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Chapter 2

Ecosystem Service Benefits Generated by Improved Water Quality from Conservation Practices

The Council on Food, Agricultural & Resource Economics

Authors

Synthesis Chapter - The Valuation of Ecosystem Services from Farms and Forests: Informing a systematic approach to quantifying benefits of conservation programs

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Chapter 2: Ecosystem Service Benefits Generated by Improved Water Quality from Conservation Practices

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Please note, the report and its chapters are intended to demonstrate a framework approach to ecosystem service valuation. The report or any chapter there within is not to be cited for the purpose of supporting or opposing any government or private program.

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Abstract

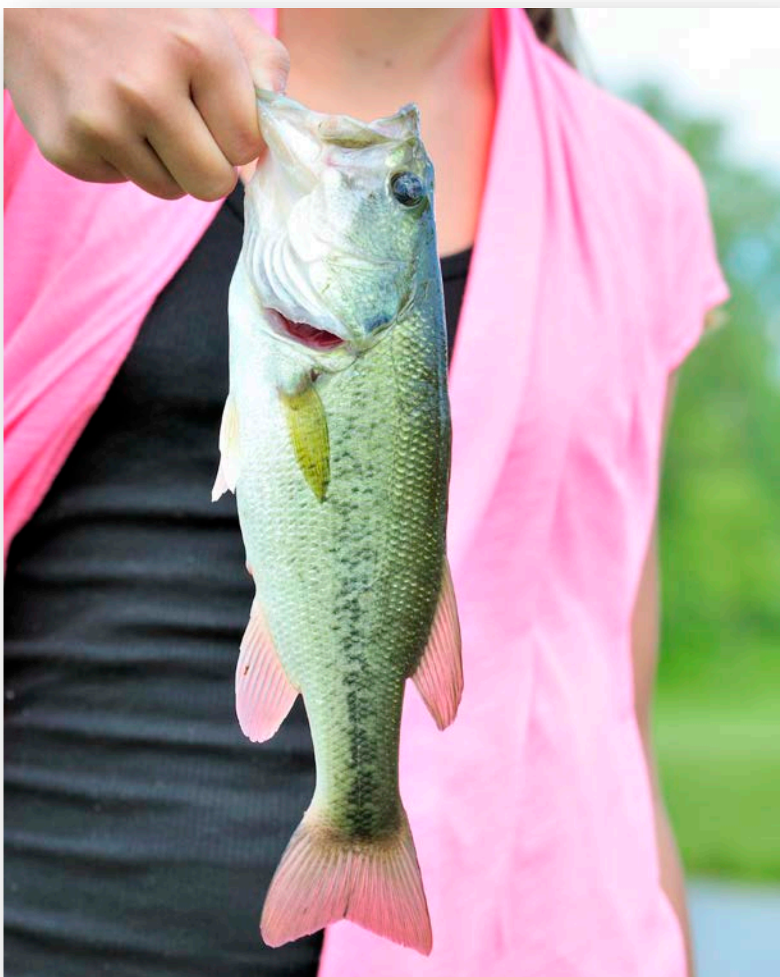
This report presents results of an interdisciplinary team who evaluated the state of the science for valuing ecosystem services derived from implementing conservation practices on agricultural and forest lands. A case study of conservation practices on farms within the Western Lake Erie Basin was used to illustrate methods for estimating monetary values and non-monetary benefit indicators for five ecosystem service benefits generated by improved water quality: 1) enhanced property values; 2) improved sport fishing; 3) improved aquatic community condition (non-use value); 4) improvements to drinking water supply and other uses of reservoirs (reduced sedimentation); and 5) avoided operation and maintenance costs in commercial navigation (reduced sedimentation). The team found that these benefits were all monetizable using benefit transfer approaches and available data and models. However, many values were likely to have low precision due to assumptions needed to overcome data gaps. Data gaps limited the ability to connect changes in nutrient loads to changes in some of the in-situ water bodies conditions needed for valuation. Further, only one primary valuation study was available for transfer for two of the ecosystem service benefits. In contrast, the recreational fishing benefits were estimated using a benefit transfer function developed from national data, thereby exemplifying approaches that could increase precision of value estimates. The case study revealed that valuation with benefit transfer methods is only possible when ecological modeling outputs align with variables used in valuation and that additional primary valuation work would improve the ability to adjust values to local conditions.

Introduction

The Natural Resources Conservation Service (NRCS) funded $84.8 million multi-purpose conservation activities on private land conservation to improve water quality (U.S. Government Accountability Office, 2016). Multiple programs provide technical and financial assistance to private landowners to improve natural resource conditions that enhance agricultural production and other ecosystem services. These enhancements, in turn, generate many off-farm societal benefits brought about by improvements in water quality, water supply, air quality, and wildlife habitat, among other changes (e.g., Swinton et al. 2007). The benefits of conservation programs are potentially substantial, but they are not often identified, much less quantified or valued in full.

This report evaluates a sub-set of benefits that result from improved nutrient and sediment management on farms and associated improvements in water quality. Such benefits can be difficult to isolate because changes in agricultural practices are occurring against a backdrop of land use change, shifts in crop and animal production systems, climate variability, air quality change, and economic trends. Yet, advances in monitoring and modeling are improving the potential to evaluate how ecosystem services are affected by conservation actions.

The aim of this effort is to demonstrate a replicable process for estimating the ecosystem service benefits of water quality improvements derived from adoption of on-farm conservation practices. The information provided is primarily intended to be illustrative of potential methods rather than a precise estimation of benefits. The methods serve to demonstrate the potential of using ecosystem services to capture a broad suite of public benefits with available data and tools.[1] In addition to demonstrating methods, this effort highlights gaps in data or understanding that limit valuation. Because ecosystem service benefits depend on local biophysical and socio-economic conditions, this report uses the Western Lake Erie Basin as a case study site to leverage recently completed studies that provide the type of scientific input needed to support economic analyses.



[1] Private on-farm benefits of conservation practices are also likely to be significant, but they were beyond the scope of this analysis.

METHODS

Conceptual value diagram of ecosystem service benefits

A multi-disciplinary team of federal and academic scientists was assembled by the Council on Food, Agriculture, and Resource Economics (C-FARE) to conduct the ecosystem service valuation using established methods for ecosystem service analysis and valuation of benefits. The team began the analytic process by creating a conceptual value diagram of the expected relationships among conservation practice implementation, water quality, and other biophysical changes resulting from implementation, and ecosystem service benefits (or disadvantages) derived from ecosystem change (Figure 1).

The team defined ecosystem services benefits as outcomes that were directly used or appreciated by people, thereby enabling rigorous analysis using monetary valuation or non-monetary benefit indicators (for outcomes that could not be valued in monetary units). Using so-called "final ecosystem services" as the basis for analysis (Boyd and Banzhaff 2007, Johnston and Russell 2011) ensures that beneficiaries are identified and that changes are evaluated in terms of enhanced well-being of those beneficiaries. In contrast, measuring ecosystem services as a change in a biophysical quantity, such as phosphorus loads, does not specify who benefits, by how much, or why.

The list of ecosystem service benefits includes use and non-use (or passive use) values to reflect the full range of values that people derive from ecosystem enhancements (Freeman et al. 2014). Use values are the tangible benefits that people derive from using (consumptive and non-consumptive uses) ecosystems or being near them. One example is anglers, who benefit from a greater abundance or diversity of fish species. Non-use and passive use values represent the (intangible) satisfaction that people derive from being good stewards of the environment and making it available to other users (altruistic values) or future generations, as distinct from any expected use of the resource (Krutilla 1967). For example, people may value knowing that habitat extent for a rare fish has been increased, even if they will not be able to see, catch, or otherwise benefit from that fish population.

Figure 1 provides a conceptual value diagram of how conservation practices, on the left side of the diagram, can generate public ecosystem service benefits derived from water quality improvements, shown on the right. Actions, in this case conservation practices, lead to changes in nutrient and sediment movement from the edge of the field. Edge-of-field reductions limit loads to streams, rivers, ponds, lakes, and estuaries and generate changes in water bodies. The fourth box from the left (ecological outcomes) lists common biophysical measures of water body impacts. The fifth column shows benefit relevant indicators, which may reflect the magnitude or intensity of effects on beneficiaries and can be an intermediate step to estimating final ecosystem service benefits shown in the right-hand column.[2] The arrows between the boxes represent data or models that would be used to estimate the relationship between inputs and outcomes.

[2] The relationship between the magnitude of benefit relevant indicators and the magnitude of values is often expected to be positive, but is not necessarily so over all ranges of possible impacts. For example, some currently rare species could possibly become a nuisance at high abundance levels. More generally, the relationship between benefit relevant indicators and values is often non-linear, e.g., such that a 10 percent increase in an indicator does not necessarily imply a 10 percent increase in ecosystem service value. Non-linearities such as this are among the reasons that it is important to measure economic values in addition to benefit relevant indicators.

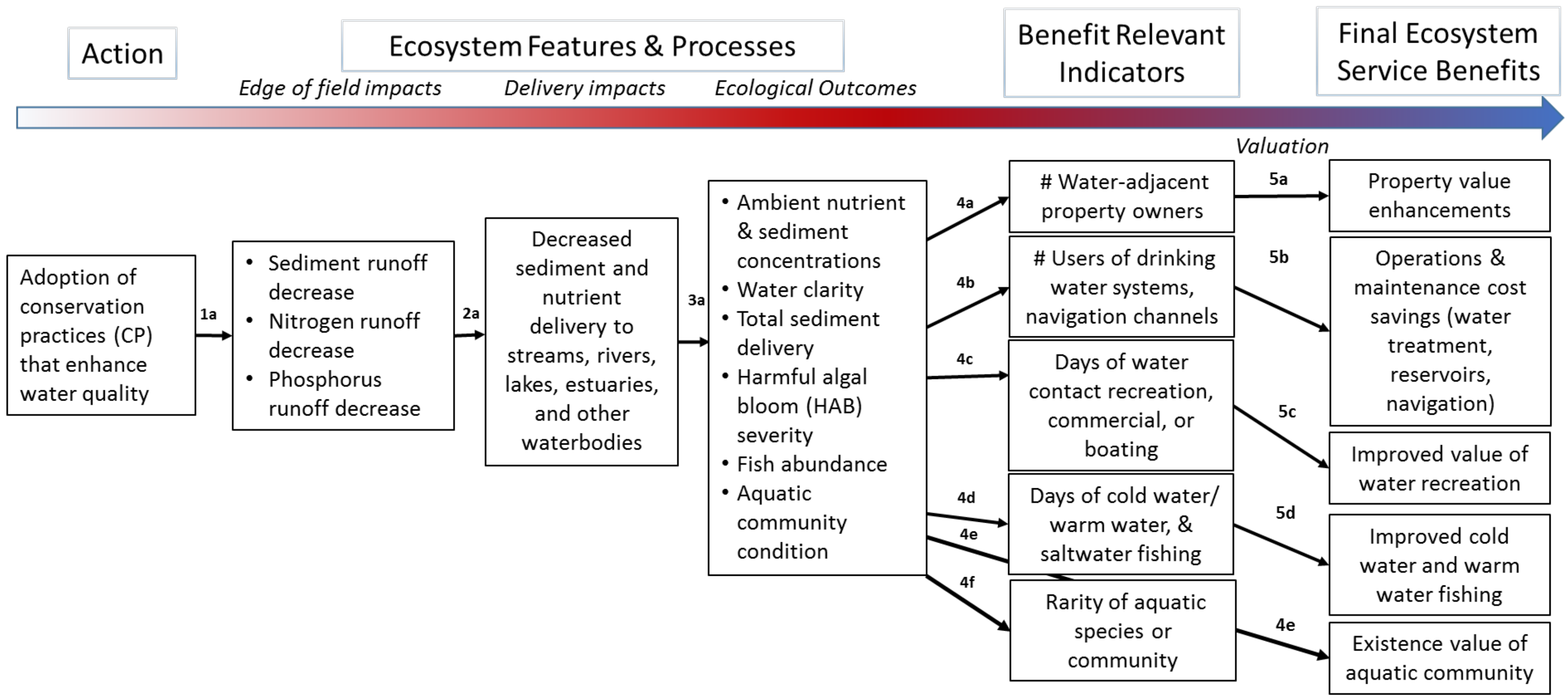


Figure 1. Conceptual value diagram of public ecosystem service values from conservation practices that influence water quality

Describes the cause and effect relationships between conservation practices (on the left) and public good/service benefits generated (on the right). The middle of the diagram shows the ecological changes that determine the quantity and quality of ecosystem services.

Selecting Appropriate Scale of Analysis

A first step of the ecosystem service analysis was to consider the spatial scale at which benefits could be appropriately assessed. Although the team recognized that a national scale was desirable for evaluating national USDA programs, a lack of consistent data at this scale and the location-specific nature of many benefits led the team to choose a case study within the Great Lakes Restoration region (Figure 2). The purpose of using a regional case-study was to demonstrate rigorous methods of quantifying ecosystem service benefits when sufficient data and resources are available. Not all areas will have such rich data sources, and data may be inconsistent at the national scale, but this case study demonstrates the types of data needed for robust economic assessments.

Case Study Overview

The primary water quality problem in the Western Lake Erie Basin is excess nutrient delivery to Lake Erie, especially phosphorous. Impaired water quality in Lake Erie and its tributaries has led to excessive and harmful algal production and reductions in quality of habitat for fish and other aquatic communities (Michelak et al. 2013, Kraus et al. 2015). Water quality, in turn, may impact production of multiple ecosystem services that society values, including safe drinking water, support of property values, commercial and recreational fishing opportunities, other types of outdoor recreation, and existence (non-use) values, among others (Loomis et al. 2000; Dodds et al. 2009; Scavia et al. 2014).

The Western Lake Erie Basin has rich data sources because of the intensive restoration and scientific analysis efforts currently underway. Several USDA conservation programs support voluntary adoption of conservation practices in the Western Lake Erie Basin, including initiatives (e.g., Great Lakes Restoration Initiative, NRCS Western Lake Erie Basin Initiative) and partnership programs (e.g., Regional Conservation Partnership Program). The educational, technical, and financial assistance support conservation work through and in conjunction with the authorized Farm Bill programs, (Environmental Quality Incentives Program (EQIP), Conservation Stewardship Program (CSP), Agricultural Conservation Easement Program (ACEP), Healthy Forests Reserve Program (HFRP), Conservation Reserve Program (CRP) and Conservation Reserve Enhancement Program (CREP), which provide conservation options for land owners and managers in the Basin.

Data to support the case study analysis were derived from the Western Lake Erie Basin's Conservation Effects Assessment Project (CEAP) Croplands study (USDA 2016a) and associated CEAP Wildlife and CEAP Watersheds documents as well as related work that monitor or model ecological effects (Richards and Baker 2002, Richards et al. 2005, Richards et al. 2009, Ohio EPA 2014, Miltner 2015, Keitzer et al. 2016a). Survey data within the CEAP Cropland report showed that between 2003–06 and 2012, agricultural producers in the Western Lake Erie Basin significantly increased their use of conservation measures to improve and protect water and soil quality (USDA NRCS 2016a). This report provided two types of results: 1) survey-based implementation results documenting changes between a baseline dataset collected from 2003–2006 and in a 2012 survey; and 2) model estimates of changes in nutrient and sediment losses at the edges of fields and associated ecological changes that could be expected from implementing conservation practices on vulnerable[3] agricultural lands. A variety of ecological data and models were created to estimate outcomes using field data from the National Resources Inventory and NASS CEAP Survey, the Agricultural Policy/Environmental eXtender (APEX) model, soil survey data, and weather data.

[3] Vulnerability was defined as the likelihood of erosion and nutrient runoff based on slopes, soils, and crop types.

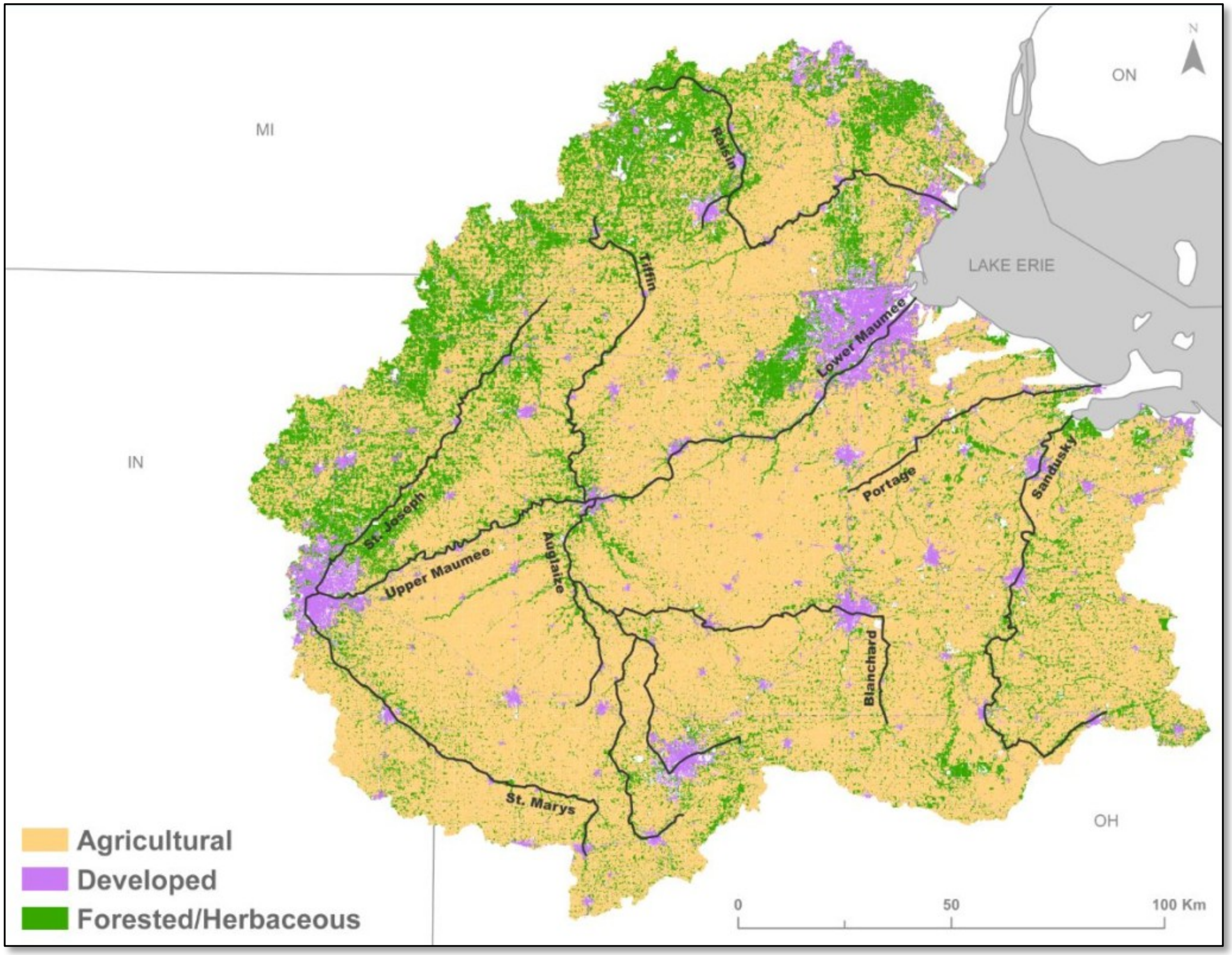


Figure 2. Western Lake Erie Basin Watershed land use.

In the basin, 70 percent of the land is used for agricultural purposes, while developed areas like cities cover about 12 percent of the watershed. Forested land covers another 12 percent of the area, scattered in fragmented woodlots or along the riparian areas of some streams and rivers. Map source: Maura O'Brien and Conor Keitzer (The Ohio State University). Data source: USDA National Agricultural Statistics Service Cropland Data Layer, 2014.

PROJECTING CHANGES RELATIVE TO BASELINE

Economic analyses to demonstrate the benefits of any action compare changes due to an action relative to a baseline future without the action (Freeman et al., 2014). In this case study, the baseline was the documented level of conservation practices observed in the 2003–06 CEAP Assessment (USDA NRCS 2011). Soils, weather, crop rotations, and the level of practice implementation were assumed to remain the same into the future for analyses that examined a future stream of benefits.

Three different scenarios were used to estimate changes in conservation practice implementation that varied by ecosystem service benefit. In Scenarios 1 and 2 in, acres vulnerable to erosion and nutrient runoff were assumed to be treated with practices that would bring them below nutrient and sediment loss thresholds (Keitzer et al. 2016a and 2016b). The CEAP Cropland study (USDA NRCS 2011) estimated that treatment of the "high need acres" (or acres with high vulnerability) in the Western Lake Erie Basin would reduce edge of field source loads for sediment by over 20 percent (USDA NRCS 2011, Table 56) and nutrients by six to eight percent (USDA 2011, Table 59 and Table 62). The CEAP Cropland assessment estimated that treatment of the "high and moderate need acres" in the WLEB would reduce edge of field source loads for sediment by over 50 percent (USDA NRCS 2011, Table 56) and nutrients by 30–35 percent (USDA NRCS 2011, Table 59 and Table 62) on the acres treated.

Scenario 3 defined the action as all new (net) practices that were shown to be implemented in the 2012 CEAP survey relative to practices implemented at the time of the baseline survey in 2003–06. This scenario includes changes in total acreage of farmland over this period and accounts for a different area of implementation of a large list of specific conservation practices (USDA NRCS 2011 and 2016a). For 2012, the model predicted that added conservation practices had reduced sediment losses from cropland by 50 percent relative to the baseline condition (Scenario 3, Table 2). Sediment loss from the edge of field decreased from 1.1 to 0.5 tons per acre per year. The model also predicted that by 2012, particulate phosphorus runoff was reduced by 17 percent and nitrogen losses in runoff decreased by 35 percent relative to the baseline condition (USDA NRCS 2016a). It was necessary to use different future scenarios when analyzing different ecosystem services because of 1) different choices made by ecological modelers in the CEAP program, or 2) a need to match available data to economic model input needs.

Table 1. Scenarios used to assess effects of changes in conservation practices

Service	Scenario 1. Conservation practices implemented on acres with *high* conservation need (8% of cropland)	Scenario 2. Conservation practice implemented on acres with *high* and *moderate* conservation need (48% of cropland)	Scenario 3. Difference between the 2003–06 and 2012 CEAP farmer surveys (21% increase in acres adopting for the practices evaluated)
Property enhancement	√	√	
Aquatic community		√	
Sport fishing		√	
Reservoir sediment reduction			√
Navigation sediment reduction			√

Estimates of Practice Impacts on Sediment and Nutrient Delivery to Waterbodies

Two models, APEX (v1307)[4] and HUMUS/SWAT, were used to estimate effects of conservation practice adoption on nutrient and sediment delivery to water bodies (USDA NRCS 2016a) (Figure 1, arrows 1a and 2a). The Agricultural Policy Environmental eXtender (APEX), is a physical, process-based model that provides edge of field impacts (Figure 1, arrow 1a). It is used to simulate long-term effects of conservation practice adoption at the field scale (Williams et al. 2006, Williams et al. 2008, Gassman et al. 2009) using interactions between weather, farming operations, crop growth and yield, and the movement of water, soil, carbon, nutrients, sediment, and pesticides. APEX and its predecessor, the Environmental Policy Impact Calculator (EPIC), have a long history of use in simulation of agricultural and environmental processes and the effect of agricultural technology and government policy on natural resources (Izaurralde et al. 2006, Williams 1990, Williams et al. 1984, Gassman et al. 2009).

The watershed model, SWAT, was used to estimate changes in nutrients and sediment delivered to water bodies (Figure 1, arrow 2a). SWAT was used in conjunction with HUMUS, which manages and analyzes input data needed to run and calibrate SWAT. A detailed discussion of these models can be found at (Arnold et al. 2010), and results that were used in analysis are described in the relevant sections.

The APEX and SWAT modeling suggested that increases in conservation practice implementation had improved soil condition and reduced sediment and nutrient deliveries to water bodies (USDA 2016a).

Estimating changes in secondary biophysical effects: Water quality conditions of waterbodies

The APEX and SWAT models do not directly provide water quality characteristics of the affected water bodies as needed to link to economic models either directly or through intermediate ecological models (Figure 1, arrow 3a). This gap was partially filled by ecological modelers within the CEAP program, who developed methods of assessing instream water quality conditions (e.g., Keitzer et al. 2016a) for the purposes of aquatic ecosystem modeling. However, those models did not include

[4] This version has been shown to more accurately simulate the mitigating effects of buffers, filters, and drainage water management on edge-of-field losses than previous versions.

changes in Lake Erie water quality, thereby limiting the geographic scope of the economic analysis to the streams that had been modeled.

Models that connect changes in edge of field losses to changes in water quality within water bodies are critical to developing economic models to value ecosystem service changes. Although it could not be applied in this case study, a primary modeling resource to make such links is SPARROW (SPAtially Referenced Regression on Watershed attributes), which uses monitoring data to relate watershed conditions to instream water quality loads and is available by region within the continental U.S. (Smith et al. 1997; Schwarz et al. 2006). SPARROW has not historically been able to project water quality changes due to changes in conservation practices, however, SPARROW modelers are working to address this limitation. A recent application (Garcia et al. 2016) used SPARROW, in conjunction with USDA Conservation Effects Assessment Project (CEAP) results, to empirically estimate total nitrogen and phosphorus loads in rivers and streams of the Upper Mississippi River Basin, suggesting that SPARROW is likely to be a critical future resource for economic assessments of regional and national USDA programs. (See Appendix A, for further information.)

Connecting water quality outcomes to economic values

To proceed with the valuation of changes in ecosystem services caused by changes in water quality, analysts must link the available biophysical endpoints (changes in delivered loads of nutrients and sediments or water quality conditions) to changes in outcomes that people care about (recreation, habitat). Such connections can be derived from either complex or simple modeling approaches. Complexity arises when a series of interlinked sub-models are used to represent the important processes and socio-economic conditions that determine value. For example, when modeling the sport fishing ecosystem service, edge of field loads are evaluated for their contributions to instream conditions (e.g., water clarity). Next, instream conditions are linked to an outcome of direct relevance to beneficiaries (e.g., changes in sport fish abundance). Finally, the number of beneficiaries and their willingness to pay for a specific change is evaluated using spatial demographic data and economic models (e.g., willingness to pay for higher catch rates). An advantage of linked models, such as these, is an ability to explicitly model the dynamics through which changes in human actions (e.g., USDA conservation program activities) lead to changes in ecosystems and ultimately to changes in human value. However, linked models such as these cannot be developed when data and analysis costs are prohibitive or when the complexity of aquatic ecosystem interactions and their natural variability prevent estimates of relevant outcomes (e.g., Breitburg et al. 2009).

As an alternative to more complex models, analysts often use observed empirical relationships between ecological changes and values to simplify the many underlying relationships. For example, multiple efforts have developed a single water quality index (WQI) that combines information on multiple physical and chemical water quality parameters to comprehensively represent water quality conditions (Abbasi 2012, Van Houtven et al. 2014, Terrado et al. 2010, Walsh and Milon 2015). Such indices are commonly used as a component of economic analyses that estimate total willingness to pay for water quality changes, including both use and nonuse values. For example, many stated preference studies[5] use variants of a WQI that communicate water quality changes to survey respondents, as a precursor to eliciting their willingness to pay for these changes (Johnston and Besedin 2009). However, while they have been used by many valuation studies in the past,

[5] Stated preference studies use sophisticated survey techniques to elicit values for goods and services and individuals' willingness to make tradeoffs among these services. They are the most flexible approaches to valuation and the only ones capable of measuring non-use values, in addition to use values. See Champ et al. (2012) for further information.

WQIs have several limitations, including limited ability to account for the effects of different water quality parameters (e.g., nutrients vs. temperature) on different uses and values (U.S. EPA SAB 2009). Benefit estimates can also be sensitive to the WQI that is used (Walsh and Wheeler 2013). An additional weakness of these unified, multi-metric indices is that it can be difficult to disentangle motivations for willingness to pay, such as distinguishing use and non-use values for water quality improvements (Johnston et al. 2003).

Using benefit relevant indicators to assess non-monetary benefits

The team found the resources necessary to estimate values for expected changes in multiple ecosystem services using monetary valuation and created benefit relevant indicators (BRIs) (National Ecosystem Services Partnership 2015) as intermediate steps in the monetary valuation. BRIs are metrics that are used to relate environmental changes to human well-being and through such links, indicate potential value of changes. However, BRIs do not represent willingness to pay for a change. Instead, they provide information that may inform monetary valuation or substitute for it when monetization is not appropriate or possible. For example, the number of households harmed by poor drinking water quality is a complement to the monetary value of willingness-to-pay for water quality improvements because it may be a more equitable way to compare impacts when household income levels differ.

Benefit relevant indicators can be biophysical metrics that have importance to stakeholders, socio-demographic metrics that represent the number of beneficiaries or likely intensity of their concern, or a combination of the two. For example, *the number of residential wells with nitrate levels sufficient to cause health problems* is a combination of a biophysical water quality parameter (nitrate concentrations more than a threshold) with a demographic measure (number of affected households). The Conservation Reserve Program, similarly, uses Environmental Benefit Indicators (EBIs) to suggest benefits (USDA FSA 2013). EBIs may be BRIs; however, when BRIs go beyond biophysical measures, and can represent potential magnitude or intensity of concern for a biophysical change, they are often better able to represent potential social value of a resource change. For example, a percent increase in habitat area of a threatened species is more clearly a potential benefit than a metric of increased habitat acres for a species of uncertain conservation status.

Benefit relevant indicators are particularly useful for representing social benefits when they are selected based on stakeholder interaction of some kind, such as having been valued in another setting, through primary economic valuation techniques. For example, indices of biotic integrity (IBI) for fish and invertebrate IBIs have been shown to be valued by the public through stated preference studies (Johnston et al 2011, 2016a). These indices are typically based on indicators representing the conditions of multiple aquatic species, particularly invertebrates and fish (Roth et al. 1996, Allan et al. 1997, Diana et al. 2006, US EPA 2006, USGS 2013). Additional support for using biotic condition metrics to represent social value comes from governmental or non-governmental organizations that establish conservation goals. For example, the abundance or richness of native species (e.g., Hawkins 2006) has been recognized as an important indicator of the nation’s natural ecological capital (National Research Council 2000).

When selecting a benefit relevant indicator to use for evaluating an ecosystem service, it is useful to consider whether that indicator will serve as a stepping stone to valuation. If it is meant to support an economic benefit transfer analysis (as explained in the next section), ecological studies can be designed to generate a metric that supports transfer, for example, by selecting a biophysical metric that matches those used in prior valuation studies that might be used for transfer. Similarly, if primary economic valuation is the goal, ecological outcomes can be measured in ways that are suitable for use within different types of valuation studies (Johnston et al. 2012, Schultz et al. 2012, Boyd et al. 2016).

Monetary valuation of outcomes

Monetary values of ecosystem service changes that are estimated in this report relied exclusively on benefit transfer methods, which use existing valuation studies to estimate values for unstudied sites (Johnston et al. 2015, Johnston and Rosenberger 2010, Wilson and Hoehn 2006). Benefit transfer methods can include both unit value transfers or benefit function transfers. Unit value transfers include the transfer of a single value or adjusted value; for example, the average willingness to pay per household for a change in a specified ecosystem service, either measured by a single study or a group of studies. Benefit function transfers, in contrast, calculate values using a function estimated from empirical data that allows multiple site factors, such as income levels, to be used to adjust the study site value to the policy site (Johnston et al. 2015). Generally, benefit function transfers rely on functions estimated by primary studies in the published literature, although these functions may also be estimated via a meta-analysis of results from many prior primary studies (Johnston et al. 2015).

The choice to use benefit transfer does not reflect a recommendation of this paper, but rather was a practical concession to the time and resource constraints of this project and expected limitations on the time and resources available to federal agencies. In general, high-quality primary valuation studies are considered to provide the most robust and accurate estimates of value of an environmental change. However, time, funding, data, and other constraints often preclude the use of primary studies for ecosystem service valuation across multiple sites of interest. However, benefit transfer can be conducted in ways that reduce error, and can be a feasible alternative to primary study valuation in many instances (Johnston et al. 2015).

To conduct benefit transfer, the analyst should consider several steps that build on the analyses described to this point (adapted from Johnston and Wainger 2015).

1. Identify appropriate literature for conducting unit value or functional benefit transfer of the relevant BRI.
2. Evaluate whether socio-demographics, ecosystem type, and aquatic ecosystem baselines and degree of change are of sufficient similarity to the context of the available sites to justify benefit transfer.
3. Either extract an appropriate unit value from available studies (as used in non-use value for aquatic community condition) or apply a benefit function transfer (as used in the sport fishing section below). For unit value transfer, adjust study values to current values using an appropriate consumer price index.
4. Multiply the resulting estimate of unit value willingness to pay (WTP) (per year) by the population that is assumed to hold values for the service, to generate total WTP per year for this service.
5. If desired, aggregate over different policy sites and/or time periods. Aggregation over time periods requires appropriate discounting to allow future benefits or costs to be represented as present values (for further explanation, see Arrow et al. 2013 and Cropper 2013).
6. Conduct sensitivity analysis to assumptions in the benefit transfer (e.g., populations assumed to hold values, the magnitude and extent of IBI change, etc.).
7. Evaluate and report sources of uncertainty in the estimate.

There are many issues to consider and challenges in each of these steps, particularly where values are scaled or aggregated over people, areas or time periods. Johnston and Wainger (2015) discuss many of these issues. A general note is that empirical evaluations have shown that, on average, benefit function transfers tend to be more accurate than comparable unit value transfers (Johnston et al. 2015, Kaul et al. 2013, Rosenberger 2015, Johnston and Rosenberger 2010).

Non-use benefits present distinct challenges for monetary valuation and for the development of suitable biophysical indicators. The legitimacy of nonuse values as a component of total economic value is supported by economic theory (e.g., Freeman et al. 2014, Richardson and Loomis 2009) and considerable indirect evidence.[6] Stated preference methods (the only method available to monetize nonuse benefits) have "… been tested and validated through years of research and are widely accepted by federal, state, and local government agencies and the U.S. courts as reliable techniques for estimating nonmarket values" (Bergstrom and Ready 2009, p. 26; cf. Kling et al. 2012). Despite this, some federal agencies still implicitly or explicitly discourage use of the stated preference valuation methods, as primary studies or in benefit transfers, due to the controversy over these methods. A restriction on the use of such methods prevents monetary valuation of nonuse values.

The alternative to monetary valuation of non-use services is to use BRIs to represent benefits. However, BRIs for intangible benefits, as with all BRIs, need to be selected based on evidence that they represent public values and clearly communicate value to most members of the public. In some cases, the usefulness of a biophysical metric—particularly for representing the value of a regional or national program—can be enhanced by adding a complementary metric that quantifies the rarity of a service or the conservation significance of the species habitat or ecosystem being improved (Krutilla 1967). For example, the changes due to the project could be placed in the context of a national restoration goal for a scarce or unique element (e.g., achieves two percent of the goal to restore 15 percent of historic habitat of endangered freshwater mussels) to demonstrate a national interest.

The remainder of this report demonstrates use of these methods for valuing the four ecosystem services shown in the conceptual value diagram (Figure 1).

Property Owner's Benefits of Improved Water Quality

Valuation approach

Improvements in water quality from agricultural conservation practices often take place in areas with predominately private land. As such, many of the beneficiaries of improvements will be residents living along lakes and streams with improved water quality. Economic valuation has demonstrated that water quality is capitalized in property values (i.e., home values and prices of undeveloped lots), meaning that residential properties located near clean water bodies are worth more than ones near polluted water, all else being equal. Because of this relationship, the BRI for property value enhancements is the number of houses and lots adjacent to and up to a quarter mile from the water body, as has been generally demonstrated in many prior modeling efforts (e.g., Walsh et al. 2015, Klemick et al. 2015).

Changes in property value due to water quality changes can be quantified using the hedonic property method (Freeman et al. 2014). This method applies a statistical model to quantify differences in the sale prices of houses on water bodies with different levels of water quality, after controlling for all

[6] For example, individuals are often observed to donate money to preserve wildlife species or habitats that they have no plans to use. Such donations suggest that nonuse values exist, although they cannot be used to develop theoretically "clean" and defensible estimates of nonuse value due to free riding and other concerns (Champ et al. 1997).

the other structural and neighborhood features of the home that modify value, such as numbers of bedrooms and baths. The preferred approach is for the analyst to collect data on sale prices of homes and housing characteristics within the watershed of interest. However, if that is not possible due to budget or time constraints, a benefit transfer can be conducted by using prior studies to estimate a percentage increase in average house price associated with specific measures of improvement in water quality. It is important to note that the dollar values from a hedonic property model are present values of a future stream of benefits. Such values must be annualized to produce an annual value, if they are being compared to annualized costs or aggregated with other annual benefits.

Case study – Value of changes in nitrogen and phosphorus concentrations to property owners

A benefit transfer approach was used to illustrate how property values in portions of the Western Lake Erie Basin could be affected by expected changes in instream concentrations of nitrogen and phosphorus. A hedonic property model (Liu et al. 2014) was available for the Upper Big Walnut Creek watershed, an area just south of the Western Lake Erie watershed and centered around Columbus, Ohio. Upper Big Walnut Creek is a CEAP Watershed Assessment Study and thus has long term, good quality water quality monitoring and ecological data and assessments. This study provided estimates of how nitrogen concentrations, phosphorus concentrations, and Secchi depth measurements affected waterfront and non-waterfront property values.

To apply the hedonic residential property value model to the Maumee River, water quality improvements that had been estimated from Keitzer et al. (2016a), as changes in nitrogen and phosphorus, were connected to the water quality metrics used in the hedonic property model. Keitzer et al. (2016a) estimated that conservation practices adopted on only eight percent of the Maumee River watershed—the area with the greatest need for conservation (, Scenario 1)—resulted in a seven to eight percent reduction in total nitrogen (mg/L) and total phosphorous (mg/L) concentrations in river water quality (Keitzer et al. 2016a). Data from Liu et al. (2014) were used to estimate that a one percent reduction in mg/L of nitrogen and phosphorous increased property values by 2.46 percent and 8.8 percent, respectively. (Methods are detailed in Appendix B.)

In a complete benefits analysis, all the houses within a reasonable distance of all significant rivers (e.g., within a quarter mile of a water body) would be identified using GIS data or county assessor's office data and included in the estimate of property value changes.[7] However, in this report, we calculated affected properties using a sample of six towns along just one of the main rivers in the watershed, the Maumee River. The six major towns along the Maumee had a total of 25,613 single family or townhomes, creating the benefit relevant indicator (Figure 1, arrow 4a). The median home value per town was averaged across the six towns to generate a value of $123,785 per home to use in the illustrative benefit transfer.

The change in property value per home that can be attributed to the change in total nitrogen concentration (mg/L) is then calculated as:

(6.6% change in nitrogen) * (2.46% change in home value per 1% change in nutrient) * ($123,875 home value) = $202/home

Equation 1. Estimated change in value per home for projected change in total nitrogen concentration

[7] Characteristics such as ownership, topography, and other factors may need to be considered when identifying affected properties.

A similar calculation is made for the expected change in phosphorus, and the values for both nutrients were summed. The water quality changes add about $202 and $871 per change in N and P respectively, which, when summed to $1,073, were roughly a 0.9 percent increase in value. Multiplying $1,073 by 25,613 houses generates a total change of about $27 million in capitalized property values.

Thus, for the scenario in which the most vulnerable eight percent of the Maumee watershed area adopted agricultural conservation practices, nutrient reductions of seven to eight percent yielded a $27 million (around one percent) increase in economic value to the sampled (25,613) homeowners (Figure 1, arrow 5a). For comparison, substantially increasing the percentage of adoption within the Maumee River to about half the watershed (, Scenario 2), would result in a $131 million (four percent) increase in economic value just to the houses in the five example towns along the Maumee River. These benefits are present values of a future stream of values and would need to be annualized to be compared to other annual costs or benefits.

Although we used a hedonic model to estimate changes in the Maumee watershed, we were not able to estimate change in property values for homes on Lake Erie, even though a hedonic study was available for transfer (Ara et al. 2006a and 2006b). The Ara et al. model could not be transferred because the predicted change in water quality (nitrogen and phosphorous) could not be connected to the measure of water quality in the Lake Erie hedonic property model (water clarity as increased Secchi depth) for two primary reasons. First, data was not often collected in the nearshore zone, which was the most direct relationship to the hedonic model. Second, collecting Secchi depth measurements is technically challenging and somewhat confounded by the seasonal presence of algae blooms and when thick mats are present (therefore these measurements are generally limited in nearshore zones). The disconnect and data gap would need to be addressed, if technically possible, to provide a more complete accounting of the property value benefits of improving water quality from agricultural practices in the Western Lake Erie Basin.

Nonuse value of Improvements in Aquatic Community Condition

Valuation background and approach

In the ecosystem service benefit chain diagram (Figure 1), the ecosystem service benefit, protecting the aquatic community (arrow 4e), may initially appear to be an intermediate ecosystem service for which independent values should not be estimated (Johnston and Russell 2011). However, this service, which might be more explicitly characterized as satisfaction from protecting the aquatic community (a source of potential nonuse value), does constitute a final ecosystem service for many beneficiaries, and is critical to assessing conservation practice benefits. Many people derive value from protecting and restoring ecosystems and habitats, even if they do not use those systems in any tangible manner. Evidence that people hold such nonuse or passive use values for aquatic species and ecosystems has been found by many stated preference studies (e.g., Rudd et al. 2016, Richardson and Loomis 2009, Johnston et al. 2002, 2011, and 2016), surveys (NSRE 2000), and legal findings that state that governments can and should use such values in natural damage assessments (Carson et al. 2003, Johnston et al. 2017).[8]

[8] For example, Johnston et al. (forthcoming, 2017) note that "in 1989 the U.S. Court of Appeals ruled that compensable values include 'option and existence values,' which opened the door for [contingent valuation] estimates to be used in litigation, since only [stated preference] methods can measure non-use values." See 880 F.2d 432, 279 US App. D.C. 109, p. 44.

In the Lake Erie Basin, efforts to reduce nutrients and sediments have been linked to fish condition by multiple biophysical models. Keitzer et al. (2016a and 2016b) used regression models to relate stream discharge, nutrients, and sediments to the ecological condition indicators of the Fish Index of Biotic Integrity (IBI), a Piscivore Index, and Sensitive Species Richness. Using hypothetical scenarios of conservation practices, they projected that implementation of erosion control and nutrient management on 48 percent of the vulnerable cropland area would, on average, improve the Fish Index of Biotic Integrity by six percent, the Piscivore Index by 42 percent, and Sensitive Species Richness by seven percent, relative to baseline (1990–2010) conditions. Additional forecasting models projected that these improvements would result in 12 percent and 25 percent fewer stream miles in the basin where nutrients and suspended sediments limited the Fish Index of Biotic Integrity and Piscivore Index, respectively.

To value (monetize) the nonuse benefits derived from protecting or restoring an aquatic community (independent of uses such as recreational fishing), the analyst must rely on stated preference estimates, either from original studies, or via benefits transferred from prior stated preference studies conducted in other areas. Assuming a primary stated preference study is infeasible, the first step in valuing a change in a multi-metric index of aquatic community condition is to evaluate whether the economic literature has studies that are sufficiently like the case study site to support a unit value or benefit function transfer. Databases of valuation studies are maintained by several groups to facilitate such assessments.[9]

Case study—Nonuse values of changes in the Index of Biotic Integrity (IBI)

For the case study example, the team found a single paper that could be used to conduct a unit value transfer of the changes in the Fish Index Biotic Integrity. Johnston et al. (2011) synthesized the results of a choice experiment study[10] in which a fish IBI was an attribute designed to reflect overall ecosystem condition. As explained by Johnston et al. (2011), respondents were willing to pay for improvements in ecosystem condition (holding other ecosystem effects constant), due to significant nonuse value for such changes. In their study, a 100-point Fish Index of Biotic Integrity was used to characterize overall aquatic ecological condition within a stated preference discrete choice experiment, and changes in this attribute were associated with nonuse willingness-to-pay (WTP). Details of this primary study are omitted here for conciseness, but are provided by the published article (Johnston et al. 2011; also, see Johnston et al. 2012, 2013).

There are potentially significant differences between the Johnston et al. (2011) study and the case study area of the Western Lake Erie Basin that are potential sources of error in value estimates. For example, the original study was conducted in Rhode Island, whereas the case study encompasses portions of Ohio, Indiana, and Michigan. These areas differ across many characteristics that are potentially relevant for WTP. Agriculture is the dominant land use in the case study, unlike in Rhode Island, where agriculture covers a relatively small proportion of total land. Moreover, there are non-trivial differences in the population characteristics of the case study and the study sites, such as income. Perhaps more importantly, the original study addresses habitat improved by the provision of fish passage over dams, whereas in the Western Lake Erie Basin, the team is considering habitat improved via water quality change only. These and other differences suggest that transfer errors in this illustrative example are likely to be substantial. Economic analysts must always evaluate

[9] For example, the Environmental Valuation Reference Inventory (EVRI, http://www.evri.ca) is the largest nonmarket valuation database in existence, with data from thousands of valuation studies worldwide (Johnston et al. 2015).

[10] Choice experiments are a type of stated preference study in which survey respondents are asked to choose among bundles of goods with different prices. Statistical analysis is then used to isolate willingness to pay for increases in goods or services. For further information, see Freeman et al. (2014).

whether values generated are sufficiently precise to support the program or policy decisions being considered.
If the analyst chooses to continue with the transfer, while recognizing these limitations, the value function estimated by Johnston et al. (2011) provides a single implicit price (or marginal WTP estimate) per unit of change in the IBI. This result implies that, on average, households in the sampled area (the State of Rhode Island) were willing to pay $1.19 (in 2008 dollars) per one percentage point increase in the 100-point IBI. Hence, conservation practices leading to a 10 percent improvement in the Fish Index Biotic Integrity (e.g., from 60 to 70 on the 100-point index scale) would lead to an average per household value of $11.90 per year.

Among the challenges facing those seeking to estimate aggregate nonuse values for ecosystem services is a determination of the "extent of the market," or the total population of people who hold values for changes (and where those people live), and who have standing for any given benefits analysis (Loomis 1996, 2000). A related challenge is the need to understand how nonuse values might change within that area—for example, do average per household nonuse values decline as one moves farther away from an affected area? Challenges such as these apply to all types of aggregate benefit estimation, but are particularly imposing for benefit transfers of nonuse values since interest in a service cannot be observed (Hanley et al. 2003, Bateman et al. 2006, Schaafsma 2015, Johnston et al. 2015b, 2016). In the present case, results from Johnston and Ramachandran (2014) suggest that WTP did not decay with distance for changes in this IBI, meaning that Rhode Island residents living closer to the Pawtuxet watershed have, on average, the same WTP as residents living at a greater distance. Results further suggested that the full set of beneficiaries likely extended beyond the watershed. For simplicity, the team used the entire state of Ohio in the current transfer.

Given these assumptions, a simple multiplication then provides the value of a change in fish IBI. Here, we illustrate calculations for residents of Ohio. The value of $1.19 (in 2008 dollars) is converted to $1.31 in 2015 dollars using the Bureau of Labor Statistics inflation calculator (BLS 2016). Total value of change in fish IBI (Figure 1, arrow 4e) is calculated as:

(6% change in fish IBI) * [$1.31 (2015$ value per % change] * (4,570,015 people in Ohio) = $35.9 million

Equation 2. Total annual value of change in fish IBI

Because Rhode Island has a higher median household income than Ohio and income can have a strong effect on willingness to pay, the value per percentage change in IBI might be adjusted by the ratio of incomes between the populations being used. This is an ad hoc adjustment and reflects an assumption that WTP changes in fixed proportion to income, which implies that the income elasticity of WTP is equal to one. In 2014, Ohio's median household income was $49,308 and Rhode Island's median household income was $56,253, meaning that Ohio's household income was 87.6 percent of Rhode Island's. Therefore, as a sensitivity analysis, the team also compared a per unit value change of $1.14 and calculated a total annual value of $31.3 million, which only modestly reduced the total annual value. Further, as an additional sensitivity analysis, the team calculated a more conservative value, using the population of the watershed alone (1.2 million people) to generate an annual value of $9.4 million in nonuse value, a value substantially lower than that generated when the entire state's population was used. Since we have not measured the spatial extent of willingness-to-pay for this benefit in this location, we cannot be certain of the appropriate extent to use in estimated value.

These variations in value demonstrate the relatively large effect of assumptions regarding the affected population of households on the aggregate value. Assumptions regarding changes in beneficiary population sizes (or the size of the market) often outweigh the effect of the assumptions of value per person in terms of influence on aggregate benefit estimates. Note that ad hoc adjustments to the estimated values are not guaranteed to improve the accuracy of the benefit transfer. The team recommends that benefit transfers used to inform decisions apply more sophisticated adjustments to enhance transfer accuracy, as discussed by Johnston et al. (2015). However, even these rudimentary benefit transfers can serve to demonstrate the potentially significant nonuse values generated by improvements to aquatic ecological condition.

There are many limitations to this type of benefit transfer, suggesting that transfer errors could be considerable (Johnston et al. 2015). In general, more similar sites are associated with lower-error benefit transfers, although the full set of variables over which site similarity is important is not generally known (Johnston et al. 2015). Further, the uncertainty of the ecological calculations introduces additional error. The use of biotic multi-metric indices is a common approach to measuring restoration outcomes, but indices such as these have a variety of limitations when used for valuation (Boyd et al. 2016). For example, metrics such as these can be insensitive to restoration, such that restoration activities do not increase index scores (Palmer et al. 2014). Therefore, it is important to use such relationships in the correct context and at the appropriate scale. Factors limiting their use for conservation practice evaluation include:

1) water quality (or outcomes of conservation practices) is one of many factors limiting habitat quality;
2) streams, lakes, and estuaries, and even particular reaches of each, will vary in their sensitivity to water quality changes due to physical conditions alone; and
3) the current level of water quality degradation may prevent a response in the aquatic community until a threshold level of pollution reduction is achieved or until the system shows a response to lower loads (i.e., given lagged responses).

Further, IBIs are in widespread use and may be convenient for many case studies. However, there is no clear scientific consensus on the best single measure of aquatic system health, and the use of IBIs can be controversial (Borja et al. 2015). Therefore, analysts, in consultation with local area scientists and water quality stakeholders, may want to consider other multi-metric indices, such as those in Borja et al. (2014), for use as benefit relevant indicators. Ultimately, the most appropriate indicator of ecosystem health or condition—where these measures are directly relevant to social value—will vary across sites based on available data and understanding.

Sport fishing Valuation Section

Valuation background and approach

Recreational fishing is recognized as a major ecosystem service of freshwater systems. The annual net value of recreational fishing in the Great Lakes basin is estimated to range from $393 million to $1.47 billion (in 2012 dollars, Poe et al. 2013). Although recreational fishing effort and benefits are substantial in many regions, demonstrating a recreational fishing benefit due to conservation practices can be challenging, primarily because of the difficulty of demonstrating effects of conservation practices on sportfish populations. However, if effects on sportfish populations can be credibly estimated, a national analysis has been developed that is potentially appropriate for estimating changes in recreational fishing values throughout the U.S. and Canada (Johnston et al. 2006). This model used a large set of existing studies to estimate a meta-regression equation (a type of benefit transfer function), which enables forecasting of the value per (additional) fish caught, as a function of multiple variables characterizing factors such as species, geographical region, anglers, fishing type, and catch rates.[11]

Before applying the meta-regression benefit transfer, we must establish that the equation is appropriate to the case study context. First, the studies used to estimate value should share characteristics of the case study, such as including the same species in similar ecological and socio-demographic settings. Next, it is necessary to determine values for model variables that characterize the case study area and that will be "plugged in" to the model, to estimate the willingness to pay per marginal fish (Johnston and Wainger 2015). For example, if the benefit function includes average angler income as a variable, it is necessary to obtain a measure (or approximation) of angler income for the case study site, e.g., group of counties, to use when applying the model.

The complete process of applying the meta-regression benefit transfer typically requires multiple models or intermediate calculations to estimate the necessary model variables. The primary ecological indicators used to assess changes in recreational fishing quality are the relative abundance of representative sportfish species. Thus, to connect conservation practices to benefits, current fish abundance is assessed and changes in abundance due to management actions are estimated as a function of changes in the key water quality outcomes of management actions—such as nutrient or suspended sediment concentrations (Figure 1, arrow 3a). In general, there is clear experimental evidence that nutrient enrichment causes changes to algal communities in streams, which can, in turn, influence invertebrate and fish communities (e.g., Perrin et al. 1997). Further, a growing number of observational studies have shown associations between agricultural practices, stream nutrient and suspended sediment concentrations, and the composition of invertebrate and fish communities (e.g., Wang et al. 2007, Chase et al. 2016, Holmes et al. 2016).

[11] A meta-analysis is a statistical analysis of results from many prior primary studies, generally used within benefit transfer to estimate a single, umbrella benefit function that enables the analyst to estimate benefits for a wide range of possible case study sites. See also Appendix C for further information.

However, the effect of nutrient enrichment on valued fish species can be positive or negative, thereby necessitating careful evaluation of likely fish community effects, before estimating economic values. Empirical models that demonstrate a relationship between stream chemistry and fish populations are often weak unless several other factors (e.g., stream size, habitat quality) are included—and these other factors tend to exert stronger influence on the models than do chemical factors (e.g., Wang et al. 2007). Hence, researchers often have difficulty developing defensible quantitative relationships to directly link changes in agricultural management to changes in sportfish abundance. The absence of defensible cause and effect functions between management actions and ecological outcomes is a common impediment to valuing changes in sport fishing as well as many other types of ecosystem services (Wainger and Boyd 2009).

Case study—value of changes in sportfish catches

Before applying the benefit transfer function, we evaluated its appropriateness for the case study. First, we confirmed that the species affected by the management effort included those used to fit the meta-regression. Here, the walleye, trout, bass, perch and other types of panfish, and other species affected in the Western Lake Erie Basin are the same types used in the meta-regression. Many other factors could influence transferability that are not represented in the transfer equation; for example, types and level of environmental degradation may affect the desirability of fish caught. Since a substantial portion of the primary studies used in the meta-regression were from the Great Lakes, the team had confidence that the conditions in the Western Lake Erie Basin would be represented by the meta-analysis equation.

Next, it was necessary to evaluate whether the critical data needed to use the meta-analysis model were available. Data necessary to populate the meta-analysis of Johnston et al. (2006) include:

1. Baseline total sportfish catch (no action scenario)
2. Proportion of total angler catch of each of the affected recreational species (percentage)
3. Catch rate (per day)
4. Change in catch under a management scenario (as a percentage of current catch)
5. Average income of anglers (in dollars)

These data needs were filled with a combination of existing case study data, national surveys, literature values, and simple analyses. Both the baseline catch and change in catch as a function of conservation practices required connecting several models and calculations. The following text describes methods used to estimate baseline catch (item 1) and change in catch (item 4) using incomplete data. This description shows how analysts can overcome data gaps and some of the uncertainty often associated with estimates. Other calculations are described in Appendix C.

The total recreational fish catch is rarely available directly, but can often be empirically estimated from available data. To assess the current condition of a recreational fishery, many state fisheries departments estimate fish populations and recreational harvest using creel surveys.[12] Many of the valuation estimates of recreational fishing have relied on empirical models developed from creel surveys to project effects of a management action (Johnston et al. 2006; e.g., Schuhmann and Schwabe 2004). Alternatively, fish abundance can be estimated from community samples (e.g., Mazzotta et al. 2015, Johnston et al. 2016), which are routinely collected as part of many biological monitoring programs (USEPA, 2002); catch is typically assumed to be a percentage of the population. Alternatively, if neither creel nor community surveys have been conducted, an analyst

[12] Creel surveys are angler interviews or questionnaires used to obtain information on recreational fishing activities, such as the number of days fished, fishing locations, the number of fish caught (often segregated by species type), angler attributes, etc.

can use national surveys of fishing participation days (USFWS 2011a) and estimate catch per day (Mazzotta et al. 2015) to estimate total fish caught for states and major watersheds, as shown in the example below. However, estimates based on past national surveys are less likely to reflect the current condition of the recreational fishery than data typically collected by state agencies.

For the case study, the team found that results of a 2015 creel survey were available for only part of the watershed and published reports only provided catches for two sportfish species, walleye and white bass (ODNR 2016). Additional data and local knowledge were likely available to extrapolate results from the creel survey to the entire Western Lake Erie Basin. However, in the interest of time, the creel survey was supplemented with national data on sport fishing participation and typical catch rates to estimate total fish caught recreationally. Using national data to estimate total sportfish catch provided a means to scale catches to the entire Western Lake Erie Basin, rather than the sub-portion of the watershed represented by the survey data.

The team estimated the baseline recreational fishing effort in the entire watershed using the readily available National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (FHWAR) survey for Ohio (USFWS 2011b).[13] The team apportioned the entire state's fishing effort (excluding the 956,000 days fished in the Great Lakes) to the Western Lake Erie Basin by assuming fishing effort was uniform over the state and setting the portion of days fished in the basin equal to the portion of total state area that fell in the basin.[14] To complete the calculation of total fish caught, fishing days were multiplied by fish caught per day. An average catch rate from the literature of 2.3 fish/day from Mazzotta et al. (2015) was applied to generate an estimate of total recreational catch (an assumption that can be used in data poor situations).

A rough calculation of the benefit relevant indicator of the baseline total fish caught per year (baseline condition for the Western Lake Erie Basin) was calculated as:

(16.9 million Total OH sportfishing days) * (17% of OH in Western Lake Erie Basin) * (2.3 fish caught per day) = 6.6 million total fish caught per year (all fish).

Equation 3. BRI of the total fish caught per year.

Note that this benefit relevant indicator was adjusted from the initial indicator (fishing days) that the group initially identified (Figure 1, arrow 4d), because it was adapted to fit the input requirements of the meta-regression equation. This result illustrates how analysts can efficiently match indicators to the available valuation literature before conducting calculations. In addition, species-specific catch estimates were later created for a representative set of fish species, to make better use of the meta-regression equation (see Appendix C).

The estimated change in this baseline fish abundance and recreational catch due to management actions was based on analyses of Keitzer et al. (2016a), who developed statistical models that predicted the degree to which concentrations of nutrients and suspended sediments (adjusting for stream size) limited the relative abundance of a subset (i.e., piscivore index) of sportfish species. Model simulations predicted that implementation of erosion control and nutrient management on the

[13] Data were also available for the Great Lakes region but that extent did not match the biophysical modeling of piscivore changes.
[14] Other methods have been developed to use population distribution and travel behavior to apportion effort by sub-region (Mazzotta et al. 2015, U.S. EPA EnviroAtlas 2015).

most vulnerable 48 percent of the Western Lake Erie Basin land area would, on average, improve the piscivore index by 42 percent relative to baseline (1990–2010) conditions.

The team used the 42 percent increase in piscivores to estimate a 42 percent increase in piscivore catches. The team approximated the magnitude of the ecological change by multiplying the expected 42 percent increase in piscivores by the total piscivores caught per year of 8.64 M (derived from data in Table C-3). The result of this multiplication provides an estimate of the increase in total fish abundance due to the management action.[15]

8.64 M piscivore catch/year * 42% increase = 3.6 M additional fish caught

Equation 4. Estimation of the increase in fish due to conservation implementation

To obtain an initial estimate of the value of this change in catch, we assume that all 3.6 M additional fish are caught[16]. (Total value was not highly sensitive to this assumption as shown in Table C-7). Given our assumptions, the majority of these fish increases are walleye and white bass (the latter of which are assumed to be valued as a panfish). If the team had information on the total catch of bass, trout, and musky, these species could also have been included explicitly. As it is, they are included implicitly because the changes in walleye and white bass are scaled to represent total change in piscivore catch. Note, there is no consideration of value increases in non-piscivore panfish, since the ecological outcome metric only estimated the change in piscivores, suggesting that this value could be an underestimate if non-piscivore fish were affected.[17]

[15] This function is limited to estimating the benefits of recreational angling, assuming that changes in fish abundance lead to concomitant increases in harvest, and that angling restrictions do not prevent anglers from catching additional fish. Alternative approaches would be required to estimate benefits that might occur in commercial fisheries over time (Anderson and Seijo 2010).

[16] Relaxing this assumption would require a more sophisticated model of recreational angler behavior and catch, as influenced by underlying fish abundance and related factors. Development of such a model is beyond the capacity of the present analysis.

[17] In simple terms, piscivores are animals that eat fish. Here, we use the term to refer solely to fish that eat other fish, including walleye and bass.

Applying the benefit transfer function

Once all the necessary variables were estimated, applying the meta-regression equation of Johnston et al. (2006) to the case study required 1) developing the appropriate variable estimates, 2) using the parameter values to estimate the change in value of the additional fish caught, and 3) aggregating values to estimate the total value of the fishery changes (Figure 1, arrow 5d, methods detailed in Appendix C). In this case study, the team had reasonably valuable information on the recreational fisheries necessary to conduct the transfer, but was nonetheless forced to make assumptions and use supplemental data sources to address the five critical data needs identified above.

With the critical data needs met, the benefit function can be applied from Johnston et al. (2006) to calculate the average value per additional fish caught (Table C-5). Many variables, other than the critical data needs, are used in the equation (Appendix C). These variables capture methodological details of the primary studies that influence valuation results. However, since these variables are not relevant to applying this equation to the case study area, those variables are set to the average value of the primary studies used in the meta-regression analysis (Johnston and Wainger 2015, Stapler and Johnston 2009). Based on calculations in Appendix C, the total WTP per additional walleye is $13.95 and per additional white bass is $3.30 in 2015 dollars.[18]

The values per additional fish are multiplied by their respective catch increases to generate a total value of the management action. The increases in the value of fish are estimated to be $17.03 million for walleye and $4.24 million for other piscivores, for a total benefit for the management scenario of $21.98 million (in 2015 dollars).

As noted above, this case study analysis is used primarily for illustrative purposes. The use of benefit function transfer is generally preferred to the transfer of unit values because of its ability to reduce error in value estimates by representing local conditions (Johnston and Wainger 2015, Johnston et al. 2015). However, a simple sensitivity analysis demonstrates that value varied dramatically (from $11–22 million) after varying assumptions about numbers of fish caught and fishing days (Table C-7). A benefit transfer with sufficient accuracy to support decision-making would likely require additional attention to obtain more accurate data for the study site and to conduct more extensive sensitivity analysis over any remaining assumptions. However, the general process would follow the outline provided above. For additional details on the use of this model for benefit transfer, see Appendix C and Johnston and Wainger (2015).

Economic benefits of reduced sedimentation rates

Reduced sediment delivery to waterways has many potential benefits, as suggested in the conceptual value diagram (Figure 1). For this analysis, we isolate the effects of reduced erosion on the magnitude and rates of sedimentation within reservoirs and within harbors and shipping channels. We then estimate the contribution of reduced sedimentation to the ecosystem service benefits of water supply and recreation in reservoirs and of commercial navigation in harbors and channels by transferring values from two existing studies, one conducted for reservoirs and one conducted for harbors and channels.

Implementation of in-field conservation practices in the Western Lake Erie Basin are estimated to have reduced the annual rate of sheet and rill erosion by nearly a million tons of sediment per year between 2003 and 2012, by one estimate (USDA NRCS 2016a, page 29), or by 0.5 tons per acre per year (from 1.3 to 0.8 tons per acre per year), by another estimate (USDA NRCS 2016a, Table 3.2),

[18] This is the value after applying log transformation and adjusting for inflation.

which, when scaled to the watershed, represents an estimated reduction of 2.4 million tons of sediment per year.[19] The amount of cropland in WLEB treated with at least one structural practice designed to control or trap runoff losses of sediment increased from 34 percent in 2003–06 to 55 percent in 2012 (USDA NRCS 2016a, Table 2.1), or over 1 million acres.

Reservoir impacts

No studies in the case study area provide direct estimates of links between field practices and sedimentation delivery to reservoirs (Figure 1, arrow 3a). However, Hansen and Hellerstein (2007) developed empirical relationships to characterize relationships between adoption of any field practices that affect sheet and rill erosion and sediment delivery for reservoirs throughout the U.S. Hansen and Hellerstein (2007) estimated marginal changes in sedimentation rates using a two-step process of 1) estimating sedimentation delivery and 2) valuing changes in sediment delivery as dredging costs avoided. For the first step, they empirically estimated sedimentation rates as a function of up-stream, on-field erosion rates and reservoir sizes using historical data for reservoirs across the U.S.

In the second step, Hansen and Hellerstein (2007) valued reservoir sedimentation changes using a replacement cost approach. Dredging costs were used to reflect the replacement value of all services that are impacted by sediment delivery to reservoirs, including drinking water and recreational boating. This valuation approach relies on the assumption that decisions to replace or repair goods are economically efficient—that is, the value of the restored services must be greater than or equal to costs. However, this assumption does not always hold and therefore, results of such an approach must be used cautiously (Champ et al. 2012). The values estimated by Hansen and Hellerstein (2007) capture the multi-year decrease in cost of a one-time reduction in soil erosion due to increased reservoir capacity created between dredging events.

Further analysis of the Hansen and Hellerstein (2007) work by Hansen and Ribaudo (2008) provided values for one-ton reductions in sheet and rill erosion for watersheds (8-digit hydrologic unit codes or HUCs) throughout the U.S., including those within the Western Lake Erie Basin (see Appendix D for further information).[20] Values ranged from $0.003 to over $0.20 per ton of sediment reduction and averaged $0.09 per ton across the basin (Table D-1). Applying the average value to the expected reduction in sheet and rill erosion of 2.4 million tons per year for changes from 2003–2012 (Scenario 3), generates an annual value for the decreased level of sedimentation in reservoirs of approximately $219,000 per year for 2003–2012. This value estimate is sensitive to the choice of baseline year since substantial effort at controlling sheet and rill erosion (resulting in an up to 75 percent reduction) occurred prior to 2003.

This approach to estimating benefits does not require an intermediate BRI. However, BRIs appropriate for evaluating the potential magnitude of benefits due to reduced sedimentation in water bodies are identified in the conceptual value diagram as the number of users of systems affected by sediments such as municipal water supply infrastructure and shipping channels (Figure 1, Arrows 4b and 4c). For enhancements to municipal drinking water supply, a potential BRI is the number of water users dependent on reservoirs with altered sedimentation. However, data were not available to estimate this BRI for the case study.

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[19] This estimate of erosion reduction, by design, does not include sediment trapping in edge of field buffers, which declined by 0.6 tons per acres per year (from 1.1 to 0.5 tons per acre per year). This sediment trapping was excluded because the benefit transfer function was developed for changes in sheet and rill erosion only.

[20] The per-ton benefit values are available in two databases on the ERS web site (www.ers.usda.gov). One provides per-ton benefits of soil erosion reduction for the 3,074 counties within the 48 contiguous states. The other provides per-ton benefits for the 2,111 8-digit Hydrologic Unit Code (HUC) watersheds within the contiguous states.

Harbor and channel impacts

The benefits of reduced sedimentation in harbors and channels include reductions in delays in shipping and damages to ships due to groundings. The analysis of these avoided costs to commercial shipping follows roughly the same approach as that used for reservoir effects. In this case, Hansen et. al. (2002) used multiple national datasets and a hydrologic model to estimate the annual average quantity of sheet and rill erosion upstream of each harbor and shipping channel and used those estimates as proxies for sedimentation rates. The monetary benefit of reduced sedimentation was the present value of the damages avoided from ship groundings and shipping delays that the sediment might have imposed. Benefit measures were derived from dredging cost data reported by the U.S. Army Corps of Engineers, based on a replacement cost approach. The underlying assumptions were that 1) after a channel/harbor is dredged, sediment initially imposes no damages; 2) damages were assumed to increase linearly with respect to sediment accumulation; and 3) decisions to dredge were economically efficient—e.g., dredging occurs when the present value of benefits justifies costs. For details, see Hansen et al. (2002).

Using results from Hansen et al. (2002), Hansen and Ribaudo (2008) generated values of one-ton reductions in field erosion for most HUCs, including those within the Western Lake Erie Basin. Values ranged from $0.03 to $0.06 per ton, an average of $0.05 per ton (Table D-1). Within our case study, each year that the conservation practices are in place, sheet and rill erosion at the edge of field was expected to be 2.4 million tons lower, on average, since the baseline year of 2003. Given the $0.05 per-ton impact, the benefits (from the reduction in the portion of sediment reaching shipping channels) were estimated to be $122,000 per year from the 2003 baseline. When added to the value of erosion reduction impacts on reservoir services, the total value of the reduction in sedimentation due to the 2.4 million-ton annual reduction in sheet and rill erosion was $349,000 per year.

Summary and Discussion

This effort generated monetary values and, in some cases, non-monetary benefit indicators, for five ecosystem services:

1. Enhanced property values
2. Improved sport fishing
3. Improved aquatic community condition (non-use value)
4. Improvements to drinking water supply and other uses of reservoirs
5. Commercial navigation improvements (operation and maintenance costs avoided).

The results suggested that the largest benefits were associated with nonuse values and recreational fishing (Table 2). All five of these benefits were monetizable with available data from within and external to the case study. However, values are meant to be illustrative only, since they have major sources of error associated with data or information gaps. We emphasize, once again, that the goal of this analysis was to illustrate the types of methods that can be applied to estimate values of ecosystem services derived from water quality changes—without the use of new primary studies. More accurate and reliable benefit estimation would require time and resources not available for the present analysis. In addition to these caveats, the values reported here are not completely separable nor mutually exclusive and thus cannot be aggregated without further analysis to disentangle effects.

Table 2. Benefit relevant indicators and example monetary values of ecosystem service benefits of conservation practice adoption in the Western Lake Erie Basin

Service	Geographic Area of Management Scenario	Level of conservation practice adoption	Benefit Relevant Indicator(s)	Monetary Value[1]
Property value enhancement	Six towns in Maumee Basin	High vulnerability (8% – 48% of cropland)	26,000 residences	$ 27 – 131 million in present value
Aquatic community condition (non-use value)	State of Ohio	High and moderate vulnerability (48% of cropland)	6% improvement the Fish Index of Biotic Integrity	$36 million annually
Sport fishing	Maumee watershed	High and moderate vulnerability (48% of cropland)	16.9 million sport fishing days 6.6 M fish caught annually (baseline condition)	$22 million in annual angler benefits
Reservoir effects on drinking water supply and water-based recreation (replacement costs)	Western Lake Erie Basin	>1 M acres adopting structural soil conservation practices (21% of cropland)	Reduction in sheet and rill soil erosion (~2.4 million tons/yr)	$219,000 annually since 2003[2]
Harbor and channel effects on commercial shipping (damage costs avoided)	Western Lake Erie Basin	>1 M acres adopting structural soil conservation practices (21% of cropland)	Reduction in sheet and rill soil erosion (~2.4 million tons/yr)	$122,000 annually since 2003[2]

[1] *Values are for illustration purposes only and are not intended to represent the full value of conservation efforts. Values do not all represent the same geographic area nor time horizon and are not additive due to potential for overlapping sets of beneficiaries.*

[2] *Results for sheet and rill erosion are highly sensitive to the choice of baseline year because adoption of structural practices between 1982 and 1987 was substantial. In addition, values for sediment reduction do not include sediments trapped by edge-of-field buffers to maintain consistency with the benefit transfer technique.*

This initial valuation effort, conducted with limited resources, demonstrated that the types of data currently being developed in USDA CEAP assessment programs and elsewhere create opportunities for monetary valuation of benefits. However, error in estimates could be reduced by strategically investing in data collection and modeling. The Western Lake Erie Basin was an exceptional case study because substantial resources had been invested in generating economically-relevant ecological outcomes, such as in-situ water quality conditions and sportfish changes. Yet, even in this study, some of the ecological outcomes produced were either missing or mis-aligned (spatially or temporally) with economic analysis needs that were eventually identified.

The case study illustrations suggest that the estimation of ecosystem service values—particularly using benefit transfer—often requires multiple strong assumptions, even in a data-rich case study such as this one. Avoiding these assumptions requires more resource-intensive primary valuation studies. A more typical limitation to valuation in data-poor case studies is that the ecological outcomes that are modeled are unable to serve as a direct input to economic modeling. For example, this case study was unusual because, rather than stop with changes in nutrient loads, modelers had translated those nutrient changes into changes in the game fish population and aquatic community in general. This remains a data limitation, as that type of regional data and analysis is generally not available elsewhere in the U.S. at present.

The economic data needed to value changes in services was available in some form here, since the team had screened those benefits that were possible to value. Yet, substantial data collection and modeling resources would be needed to improve the precision of valuation estimates. The limits included a lack of depth in the available valuation studies that required the use of values that may not have been wholly appropriate for use in the case study. Additional valuation studies would have enhanced the accuracy and reliability of the benefit transfer estimates.

What this study primarily reveals is that while it is possible to value many ecosystem service changes using existing ecological and economic benefit transfer, many analytic challenges remain. Foremost among the challenges is that major assumptions were required for all ecosystem services to connect all components from actions to benefits. Thus, many value estimates are likely to have high error (low precision). Whether those errors are problematic depends on the decision context in which the values are being used. In some cases, demonstrating a rough order of magnitude of benefits may be useful in the decision process. However, for those decisions that require precision, more time and effort would need to be invested to reduce the error rate of the estimates.

Future data and model development needs

This report was not intended to be a comprehensive assessment of all benefits derived from water quality. Rather, the strength of the report is that it identifies sources of relevant information and demonstrates how such information can be used to assess public benefits that are often omitted from analyses. Additionally, the report sheds light on the types of public/private research that might contribute substantially to our ability to value the impact of agricultural policies and programs on ecosystem services.

The information gaps highlighted here indicate a need for valuation of ecosystem services to be designed as an interdisciplinary effort at the start of research programs. Conceptual value diagrams of ecosystem service values (e.g., Figure 1) that demonstrate the specific connections between biophysical models and economic models can be used as a tool to promote better integration of biophysical and economic models. Further, the team finds that the accuracy and breadth of ecosystem service valuation of agricultural conservation practices by field offices could be improved if the USDA performed a strategic set of a few original ecological and economic models specific of typical agricultural conservation practices in different but representative geographic areas of the country. To support some kinds of decisions, greater investment in understanding the uncertainty of value estimates will be needed, starting with understanding the variability of instream conditions (e.g., as discussed in Appendix A).

Many types of data and tools could be created to support valuation and reduce the level of resources needed to estimate values for case studies and national programs. The sport fishing valuation demonstrated that models that are specifically designed to facilitate benefit function transfers within the U.S. and Canada (Johnston et al. 2006) can greatly simplify analyses. Such meta-regression analyses, which are developed from multiple primary valuation studies, allow the user to tailor the value estimates to current or scenario-estimated conditions.

Another major challenge demonstrated by this case study is that separating the effects of conservation practices from land use change and other trends requires an investment in monitoring and modeling to different baseline conditions from effects of management actions. Both field measurements and models are needed to understand the net effects of conservation practices. In this report, except for the benefits associated with damages avoided to reservoirs, harbors and shipping channels, we valued hypothetical scenarios of conservation practice implementation, rather than effects of a specific set of practices associated with a specific USDA program. Therefore, it is essential that programs, such as the National Resources Inventory (NRI) and CEAP, continue to monitor the adoption of practices, the extent of that adoption directly attributable to conservation programs, and the combined measured impact of multiple practices under different weather conditions (being done in CEAP watersheds), to be able to evaluate program benefits.

We found that the ability to estimate values of program actions at the national scale will be limited by inconsistent data availability and a lack of tools to capture location factors that determine value such as degree of water quality degradation, presence of sensitive species, and variability of preferences, among other concerns. Although changes in nutrient and sediment loads may be well captured at the national scale, the effects on conditions within the water body, such as clarity or biotic status, are not routinely modeled. Recent developments to combine APEX and SPARROW modeling (Appendix A), may eventually serve as a source of nationally consistent data regarding instream concentrations of nutrients and sediments. Such work could support benefit transfer that uses water quality indices[21] as discussed under Methods, but is not illustrated in this report.

[21] See Abbasi (2012), Van Houtven et al. (2014), Terrado et al. (2010), Walsh and Milon (2015) for further information.

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Appendices

Appendix A – SPARROW

Economic analysts seeking to evaluate the water quality benefits of USDA programs require a means to estimate changes in water body conditions as a function of changes in implementation of conservation practices. In this report, we demonstrated the results for a well-developed case study with detailed watershed simulation modeling. However, a weakness of the simulation approach is that it does not establish an empirically demonstrable link between conservation practices and monitored stream water quality. A promising tool for addressing this need is the SPARROW (SPAtially Referenced Regression on Watershed attributes) model of water quality.

The SPARROW model uses statistical methods to relate mean water-quality loads estimated at select water-quality monitoring stations to spatially-referenced features of the upstream monitored basin (Smith et al. 1997, Schwarz et al. 2006). Although this tool currently has a limited capacity to directly evaluate the effectiveness of individual conservation practices in small watersheds, such as the subject of this study, new work has established the utility of combining SPARROW with simulation models to empirically establish the effect of an aggregation of conservation practice implementation in large regions. A useful by-product of the empirical approach is the ability to characterize the uncertainty inherent in the assessed effect of management actions on instream water quality.

SPARROW models, both regional and national in extent, have been developed to evaluate total nitrogen and total phosphorus (Smith et al. 1997, Alexander et al. 2008, Hoos and McMahon 2009, Ator et al. 2011, Moore et al. 2011, Garcia et al. 2011, Robertson and Saad 2011, Brown et al. 2011, Rebich et al. 2011, Wise and Johnson 2011, Domagalski and Saleh 2015, Saleh and Domagalski, 2015), suspended sediment (Schwarz 2008, Brakebill et al. 2010), organic carbon (Shih et al. 2010), and dissolved solids (Anning et al. 2007, Kenney et al. 2009, Anning 2011, Anning and Flynn 2014). A number of applications have been derived from these models, including the determination of contaminant loadings to estuaries and the Great Lakes (Alexander et al. 2000, Robertson and Saad 2011), the identification of streams with a high probability of violating water-quality standards (Smith et al. 1997), the estimation of background nutrient loadings (Smith et al. 2003), the attribution of contamination to individual sources (Alexander et al. 2008, Robertson and Saad 2011), the prioritization of basins for management (Alexander et al. 2008, Robertson et al. 2009), and the forecasting of future water-quality conditions (Roberts et al. 2009, Robertson et al. 2016).

A recent application (Garcia et al. 2016) used SPARROW in conjunction with USDA Conservation Effects Assessment Project (CEAP) results to empirically assess the effectiveness of agricultural best management practices in affecting total nitrogen and phosphorus loads in rivers and streams of the Upper Mississippi River Basin. The analysis was built on the previously developed SPARROW model for the Great Lakes, Upper Mississippi, Ohio and Red River Basins (Robertson and Saad 2011), which contained numerous spatial variables germane to the determination of instream loads, but lacked spatially detailed information on management practices to include in the model specification. To correct for the lack of spatial data, the Garcia et al. study used 4-digit hydrologic unit estimates of the relative change in edge-of-field loadings attributable to implemented management practices. Practice levels were estimated using the Agricultural Productivity and Extension (APEX) farm-scale model simulation (NRCS 2012) of actual farm management practices reported in the National Resources Inventory (NRI)-CEAP Cropland Survey (Goebel 2012).

Without the management variable, the original SPARROW model predicted with-management loads without any spatial specificity as to where management actions were taken. The inclusion of the management variable in SPARROW allowed the model to be sensitive to spatial variations in management intensity. The empirical finding that the management variable was statistically significant for the nitrogen model implied that the management effect could be isolated using the ambient loadings data across monitoring stations. The phosphorus model showed the management variable to have the expected effect but the signal was not statistically significant; the magnitude of the effect was not large enough to overcome the amplified noise in the loadings data symptomatic of this contaminant.

The study used the uncertainties of the estimated effect of management actions to derive a likely range of stream contaminant reductions caused by such actions. As could be expected based on results from model estimation, the ranges of nitrogen reduction across sub-regions of the study area were unambiguously positive, and generally consistent with the CEAP estimated effects. Conversely, the enhanced uncertainty inherent with the phosphorus model resulted in ranges that could not definitively demonstrate a management effect on water quality, with CEAP estimates consistently overstating the upper bound of the reduction range.

Future improvements to the approach will be possible by further refining the APEX estimate of edge-of-field relative change in load due to management actions. The existing APEX characterization of management actions includes nutrient reduction. This creates a problem for the Garcia et al. (2016) SPARROW analysis because the original SPARROW model without the management variable controls for nutrient sources as actually applied, which presumably reflects conditions inclusive of nutrient reduction. Therefore, nutrient reduction is effectively accounted for twice in the model, both as affecting the source loadings and as affecting the delivery of loadings to streams. A better measure of management actions would be to determine the relative load reduction due to management actions other than nutrient reduction, a measure that would require APEX simulations that exclude the effects of nutrient reduction. Efforts are currently underway to include a refined APEX-derived management variable in a regional SPARROW model.

Appendix B—Basis of Property Owner Benefits of Improved Water Quality

This will be an indicator of one group of potential beneficiaries of the improvement in water quality. At a minimum, it would be the number of residential units and residential lots physically bordering the lakeshore or riverbank. However, if there are public access points to the rivers and lakes (even if restricted to homeowners in a home owner association, or HOA) then there are still beneficiaries that are not adjacent to the lake or river. These benefits will be capitalized in their house prices and other parcels of undeveloped residential land as well. As discussed in more detail in Section 3, property value enhancements may extend in as much as a half mile (Berjrandonda et al. 1999). There is usually a distance-decay function, where the property value effects diminish the further a house parcel is from the lake or stream. Exactly how far to go inland in counting residential units will be determined in part by the hedonic property valuation study that the analyst will apply and in part by the nature of the water resource, including accessibility by potential users. If it is a purely private lake or river with access limited only to bordering landowners, then the number affected are those living along the water body. The number of apartment units can also be counted if they are close to the water and are allowed public access to the water body. The number of businesses likely to be enhanced by improvements to water quality could also be counted but listed in a separate category (e.g., restaurants, hotels, and water-related businesses such as boat rentals).

Once the spatial dimensions of the property market have been defined, data on the number of residential units can be obtained from the U.S. Census American Fact Finder. More fine resolution data can be obtained from GIS analysis or aerial photos. County assessors' offices often maintain databases of residential units for property tax purposes. Real estate brokers and companies often have access to databases with information on the number of properties and sales or assessed values as well. If there are homeowners' associations, these too may be a good source of data on the number of homes bordering a lake or stream. A best practice would be to combine multiple sources of information to cross validate information on numbers of home units and their values. This would be particularly important if there are condominiums along a lake or stream, as it is difficult to determine, purely from secondary sources, how many units there may be in a building. By itself, this BRI will primarily reflect use values (e.g., water recreation access, wildlife viewing, scenic beauty).

Property value effects of improved water quality

The approach used to estimate the property value increases of environmental improvements is the hedonic property method. This method is based on the idea that the value of a house can be decomposed into the value of its individual characteristics—where it is in relation to workplaces and amenities, how many bedrooms and baths it has, how large it is, and the nature of its surroundings, including environmental attributes such as water quality (Freeman et al. 2014). This method has been extensively used to estimate the value that households place on improving air and water quality. In the case of water quality, it has been used to compare house prices on shorelines with poor water quality to those with good water quality (Leggett and Bockstael 2000, Boyle et al. 1999, Poor et al. 2007, Klemick et al. 2015, Walsh et al. 2015).

The basic form of a hedonic property price function is:

House Price = f*(structural characteristics of the house, neighborhood characteristics such as schools and crime rate, environmental characteristics such as air quality)*

Equation B1. Hedonic property price function

For example, we might have the following:

$$\textbf{House Price} = B_0+B_1(\text{House Size}) + B_2(\text{House Age}) + B_3(\text{Miles to Work}) + B_4\ \text{Water Quality} + B_5\text{Distance to Water Body}$$

Equation B2. Hedonic property price function example

By collecting data on houses from several areas within a town across towns in which the water quality varies, a separate coefficient on water quality (WQ) can be estimated using multiple regression. This coefficient (B_4) gives the present value of the household's willingness to pay for a one unit change in water quality (B_0 is a constant term for the real estate market). The value of a given increment can be approximated by inserting the new level of water quality, then determining the current or future without program level of water quality, and then finding the difference in house price. Like the use of any regression equation, the accuracy of applying the model to project home values for a novel change in water quality is higher when modeling water quality changes that are like the conditions in the observed data.

The analyst can also transfer results from a regression equation by converting B_4 into a percentage change in value with a one percent change in water quality. Then he/she can multiply an agricultural conservation project's percentage change in water quality times the percentage change in house value from the regression to estimate an overall total percentage change in house value associated with the agricultural conservation practice. This total percentage change in house value is applied to the average value of a house in that area.

The aggregate value of the improvement in water quality would be calculated by multiplying the change in house price by the number of residences affected by the change. If the value of environmental quality changes with the household's distance from a point source of pollution or natural resource, then this needs to be explicitly accounted for in the estimation by including a distance variable (e.g., Leggett and Bockstael 2000). Then, a spatial valuation gradient would be used to more accurately arrive at the total value to residences in the area. The key to applying this method to water quality is identifying the salient water quality characteristics that influence property prices. It may be water clarity as determined by Secchi disk readings (Boyle et al. 1999) or total suspended sediment (TSS) (Poor et al. 2007). The choice of which measure to use may be based on available data, discussion with real estate agents, or statistical criteria such as the relative statistical significance of the water quality variable and its contribution to goodness of fit (e.g., adjusted R^2 of the equation).

If the USDA expects to make multiple improvement projects over time that will improve water quality at a stream, river or lake, then it would be worthwhile for the USDA to conduct an original hedonic property method (HPM) value study, if one does not already exist. This would be the most accurate approach and represent a good investment in terms of being able to calculate property value benefits for an extended period (e.g., a decade). However, if this water quality improvement effort is a one-time project in a watershed, then it may not be cost-effective to estimate a new water body specific hedonic property model. If it is not possible for the USDA analyst to conduct an original HPM study for the local watershed experiencing water quality improvements, then a benefit-transfer approach from the existing literature can be performed. Benefit transfer can either be a point estimate transfer (e.g., in this case, a percentage change in property values with a one foot increase in water clarity, calculated from B_4 in the hedonic equation), or a benefit function transfer by applying the entire existing hedonic property equation to the location (e.g., water body and residential units surrounding it) where the water quality is to be valued. Benefit function transfer includes the use of meta-analysis benefit functions as well (Klemick et al. 2015). Benefit function transfer is often more accurate (Rosenberger and Loomis, forthcoming). The reason for the improved accuracy is that inserting the local values of all the variables in the transferred hedonic price equation (Equation 1) will cause the resulting calculated value of the houses affected by the policy change to better reflect the housing characteristics in the study area.

Empirical meta-analyses

In our case study, we were fortunate to find an original hedonic property study of Lake Erie that could be applied to the western shore of Lake Erie. There were two papers that used meta-analysis to evaluate improvements in water quality. One, by Klemick et al. (2015), used a meta-analysis equation for 14 Maryland counties around Chesapeake Bay. The original hedonic property model included a variable for water clarity. The measure chosen was a measure of light attenuation or light diffusion, referred to as KD. The higher the KD, the cloudier the water (i.e., lack of water clarity).

This variable was negative and statistically significant for waterfront houses. A 10 percent reduction in KD resulted in a 0.33 percent to a 1.5 percent increase in property values. Houses that were not waterfront but were within a quarter mile of the lake had a smaller effect—a 10 percent reduction in KD results in a 0.2 percent to a 0.6 percent increase in property values. The authors used this hedonic property model to estimate out of sample benefit transfer error for applying the model to other counties in the states of Delaware and Virginia, and to Washington DC. The absolute value of the meta-analysis benefit transfer errors of eight percent to 13 percent for waterfront homes to 13 percent to 18 percent for homes within a quarter mile of the bay. This should provide analysts with some rough estimate of the degree of error with benefit transfer for a meta-analysis equation when the geographic extent of the transfer is relatively small (i.e., the area of the original valuation is reasonably close—less than 200 miles—to the area to be valued with the benefit transfer). Thus, this is a reasonably small error by benefit transfer standards.

For the state of Maine, Boyle et al. (1999) estimated increased property prices with an increase in water clarity measured in meters of depth calculated using a Secchi disk. The researchers used data from 25 lakes in Maine, and focused just on lakefront properties. The mean size of the lake was large at 3,154 surface acres. These authors found, in the linear model, that each meter increase in lake water clarity has a value of $9,400 per property in 1995 dollars. But the value depended on the community, with an additional meter being worth as little as $2,337 in Bangor, Maine to as much as $12,938 in Camden, Maine.

One of the biggest challenge is linking the biophysical units of water quality improvements from farm management practices to the water quality variables represented in the hedonic property model. While this is a difficult challenge to overcome, it is not impossible. If the water quality specialists can communicate the units that the water quality improvement will be measured in, then that data can be included as the measures of water quality in the hedonic property model, or a more human sense perceptible proxy can be developed that maps biophysical water quality parameters such as sediment, nitrogen, and phosphorous. The SPARROW model may provide such a linking model (Schwarz 2016).

Western Lake Erie Basin Case Study

For this illustrative analysis of property values, the focus is on the Maumee River and the western shore of Lake Erie. To focus on where many benefits are likely to come from, we emphasize significant towns with housing along either the Maumee River or the western shore of Lake Erie.

Benefit relevant indicator

Maumee River: The benefit relevant indicator is the number of single family homes along the rivers in the watershed. For purposes of illustration, we choose the Maumee River, and a sample of towns. In a quantification of the total value of ecosystem services to property owners, all houses within a half mile of all the rivers/streams affected by the four site conservation practices would be counted. For this exercise, we used the U.S. Census Fact Finder and took the total number of single family homes and town homes. Without having GIS capability, some of these homes may be more than a half mile from the Maumee River, hence overstating the benefit relevant indicator for these towns. Table B-1 presents the number of homes in a sample of towns along the Maumee River.

Table B-1. Number of homes in selected towns along the Maumee River that would benefit from improvements in water quality.

Towns	# of Single Family / Townhomes
Defiance	5,869
Florida	95
Independence	2,776
Maumee	5,600
Napoleon	9,003
Waterville	1,820
Total	25,613

Western Lake Erie: The number of homes in selected towns along the western shore of Lake Erie that would be affected by the four conservation practices is presented in Table B-2. Without GIS capability, some of these housing units might be beyond a half mile from Lake Erie, so the numbers in Table B-2 might overstate the number of beneficiaries.

Table B-2. Number of homes in selected towns bordering Western Lake Erie that would benefit from improvements in water quality.

Towns	# Single Family / Townhomes
Bay View	324
Huron	2,930
Marblehead	839
Sandusky	2,748
Vermillion	4,631
Total	11,472

Economic valuation of improved water quality on property owners

As the benefit relevant indicator suggests, there are two geographic areas affected by the water quality improvement. One is those along streams/rivers (e.g., the Maumee River) and along the west shore of Lake Erie. Each of these has a separate real estate market, and hence separate hedonic property models.

Hedonic property model of river/stream water quality

There appear to be no hedonic property studies for the towns along the Maumee River or rivers and streams draining the study area (Western Lake Erie Basin). However, there is a hedonic property study of water quality improvements in the Upper Big Walnut Creek watershed that covers Franklin and Delaware counties. These two counties are just south of the Western Lake Erie watershed, centered around Columbus, Ohio. As such, the valuation of houses in the Maumee River area using the property value improvements in the Upper Big Walnut Creek watershed provides a good illustration of the benefit transfer approach. Given the proximity of the two watersheds, the benefit transfer should be better than if the watersheds were in different states. The benefit transfer is also good insofar as pollution of the watershed is largely driven by the large amount of agricultural land in the watershed (Liu et al. 2014). However, the demographics and house prices in the two areas are quite different. The average house price in the two counties is $313,553 (Liu et al. 2014) versus $123,785 (the average of the median house price per town shown in Table B-1, Source: American Community Survey). Given the differences in house prices, we need to scale Liu et al.'s marginal implicit prices to our study area house prices. Liu, et al. report that their results are in absolute dollars of house price changes rather than a percentage of house price as is sometimes reported. However, we can simply calculate the percentage changes in house prices from the Liu, et al. study by dividing the changes in implicit prices (usually calculated for the average house price) by the average house price. As such, the values are:

- 1% change in mg/L nitrogen improvement = 2.46% of house price
- 1% change in mg/L phosphorous improvement = 8.8% of house price
- 1% change in water clarity (Secchi depth) 0.31% increase in house price

Application of these results to the western Lake Erie watershed study area requires knowing how changes in the four agricultural conservation practices used for the site reduce mg/L of nitrogen and phosphorous in the Maumee River, and the increase in clarity of the river. There is baseline information for mg/L for phosphorous for the Maumee River and Auglaize River in the Western Lake Erie Basin (Ohio EPA 2014). However, Keitzer, et al (2016a) provide a baseline water quality and three levels of water quality improvements associated with different geographic extents of agricultural conservation practices and conversion of agricultural land to grassland as might be done with CRP. Keitzer, et al (2016a) does this for the streams in the western Lake Erie basin watershed.

The agricultural conservation practices included erosion control via filter strips, cover crops, and tillage management. Nutrient management consisted of changing the timing and quantity of fertilizer application. The two levels of agricultural conservation practices are: (a) treatment of just critical acreage (e.g., farm fields closest to streams)—this totaled eight percent of the watershed; and (b) treatment of critical and moderate acreage—this totaled 48 percent of the watershed (Keitzer et al. 2016a) of all agricultural lands in the basin.

The overall results of the three different amounts of acreage treated are presented in Table B-3. Treating critical and moderate amounts of farm acres (about half the farmland in the watershed) results in nearly a 25 percent reduction in concentration of total nitrogen (TN) in the streams and a 30 percent reduction in total phosphorous (TP).

Table B-3. Percent reduction in total nitrogen and total phosphorus for different amounts of the watershed treated

Extent of Agricultural Conservation	T Nitrogen	T Nitrogen	T Phosphorus	T Phosphorus
	mg/L	*% Change*	*mg/L*	*% Change*
Baseline	8.75		0.25	
Critical farm acres treated (8% of cropland)	8.17	6.6%	0.23	8.0%
Critical and Moderate Farm acres treated (48% of cropland)	6.18	31.0%	0.16	38.4%
All farmland in the watershed treated	5.4	43.6%	0.12	63.4%

To calculate the economic value of this reduction in TN and TP, we use the results of Liu et al. (2014) which showed that each percentage point reduction in TN and TP increased house prices by 2.46 percent and 8.8 percent, respectively. The application of the same percentage changes in value to increasingly large percentage reductions in TN and TP highlights one of the hazards of point estimate benefit transfer. It is likely that the percentage change in value for each one percent reduction in TN and TP is not constant across such a large range of percentage reductions in TN and TP. Best practice would be a benefit function transfer, which would involve applying each level of the alternative percentage reductions in TN and TP to their respective coefficients in the original hedonic property regression model. This would predict a new level of house price with each level of reduction in TN and TP, individually or in combination. Then the analyst would use the predicted changes in the house price to recalculate the percentage change in house price with each level of percentage reduction in TN and TP. This percentage change in house price would then be applied to the median value of houses in the geographic area needing values. This best practice benefit function approach would pick up any non-linearity in the original hedonic residential property value model. For purposes of illustration in the Maumee River example, we are using the simpler point estimate transfer approach, but if this was an actual benefit cost analysis then best practice benefit function transfer should be used if possible.

Table B-4 presents the application of these marginal values to a selected number of towns along the Maumee River. This is done to illustrate the application of the HPM to valuing water quality improvement from agricultural conservation practices. To calculate the total aggregate value of the improvements to property owners, data on the number and value of all houses along all the affected streams would have to be collected. For purposes of illustration, we assume that Maumee River is one of the regions with critical areas.

Table B-4. Increase in property values of houses in selected towns along the Maumee River from improvements in TN and TP.

Extent of Agricultural Conservation (scenario)	*TN % Change*	*% Change in home value per % change TN*	*Change in House Value ($) due to TN changes*	*TP % Change*	*% Change in home value per % change TP*	*Change in House Value ($) due to TP changes*	*Total House Value Change ($)*	*Benefits ($Millions) for 25,613 houses*
Value per house			$123,785			$123,785		
Critical Farmland (8% cropland)	6.6%	2.46%	$201	8%	8.8%	$871	$1,073	$27
Critical and Moderate (48% cropland)	31.0%	2.46%	$943	38.4%	8.8%	$4,183	$5,127	$131
All Farmland	43.6%	2.46%	$1,327	63.4%	8.8%	$6,906	$8,234	$211

As indicated in Table B-4, even the benefits of our small subsample of towns (hence a small sample of houses) along just the Maumee River are substantial. With eight percent of the Maumee River watershed treated with agricultural conservation practices, the benefits amount to $27 million. With 48 percent of the Maumee River watershed treated, the benefits are substantial at $131 million. Of course, Table B-4 presents just three discrete changes in benefits of water quality associated with the discrete changes in practice implementation and water quality projections developed available in Keitzer, et al. (2016a). Many actual agricultural conservation project analyses may involve less than eight percent treatment of the watershed or treatment of between eight and 48 percent of the watershed. Therefore, the model of Keitzer, et al. (2016a) would probably need to be applied to calculate the estimated percentage reduction in TP and TN associated with the scale of the percentage of the watershed being treated. The resulting percentage reductions in TN and TP would then be inserted into the hedonic property value model to calculate the dollar magnitudes.

Of course, the costs are likely to be substantial as well, and a benefit cost analysis must account for both to determine the economic effectiveness of program expenditures. However, as the title of the Keitzer et al. (2016b) article suggests (*Thinking outside of the lake: Can controls on nutrient inputs into Lake Erie benefit stream conservation in its watershed?*), there are also benefits to Lake Erie that would need to be factored in. However, Lake Erie benefits could not be included due to a lack of data on water quality changes in the lake.

Appendix C – Sport fishing calculations

The following text provides more details of the data and assumptions used in calculations within the main report body.

Piscivore Index Analysis and Sources of Error

The Western Lake Erie Basin Case Study (Keitzer et al. 2016a) provides an example of the complexities involved in developing biophysical models that predict fish populations with water-quality data. The authors included stream discharge in their models to control for the effects of stream size—which is known to be one of many natural factors that have a profound influence on fish populations. Other natural factors such as stream gradient and hydrologic regime also control fish species distributions, as well as a host of other factors that are directly or indirectly influenced by land and water management actions, including habitat quality, water temperature, and the presence of migration barriers such as dams and diversion structures. Published accounts of models that predict the effects of nutrients on stream biota while accounting for the potential confounding influence of natural and anthropogenic factors are extremely rare. Yuan (2010) provides a promising approach for handling covariates, and found that increased nutrients reduced invertebrate species richness by as much as nine species (per unit increase in total nitrogen), but only in relatively large streams with limited shading. Yuan (2010) demonstrates that the influence of nutrients on stream fauna is dependent on the environmental context—defined as the interaction of natural and anthropogenic factors.

Estimating percentage of sportfish that were piscivores

Fish community biological monitoring data used in Keitzer et al. (2016a) were provided by the authors (personal communication) as a list of species and percent site occurrences. The authors compiled data from four state agencies for 841 sampling sites distributed throughout the Western Lake Erie Basin (Table C-1). Species in this list were classified as sportfish and piscivores. However, it was difficult to use data in Table C-1 to represent angler catch because they were developed using a mix of abundance surveys and creel surveys. As a result, we relied heavily on the creel survey for the Maumee basin (Ohio DNR 2016) and other data to estimate which species were being targeted by anglers, as required for the benefit transfer.

Table C-1. Sportfish species present in fish surveys of the Western Lake Erie Basin (C. Keitzer, pers comm).

Common	Scientific	%_of_Sites	Sportfish	Piscivore
Black bullhead	Ameiurus melas	12	Y	
Yellow bullhead	Ameiurus natalis	62	Y	
Brown bullhead	Ameiurus nebulosus	5	Y	
Rock bass	Ambloplites rupestris	37	Y	Y
Grass pickerel	Esox americanus vermiculatus	14	Y	Y
Northern Pike	Esox lucius	9	Y	Y
Channel catfish	Ictalurus punctatus	22	Y	
Green sunfish	Lepomis cyanellus	85	Y	
Pumpkinseed	Lepomis gibbosus	10	Y	
Orangespotted sunfish	Lepomis humilis	26	Y	
Bluegill	Lepomis macrochirus	56	Y	
Longear sunfish	Lepomis megalotis	18	Y	
Redear sunfish	Lepomis microlophus	2	Y	
White perch	Morone americana	3	Y	Y
White bass	Morone chrysops	3	Y	Y
Smallmouth bass	Micropterus dolomieu	26	Y	Y
Largemouth bass	Micropterus salmoides	41	Y	Y
White crappie	Pomoxis annularis	16	Y	
Yellow perch	Perca flavescens	5	Y	Y
Black crappie	Pomozis nigromaculatus	6	Y	
Flathead catfish	Pylodictis olivaris	6	Y	
Walleye	Sander vitreus	2	Y	Y

Table C-1 shows a diversity of species present in the Western Lake Erie Basin, but the Ohio DNR creel survey of the Maumee River (Ohio DNR 2016) only reports on walleye and white bass catches. They report that anglers are primarily targeting walleye when they fish in the region and that white bass are generally not target species, "…the majority of white bass were harvested as incidental catch from anglers targeting other species." (Ohio DNR 2016).

To overcome the limited information on species-level recreational harvests, we estimated the change in recreational piscivore catch solely in terms of walleye and white bass because they were well represented in recreational fishing data. We applied the recreational fishing effort data collected for Lake Erie to estimate the relative proportion of time that these species were caught (Table C-2). Data were directly available for walleye effort, an "anything" category of effort was assigned to non-piscivores, and the remaining effort (36 percent) was assigned to white bass, which was intended to represent a catch-all category for piscivore species other than walleye. The percentage of time fished per species was multiplied by total fishing days that had been estimated from Ohio survey data (see main text body and USFWS 2011b) to estimate species-specific harvest for the Western Lake Erie Basin, which excludes Lake Erie (Table C-3). Average fish caught per species were derived from the Sandusky and Maumee Rivers survey data (Ohio DNR 2016, Table 4.2.2).

Since walleye were included in both the benefit transfer function and the creel survey, catch and change in catch could be directly incorporated in benefit assessment. White bass were the other target species included in the creel survey, but were not included in the transfer function. Since white bass are rarely a target species (Ohio DNR 2016), we assumed that their value was like that of panfish, which also tended to be incidental catch and were included in the transfer function. Since changes in non-piscivore fish had not been modeled, they were not included in the benefit assessment. Estimates for total catch by fish species are shown in Table C-3. The estimate of total fish caught using these data was 9.1 million. These estimates involved many assumptions that likely introduced substantial error into estimated total catch.

Table C-2. Private boat angler hours for target species in 2015

	Lake Erie
Total fishing hours	3,298,706
Walleye fishing hours	2,081,168
"Anything" fishing hours	32,736
% time walleye	63%
% time "anything"	1%
% time other piscivores (remainder)	36%

(Data source: Ohio DNR 2016, Table 4.1.6)

Table C-3. Estimates of total fish catch (BRI) for target sportfish species in the Western Lake Erie Basin

	Fishing days per species (millions/year)	Average fish caught (fish/day)	Total fish caught (millions/year)
Walleye	1.81	1.24	2.25
White Bass	1.03	6.16	6.36
Non-Piscivores	0.03	16.01	0.46
Total	2.87		9.10

BRI estimates – total fish catch for target sportfish species in the WLEB

To estimate total catch by fish species we applied data on percent effort for three species groups, walleye, other piscivores (represented by white bass), and non-piscivores. We develop estimates of catch per day and total catch for piscivores from the Maumee creel survey (Ohio DNR 2016) to support valuation. Estimates for non-piscivore catches were not needed for valuation and were omitted. Data sources and results are shown in Table C-4.

Table C-4. Calculations of baseline total fish catch by species

	Walleye	White bass (mixed piscivores)	Data source or calculation
Catch per hour (fish)	0.31	1.54	Ohio DNR 2016, Table 4.2.1
Catch per day (fish)	1.24	6.16	Catch / hour * 4 hours / day
Proportion of effort	0.63	0.36	Calculations from Table C-2. private boat angler hours
Total fishing days (millions)[1]	1.81	1.03	*Proportion of effort* * 2.873 M total days in Western Lake Erie Basin[1]
Total fish (millions)	**2.25**	**6.36**	*Total fishing days * Fish per day*

[1] Calculations were described in main body of text. 2.873 M days is the product of 17% * 16.9 Million Ohio annual fishing days. WLEB is 17% of Ohio's area and a total 16.9 Million days was reported for Ohio by USFWS (2011b).

Monetary Value estimates

Benefit transfer calculation of WTP for fish catch increase

Conducting the benefit transfer required choosing values to "plug in" for variables representing the case study region and setting other variables to the mean of studies within the metadata, to overcome data limitations (detailed methods are described in Johnston and Wainger 2015). Coefficients from the meta-regression analysis are then multiplied by the variable values and these products are summed to generate a forecast of the natural log of WTP per additional fish caught by recreational anglers.[22] Taking the anti-log of this result (e^x) yields an estimate of WTP.

The average annual household income for Ohio resident anglers was $58,000 in 2011, based on the FHWAR (USFWS 2011b).[23] The catch per trip (cr_nonyear in Table C-5) was estimated as 2.3 fish, using original study data and additional recent papers as demonstrated in Mazzotta et al. (2015). The Ohio creel survey found that fishing trips in the Western Lake Erie Basin are usually from shore (not from boats), which was also used in the transfer function. All other variable values in Table C-5 represent sample means. To complete the value transfer, total fishing effort, in days, was calculated as the sum of days fished by resident and non-resident anglers in Ohio (USFWS 2011b), as described in the main body of the text. By default, the model calculates the value per additional panfish, unless one inserts a value of "1" for one of the other species and region variables. [24] Plugging in a value of "1" for pike_walleye, in contrast, yields a parallel estimate for pike, walleye, or similar species.

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[22] We also add the term ${\hat{\sigma}^2}/{2}$ to account for log transformation error (Stapler and Johnston 2009).

[23] Household income was adjusted to $47,440 in 2003 dollars for use in intermediate transfer function calculations (HH_inc_thou, Table C-5).

[24] Panfish is the omitted dummy variable category in the meta-regression model. So, by plugging in values of zero (0) for all the species and region dummy variables in the model, the model, by default, predicts a value (per fish) for panfish.

Table C-5. Calculation of benefit transfer function

Variable	Coefficient	Assignment (walleye)	Product (walleye)	Assignment (panfish)	Product (panfish)
Intercept	-1.457	1	-1.457	1	-1.457
SP_conjoint	-1.167	0.044	-0.051	0.044	-0.051
SP_dichot	-0.996	0.174	-0.173	0.174	-0.173
TC_individual	1.109	0.107	0.119	0.107	0.119
TC_zonal	2.048	0.041	0.084	0.041	0.084
RUM_nest	1.332	0.235	0.314	0.235	0.314
RUM_nonnest	1.789	0.304	0.544	0.304	0.544
sp_year	0.088	4.604	0.403	4.604	0.403
tc_year	-0.040	0.732	-0.029	0.732	-0.029
RUM_year	-0.003	9.373	-0.027	9.373	-0.027
sp_mail	0.544	0.051	0.028	0.051	0.028
sp_phone	1.086	0.130	0.142	0.130	0.142
high_resp_rat	-0.654	0.358	-0.234	0.358	-0.234
HH_inc_thou	0.004	**47.440**	0.184	**47.440**	0.184
age42_down	0.921	0.097	0.089	0.097	0.089
age43_up	1.222	0.271	0.331	0.271	0.331
trips19_down	0.839	0.110	0.092	0.110	0.092
trips20_up	-1.011	0.335	-0.339	0.335	-0.339
nonlocal	3.236	0.005	0.017	0.005	0.017
big_game_pac	2.253		0.000		0.000
big_game_natl	1.532		0.000		0.000
big_game_satl	2.382		0.000		0.000
small_game_pa	1.623		0.000		0.000
small_game_at	1.410		0.000		0.000
flatfish_pac	1.891		0.000		0.000
flatfish_atl	1.380		0.000		0.000
other_saltwater	0.734		0.000		0.000
musky	3.867		0.000		0.000
pike_walleye	1.041	**1**	1.041		0.000
bass_small/largemouth	1.778		0.000		0.000
trout_GreatLakes	1.872		0.000		0.000
trout_nonGreatLakes	0.863		0.000		0.000
salmon_pacific	2.357		0.000		0.000
salmon_atlantic	5.269		0.000		0.000
salmon_GreatLakes	2.214		0.000		0.000
steelhead_pacific	2.190		0.000		0.000
steelhead_GreatLakes	2.339		0.000		0.000
baseline_catch/day	-0.081	**1.24**	-0.101	**6.16**	-0.501
cr_year	-0.052		0.000		0.000
catch_year	1.269		0.000		0.000
spec_cr	0.686	1	0.686	1	0.686
fish_from_shore	-0.113	**1**	-0.113	**1**	-0.113

Table C-6. Calculation of benefit transfer function (cont.)

	Product (walleye)	Product (panfish)
Sum	1.550	0.109
Sigma	0.658	0.658
WTP (2003$)	6.55	1.55
WTP (2015$)	13.95	3.30
Baseline recreational harvest 2015 (millions of fish)	2.25	6.36
Change in population (42% of piscivores)	0.94	2.67
Change in value of recreational fishing (millions $2015)	**13.17**	**8.81**
GRAND TOTAL		**21.98**

Sensitivity analysis

The sport fishing benefit calculations are sensitive to many variables that were poorly constrained in this analysis. As a result, the estimates could change markedly under different assumptions, most notably, the number of fish caught in the baseline assessment (Table C-7). Since the number of fish caught was calculated as the product of fishing days per year and average fish per day, both of these uncertain values can affect benefit estimates. Further, the meta-analysis estimates the value of one additional fish, *per angler, per trip*. So, if one multiplies the estimated per fish value by the number of additional fish caught, an implicit assumption is that the increase in catch is distributed among anglers such that the increase in catch is approximately one fish per angler per trip. However, in the case of a 42 percent increase in the population, it is reasonable to assume that catch rates would increase by an average of 2.6 fish per day for white bass and 0.5 fish per walleye (a 42 percent increase in baseline catch). To capture the diminishing marginal returns for white bass, one would fit the equation for the first fish and then refit for the second fish (by increasing the baseline catch rate), to more accurately estimate value. However, in the case study the estimated increase in catch has a more substantial effect on total benefits than value per fish (Table C-7). However, this may not always be true.

Table C-7. Sensitivity of sport fishing benefits to input calculations

Scenario	Total fishing days (millions)	Baseline walleye catch (fish/day)	Baseline white bass catch (fish/day)	% Increase in fish	% Increase in fish caught	Total benefits (M 2015$)
Baseline	2.84	1.24	6.16	42%	42%	21.98
50% fewer fishing days in Western Lake Erie Basin	**1.42**	1.24	6.16	42%	42%	10.99
50% of piscivore increase is caught	2.84	1.24	6.16	42%	**21%**	11.92
catch rate + 2	2.84	**3.24**	**8.16**	42%	42%	20.66

Note: bold values show variable that were changed per scenario

Appendix D – Reservoir and Navigation Channel Ecosystem Service Benefits

Table D-1. Costs avoided for reservoirs and navigation channels due to reduced sediment erosion by 8-digit HUC within the Western Lake Erie Basin ($/ton of field erosion).

HUC-8	Reservoirs ($/ton)	Navigation ($/ton)
04100001	0.04	0.05
04100002	0.23	0.06
04100003	0.20	0.04
04100004	0.003	0.03
04100005	0.02	0.06
04100006	0.04	0.06
04100007	0.09	0.05
04100008	0.07	0.05
04100009	0.07	0.07
04100010	0.03	0.07
Basin Average	0.08	0.05

Source: Hansen and Ribaudo 2008.

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