

Environmental Sciences and Engineering, University of North Carolina at Chapel Hill. GBNEP-38, 6/94.

Table 1. Primary Studies in the Metadata

Author(s) and Publication Year	Obs. in Metadata	State(s)	Water Body Type(s)	Willingness to Pay (per household/year, 2007 USD)		
				Mean	Min.	Max.
Aiken (1985)	1	CO	river and lake	193.18	193.18	193.18
Anderson and Edwards (1986)	1	RI	salt pond/marshes	180.71	180.71	180.71
Banzhaf et al. (2006)	2	NY	lake	57.47	54.09	60.85
Banzhaf et al. (2011)	1	VA, WV, TN, NC, GA	river/stream	31.30	31.30	31.30
Bockstael et al (1988)	1	DC, MD, VA	estuary	149.03	149.03	149.03
Bockstael et al. (1989)	2	MD	estuary	158.30	75.67	240.93
Borisova et al. (2008)	3	WV, VA	river/stream	44.94	18.05	65.82
Cameron and Huppert (1989)	1	CA	estuary	49.53	49.53	49.53
Carson et al. (1994)	2	CA	estuary	59.40	41.21	77.59
Clonts and Malone (1990)	3	AL	river/stream	103.20	78.31	127.48
Collins and Rosenberger (2007)	1	WV	river/stream	18.19	18.19	18.19
Collins et al. (2009)	7	WV	river/stream	120.52	2.84	217.57
Corrigan et al. (2009)	1	IA	lake	123.30	123.30	123.30
Croke et al. (1987)	9	IL	river/stream	77.47	61.31	93.68
De Zoysa (1995)	1	OH	river/stream	70.18	70.18	70.18
Desvousges et al. (1987)	12	PA	river/stream	59.19	19.84	137.26
Downstream Strategies (2008)	2	PA	river/stream	12.74	10.70	14.77

Farber and Griner (2000)	6	PA	river/stream	76.16	16.58	148.59
Hayes et al. (1992)	2	RI	estuary	397.44	390.68	404.19
Herriges et al. (1996)	2	IA	lake	134.55	61.71	207.40
Hite (2002)	2	MS	river/stream	60.08	58.24	61.93
Huang et al. (1997)	2	NC	estuary	258.65	255.01	262.29
Irvin et al. (2007)	4	OH	all_freshwater	21.67	19.65	23.23
Johnston et al. (1999)	1	RI	river/stream	180.95	180.95	180.95
Kaoru (1993)	1	MA	salt pond/marshes	218.61	218.61	218.61
Lant and Roberts (1990)	3	IA, IL	river/stream	143.93	124.04	154.31
Lant and Tobin (1989)	9	IA, IL	river/stream	55.63	40.58	67.64
Lichtkoppler and Blaine (1999)	1	OH	river and lake	41.93	41.93	41.93
Lindsey (1994)	8	MD	estuary	66.80	33.40	102.20
Lipton (2004)	1	MD	estuary	63.98	63.98	63.98
Londoño Cadavid and Ando (2013)	2	IL	river/stream	38.68	35.93	41.44
Loomis (1996)	1	WA	river/stream	93.07	93.07	93.07
Lyke (1993)	2	WI	river and lake	78.75	59.75	97.74
Matthews et al. (1999)	2	MN	river/stream	21.73	18.14	25.32
Olsen et al. (1991)	3	ID, MT, OR, WA	river/stream	88.56	44.25	123.73
Opaluch et al. (1988)	1	NY	estuary	138.47	138.47	138.47
Roberts and Leitch (1997)	1	MN, SD	lake	8.35	8.35	8.35
Rowe et al. (1985)	1	CO	river/stream	134.59	134.59	134.59
Sanders et al. (1990)	4	CO	river/stream	160.69	81.01	210.04

Schulze et al. (1995)	2	MT	river/stream	20.84	17.34	24.33
Shrestha and Alavalapati (2004)	2	FL	river and lake	156.46	137.97	174.95
Stumborg et al. (2001)	2	WI	lake	84.29	66.73	101.86
Sutherland and Walsh (1985)	1	MT	river and lake	146.03	146.03	146.03
Takatsuka (2004)	4	TN	river/stream	286.88	181.90	391.85
Wattage (1993)	3	IA	river/stream	53.89	40.24	74.59
Welle (1986)	6	MN	lake	167.28	109.60	238.42
Welle and Hodgson (2011)	3	MN	lake	145.10	10.59	285.06
Wey (1990)	2	RI	salt pond/marshes	147.26	63.95	230.58
Whitehead and Groothuis (1992)	3	NC	river/stream	41.01	31.90	53.16
Whitehead (2006)	3	NC	river/stream	187.18	27.52	365.54
Whitehead et al. (1995)	2	NC	estuary	95.44	78.29	112.59
Whittington et al. (1994)	1	TX	estuary	194.72	194.72	194.72

Table 2. Meta-Analysis Variables and Descriptive Statistics

Variable	Definition	Mean (std. dev.)
ce	Binary (dummy) variable with a value of one for studies that are choice experiments (default is any non-choice experiment method)	0.105 (0.308)
thesis	Binary (dummy) variable with a value of one for studies developed as thesis projects or dissertations (default is studies not developed as theses)	0.112 (0.316)
lnyear	Natural log of the year in which the study was conducted (converted to an index by subtracting 1980, before making the log transformation)	2.212 (0.918)
volunt	Binary (dummy) variable indicating that WTP was estimated using a payment vehicle described as voluntary (default is a binding and mandatory payment vehicle, e.g., property taxes)	0.084 (0.278)
outlier_bids	Binary (dummy) variable indicating that outlier bids were excluded when estimating WTP (default is studies that did not exclude outlier bids)	0189 (0.393)
nonparam	Binary (dummy) variable indicating that WTP was estimated using non-parametric methods (default is studies using parametric methods).	0.441 (0.498)
non_reviewed	Binary (dummy) variable indicating that the study was not published in a peer-reviewed journal (default is studies published in peer reviewed journals)	0.231 (0.423)
lump_sum	Binary (dummy) variable indicating that payments were to occur on something other than an annual basis over an extended or indefinite period of time (default is payments in a lump sum or over a short period of time, e.g., less than 5 years)	0.182 (0.387)
wtp_median	Binary (dummy) variable indicating that the study's WTP measure is the median (default is mean WTP)	0.070 (0.256)
mp	Binary (dummy) variable indicating that the survey included respondents from the USDA Mountain Plains region (default is respondents from the northeast, west or multiple regions)	0.105 (0.308)
ma	Binary (dummy) variable indicating that the survey included respondents from the USDA Mid-Atlantic region (default is respondents from the northeast, west or multiple regions)	0.301 (0.460)
se	Binary (dummy) variable indicating that the survey included respondents from the USDA Souteast region (default is respondents from the northeast, west or multiple regions)	0.147 (0.355)
sw	Binary (dummy) variable indicating that the survey included respondents from the USDA Southwest region (default is respondents from the northeast, west or multiple regions)	0.007 (0.084)
mw	Binary (dummy) variable indicating that the survey included respondents from the USDA Midwest region	0.224 (0.418)

	(default is respondents from the northeast, west or multiple regions)	
nonusers	Binary (dummy) variable indicating that the survey was implemented over a population of nonusers (default is a survey of any population that includes users)	0.098 (0.298)
lnincome	Natural log of median income (in 2007$) for the sample area of each study based on historical U.S. Census data. To ensure comparability this variable was estimated for all studies in the metadata regardless of whether the study reported income for the sample	10.745 (0.171)
mult_bod	Binary (dummy) variable that takes on a value of 1 if the studied system includes multiple water body types (e.g., lakes and rivers), and zero otherwise	0.077 (0.267)
river	A binary (dummy) variable that takes on a value of 1 if the studied system includes rivers, and zero otherwise	0.692 (0.463)
swim_use	Binary (dummy) variable identifying studies in which changes in swimming uses are specifically noted in the survey (default is surveys that do not describe effects on swimming)	0.259 (0.440)
gamefish	Binary (dummy) variable identifying studies in which changes in game fishing uses are specifically noted in the survey (default is surveys that do not describe effects on game fishing)	0.056 (0.231)
boat_use	Binary (dummy) variable identifying studies in which changes in boating uses are specifically noted in the survey (default is surveys that do not describe effects on boating)	0.112 (0.316)
ln_ar_agr	Natural log of the proportion of the affected resource area which is agricultural based on NLCD. Affected resource area includes all counties that intersect the affected resource	-1.437 (0.894)
ln_rel_size	An index of the size of the affected water body (defined by total shoreline length, in kilometers), relative to the size of the sampled area, in square kilometers. For a bay, shoreline length is given by the variable *bay_len*. For a lake, shoreline is approximated by lake circumference, *lake_circ*. For a river, shoreline (assuming a left and right shoreline) is given by the variable *river_length**2. The total affected shoreline for any study is defined as *shoreline = riv_len*2 + lake_circ + bay_len*. From here, *ln_rel_size = log(shoreline / sa_area)*, where *sa_area* is the size of sampled area in square kilometers	-1.292 (3.446)
sub_frac	The proportion of water bodies of the same hydrological type affected by the water quality change, within affected state(s). For rivers, this is measured as the length of the affected river reaches as a proportion of all reaches of the same order or lower (*sub_frac_reach*). For lakes and ponds, this is defined as the area of the affected water body as a proportion of all water bodies of the same National Hydrography Dataset classification (*sub_frac_area*). For bays and estuaries, this is defined	0.188 (0.288)

	as the shoreline length of the water body as a proportion of all analogous (e.g., coastal) shoreline lengths (*sub_frac_bay*). The variable *sub_frac* is defined as max(*sub_frac_reach*, *sub_frac_area*, *sub_frac_bay*).	
riv_flow	The quantity of water flowing through affected rivers, in cubic feet per second, based on data from the National Hydrography Dataset.	4328.008 (21870.860)
lnquality_ch	Natural log of the change in mean water quality valued by the study, specified on the 100-point water quality index (McClelland 1974; Mitchell and Carson 1989); see main text for details.	2.909 (0.598)
lnbase	Natural log of the baseline (status quo) water quality from which improvements would occur, specified on the 100-point water quality index.	3.596 (0.664)

Table 3. Meta-Regression Model Results (ln(WTP); Random Effects Model, Robust Standard Errors)

Variable	Unrestricted Model Coefficient Estimates (Standard Errors)	Restricted Model [a] Coefficient Estimates (Standard Errors)
Methodological Variables		
ce	0.538 (0.227)**	0.504 (0.234)**
thesis	0.717 (0.262)***	0.969 (0.274)***
lnyear	-0.458 (0.094)***	-0.433 (0.122)***
volunt	-1.261 (0.233)***	-1.006 (0.221)***
outlier_bids	-0.395 (0.139)***	-0.437 (0.166)***
nonparam	-0.404 (0.148)***	-0.273 (0.145)*
non_reviewed	-0.817 (0.196)***	-0.886 (0.212)***
lump_sum	0.751 (0.166)***	0.476 (0.172)***
wtp_median	-0.266 (0.256)	-0.227 (0.240)
Region and Surveyed Populations		
mp	0.341 (0.193)*	0.183 (0.225)
ma	-0.383 (0.215)*	-0.367 (0.232)
se	1.092 (0.203)***	1.002 (0.251)***
sw	1.251 (0.299)***	1.466 (0.355)***
mw	0.276 (0.220)	-0.037 (0.263)
nonusers	-0.455 (0.115)***	-0.436 (0.119)***
lnincome	0.774 (0.404)*	0.936 (0.439)**
Extent of the Market, Study Site and Affected Resources		
mult_bod	-0.561 (0.171)***	-0.198 (0.163)

river	-0.254	-0.299
	(0.153)*	(0.174)*
swim_use	-0.359	-0.297
	(0.251)	(0.278)
gamefish	0.224	0.260
	(0.208)	(0.203)
boat_use	-0.279	-0.207
	(0.172)	(0.149)
ln_ar_agr	-0.402	-0.262
	(0.097)***	(0.105)**
ln_rel_size	0.050	--
	(0.022)**	--
sub_frac	0.545	--
	(0.218)**	--
riv_flow	4.28e-06	--
	(1.66e-06)**	--
Water Quality Baseline and Change		
lnquality_ch	0.284	0.253
	(0.107)***	(0.118)**
lnbase	-0.078	-0.140
	(0.153)	(0.151)
Model Intercept		
constant	-3.647	-4.824
	(4.666)	(5.062)
N	143	143
Num. Groups	52	52
R^2	0.63	0.58
Wald χ^2 *(df)*	8112.91	1075.88
	(27)	(24)
Prob. > χ^2	0.0001	0.0001
σ_u	0.155	0.263
σ_e	0.536	0.536

[a] Restricted model omits variables characterizing geospatial scale, market extent and substitute availability.

Table 4. Illustrative Scenarios: Implications of Geospatial Scale, Extent of the Market and Relative Substitutes on Benefit Transfers

Variable	Assigned Variable Values		
	Scenario 1 (Mean of Sensitivity Analysis Variables)	Scenario 2 (Min. of Sensitivity Analysis Variables)	Scenario 3 (Max. of Sensitivity Analysis Variables)
ce [a]	0.105	0.105	0.105
thesis [a]	0.112	0.112	0.112
lnyear [a]	2.212	2.212	2.212
volunt [b]	0.000	0.000	0.000
outlier_bids [a]	0.189	0.189	0.189
nonparam [a]	0.441	0.441	0.441
non_reviewed [a]	0.231	0.231	0.231
lump_sum [b]	0.000	0.000	0.000
wtp_median [b]	0.000	0.000	0.000
mp [b]	0.000	0.000	0.000
ma [b]	1.000	1.000	1.000
se [b]	0.000	0.000	0.000
sw [b]	0.000	0.000	0.000
mw [b]	0.000	0.000	0.000
nonusers [b]	0.000	0.000	0.000
lnincome [a]	10.745	10.745	10.745
mult_bod [b]	0.000	0.000	0.000
river [b]	1.000	1.000	1.000
swim_use [a]	0.259	0.259	0.259
gamefish [a]	0.056	0.056	0.056
boat_use [a]	0.112	0.112	0.112
ln_ar_agr [a]	-1.437	-1.437	-1.437
lnbase [a]	3.596	3.596	3.596
lnquality_ch [a]	2.909	2.909	2.909
constant	1.000	1.000	1.000
Sensitivity Analysis Variables: Geospatial Scale, Extent of the Market and Substitutes			
ln_rel_size	-1.292	-11.135	7.811
sub_frac	0.188	0.0003	1.000
riv_flow	4328.008	17.6976	152198.700
WTP Estimate: Unrestricted Model [c]	$50.76	$27.49	$234.50
WTP Estimate: Restricted Model [c]	$56.31	$56.31	$56.31

[a] Variable assigned a value equal to the variable mean for the metadata.

[b] Binary (0,1) variable assigned a 0 or 1 value to ensure interpretability of resulting welfare projections.

[c] Resulting welfare projection applies to annual mean WTP per household (*lump_sum*=0; *wtp_median*=0; *volunt*=0), over a general population sample (*nonusers*=0) in the mid-Atlantic region (*ma*=1; *mp*=*se*=*sw*=*mw*=0),

for a water quality improvement in a single river (*river*=1; *mult_bod*=0). The assumed water quality improvement (*lnquality_ch*=2.909) is equal to the mean over the metadata; this is equivalent to a change of 18.335 = $e^{2.909}$ on the 100-point WQI (see main text), beginning from a baseline of *lnbase*=36.440. Restricted model omits variables characterizing geospatial scale, market extent and substitute availability.