Linking Agricultural Nutrient Pollution to the Value of Freshwater Ecosystem Services

Frank Lupi *Professor, Department of Agricultural, Food, and Resource Economics and Department of Fisheries and Wildlife, Michigan State University, East Lansing, Michigan; lupi@msu.edu*
Bruno Basso *University Foundation Professor, Department of Earth and Environmental Sciences and W.K. Kellogg Biological Station, Michigan State University, East Lansing, Michigan; basso@msu.edu*
Cloé Garnache *Assistant Professor, Department of Economics, University of Oslo, Oslo, Norway; cloe.garnache@econ.uio.no*
Joseph A. Herriges *Professor, Department of Economics and Department of Agricultural, Food, and Resource Economics, Michigan State University, East Lansing, Michigan; jah@msu.edu*
David W. Hyndman *Professor and Chair, Department of Earth and Environmental Sciences and W.K. Kellogg Biological Station, Michigan State University, East Lansing, Michigan; hyndman@msu.edu*
R. Jan Stevenson *Professor, Department of Integrative Biology, Michigan State University, East Lansing, Michigan; rjstev@msu.edu*

ABSTRACT *This paper describes our efforts to integrate economic and biophysical models to evaluate the effects agri-environmental policies have on the value of freshwater ecosystem services. We are developing an integrated assessment model (IAM) that links changes in phosphorus-related management practices on farm fields to changes in the value of key freshwater ecosystem services, including biological condition, water clarity, species-specific fish biomasses, and beach algae. Our IAM approach enables examination of the effects of policies and conservation programs on ecosystem services and values. Results will help policy makers allocate conservation dollars to improve water quality, enhance ecosystem services, and promote more sustainable agricultural production.* (JEL Q24, Q51)

1. Introduction

The agricultural sector in the United States is essential to domestic and global production of food, feed, fiber, and fuel for humanity. Yet, agricultural production can also negatively impact the provision of ecosystem services important to society, including biodiversity, soil functions, climate regulation, recreation, and the supply of safe drinking water. In particular, agriculture has been identified as a key source of excess nutrient loadings to the nation's rivers, lakes, and coastal waterways (Michalak et al. 2013; International Joint Commission 2014). These excess nutrients, in turn, lead to water quality degradation, including groundwater contamination, algal blooms, hypoxic/anoxic conditions, and the loss of both fish biomass and native fish species (Carpenter et al. 1998). Effective and sustainable agricultural policies must weigh the benefits of nutrient use to enhance crop yields against its environmental costs. Policy instruments should be designed to encourage producers to consider such tradeoffs (Garnache et al. 2016).

Historically, federal water pollution control policies have largely focused on point sources, such as municipal waste treatment plants and pulp and paper mills. Congress enacted the Clean Water Act of 1972 to regulate nutrient pollution from point sources while exempting nonpoint sources and delegating their regulation to states. States have generally opted for voluntary programs for nonpoint source pollutants, relying on financial incentives to encourage farmer adoption of best management practices that directly impact fertilizer use or indirectly impact nutrient runoff, such as using cover crops, filter strips, and conservation tillage (Ribaudo 2009). Unfortunately, these programs are often less effective than intended (Kling 2011; Ribaudo 2009; Shortle

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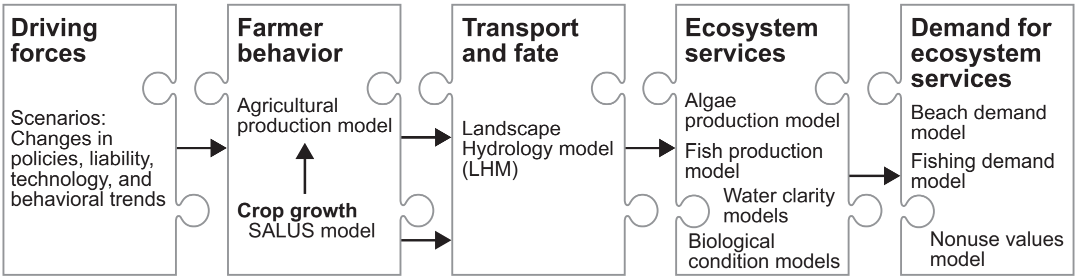
Figure 1

Linkages between the models to trace P from farm to the value of ecosystem services.



Source: Adapted from Garnache et al. 2016.

et al. 2012; and more generally, Keiser, Kling, and Shapiro2019).

Many studies have examined the effects of conservation programs on land use and farmer adoption of best management practices (e.g., Cooper 2003; Liu and Lynch 2011; González-Ramírez and Arbuckle 2015), while others have focused on relationships between water quality and recreation demand (e.g., Michael, Boyle, and Bouchard 2000; Phaneuf, Herriges, and Kling 2000). There are comparably few studies that have directly linked the effects of conservation programs to the endpoint ecosystem services that consumers value. González-Ramírez and Arbuckle (2015), for example, estimate the impact of a cost-sharing program on cover crop acreage, but do not trace the downstream effects of these changes on nutrient loadings and the ecosystem services that are ultimately valued by consumers. Van Houtven et al. (2014), on the other hand, estimate the economic value of ecosystem service changes due to a specific nutrient-loading scenario, but do not model the policies and farmer behaviors that induce the loading scenario. Our research seeks to take these approaches one step further, completing the linkage from policies through to key ecosystems services and the demand for these services by consumers.

Although agricultural nutrient pollution remains the major cause of stream and lake impairment in the United States and conservation programs incentivize farmers to change behavior, little is known about the relationship between behavior changes on farms and the influence these play on the value of endpoint ecosystem services (Smith and Weinberg 2006; Hellerstein and Lohr 2020). Research connecting policies to outcomes of interest is essential to comprehensively evaluate the effectiveness of agri-environmental policies, allowing for direct comparison of each conservation program's costs and the corresponding benefits accrued to the consumers of ecosystem services. However, the task is not trivial (Garnache et al. 2016; Smith and Weinberg 2006).

Consider policies designed to mitigate environmental impacts of excess phosphorus (P) loadings, a limiting nutrient and common freshwater pollutant (U.S. Environmental Protection Agency 2009). Program evaluation requires understanding the farm level response to a given policy, along with the associated P loadings from the farm to surface water and groundwater, as both dissolved reactive P and total P, which includes particulate P. Phosphorus then moves into streams and lakes, where each form of P has differing potential for affecting algal growth and fish species composition and biomass. The growth of filamentous algae can lead to muck formation, which covers beaches and releases foul smells when it decomposes, diminishing valued ecosystem services. Modeling that links the various pieces of what Garnache et al. (2016) refer to as the "P puzzle" (Figure 1) is essential to evaluate the effectiveness of agri-environmental policies, and the end result would be an integrated assessment model (IAM) for agricultural pollution (Hamilton et al. 2015; Kling et al. 2017; Keiser and Muller 2017).

Of particular interest here are the steps needed to arrive at and connect with the last piece in the P-puzzle: valuing changes in ecosystem services implied by the sequence of effects characterized in the other puzzle pieces (Figure 1). To do so, we focus on those ecosystem services with values that derive from recreation activities, particularly beach recreation and fishing, and on nonuse values for water clarity, game fish abundance, and biological condition from a recent stated preference study. Many stated preference studies of water quality build off the pioneering work of Carson and Mitchell (1993) and use a "water quality ladder" and/or a water quality index (Griffith et al. 2012), though few can be directly linked to specific freshwater ecosystem services (Johnston et al. 2012). Similarly, recreation demand models have long been used to value both access to water resources (lakes, rivers, and streams) and individual water quality characteristics such as Secchi transparency, odor, or fecal coliform count (e.g., Bockstael, Hanemann, and Strand 1987; Egan et al. 2009; Hicks and Strand 2000). Most studies have sought to value specific shifts in physical water quality attributes (total nitrogen or P level) rather than the ecosystem services that follow from these attributes (e.g., fish biomass or species composition) or to value broad changes in water quality (e.g., from boatable to swimmable) without trying to link the changes back to underlying policies that induced the purported changes. This has changed in recent years, with a number of studies emphasizing the need to value ecosystem services rather than inputs to those services (e.g., Boyd and Banzhaf 2007; Boyd and Krupnick 2013). Some recent efforts have tied changes in the ecosystem services being valued to an underlying model describing the source of these changes (e.g., Van Houtven et al. 2014; Esselman et al. 2015; Melstrom et al. 2015).

The goal of this paper is to describe the development of an IAM for P that links policies designed to induce changes in farmer behavior to resulting changes in the economic value of key endpoint ecosystem services of the lakes, rivers, and streams in watersheds draining to the Great Lakes from Michigan's Lower Peninsula as well as the downstream coastal zones of Lakes Michigan, Huron, and Erie. Our IAM is designed to capture farmer responses to alternative policy scenarios that in turn dictate excess P leaving the farm and includes valuation models linking changes in algal growth to beach demand, changes in fish biomass to fishing demand, and changes in water clarity, fish biomass, and biological condition to nonuse values. Connecting these pieces of the P-puzzle together are models capturing essential biophysical processes, including the relationship between plant uptake and runoff, the transport and fate of P through groundwater and surface water, and relationships between P concentrations, algal growth, and fish biomass.

Our IAM facilitates evaluation of the changes in values of affected ecosystem services from existing conservation programs that incentivize changes in P use, conservation tillage, and cover crop adoption. The IAM can be used to quantify likely *ex ante* effectiveness of future policies or management options such as mandatory soil tests combined with restrictions on P application, taxes on excess P in the soil, or changes in farmer attitude (e.g., changes in social norms). *Ex ante* evaluation of agri-environmental policy costs and benefits will help policy makers and regulatory agencies identify and design effective programs to address P pollution.

This paper describes a multi-year modeling effort and represents many completed components along with some that are underway. The reason for presenting the approach at this stage is to illuminate the extensive process of assembling an IAM for agricultural policy that is both broad in scale (state-level) yet capable of evaluating changes at fine spatial scales (farm fields). The central contribution of this paper is the presentation of integrated modeling to characterize the chain of effects from an economic model of farmer response to a policy initiative through to economic models of consumer demand for ecosystem services; we also link these ecosystem endpoints with biophysical models of P transport and fate that impact the ecosystem services available to consumers. Of relevance to economists, all of the valuation efforts we discuss were explicitly developed and designed to connect to ecological production functions and link to agricultural nutrients. Of note, our approach leverages extensive original research that has been and is being conducted by our team,

Table 1
Processes, Relationships, and Key Variables of the Models

Process	Model	Relationships and Key Variables
Crop growth and yield	System Approach to Land Use Sustainability (SALUS)	Yield = f(weather, soil, management including P application)
Farmer behavior	Agricultural production model	P use = f(yield, prices, elasticities, regulations, subsidies)
Transport and fate	Landscape Hydrology Model (LHM)	P concentration = f(land use, hydrology, sources)
		Water temp = f(recharge, air temperature, land use)
Ecosystem services	Algae production model	Algae in water and on Great Lakes beaches = f(P concentration, nearshore habitat)
	Biological condition model	Biological condition = f(P concentration, natural factors)
	Fish production model	Fish biomass = f(water temperature, P concentration)
	Water clarity model	Water clarity = f(P concentration, natural features)
	Water safety model	Water safety = f(fecal coliform counts)
Demand for ecosystem services	Beach demand model	Value = f(algae in water and on beach)
	Fishing demand model	Value = f(fish biomass by species)
	Nonuse values	Value = f(water safety, clarity, fish biomass, biological condition)

Note: See also Appendix Figure A2 and Table A1.

rather than relying on approaches such as benefits transfer. Valuation researchers seeking to improve the connection of their efforts to implications of agricultural and land use policies can benefit from seeing how such studies can be placed within the context of an IAM for agricultural nutrient pollution.

2. Methods and Linkages

In this section, we describe how an integrated model of P fate and transport from farms to key endpoint ecosystem services can quantify the likely benefits of different agri-environmental policies. Our study area is spatially large, encompassing the Lower Peninsula of Michigan and its coastal shores with Lakes Michigan, Huron, and Erie (depicted on Appendix Figure A1), though our IAM approach can be applied to solve similar issues to other sites where model inputs are available.

We leverage existing models developed by our team and others to link P from farms to the value of key freshwater ecosystem services. The models depicted in Table 1 are being adapted to enable their linkages by matching their spatial and temporal scales and defining common stock and flow variables to track P transport and fate through various media and forms. To connect people to ecological implications of P fertilization, we focus on some recreational and some more general freshwater ecosystem services. For recreation, we address recreational fishing and Great Lakes swimming/beach use, both of which have over 1 million participants annually in Michigan alone (U.S. Fish and Wildlife Service 2013; Chen 2013). More generally, we also use metrics such as the biological conditions gradient that captures the naturalness of ecosystems, and we value this using contingent valuation.

In what follows we provide more description of our IAM by identifying how our modeling system handles the five key linkages (depicted in Figure 1) connecting farm policies through to their impacts on ecosystem services that consumers value.

Linkage 1: Connecting Farmer Behavior to Changes in P-related Management Practices

To understand how farmer behavior affects freshwater ecosystems, we are modeling farmer decisions that influence P dynamics in soils and the environment. Our modeling of crop production in river basins that drain to the Great Lakes from Michigan's Lower Peninsula captures farmer crop choices, P fertilization, and soil management practices, including tillage choices. Our modeling is using positive mathematical programming (PMP), a nonlinear programming method widely used for agricultural and environmental policy analysis (Johansson, Peters, and House 2007; Jenkins et al. 2001; Jansson and Heckelei 2011). PMP allows calibration of agricultural production models against available economic information with minimal data requirements, for example, regional land use and representative crop prices and agricultural management practices and input costs (Howitt 1995). PMP has several advantages over linear programming including the ability to replicate the reference allocation without artificial constraints that may impede the model's ability to respond to policy shocks, and to provide smooth responses to policy changes rather than step-wise responses from one corner solution to the next. These models can further be calibrated against observed economic behavior and econometric estimates of supply elasticities to ensure realistic price responses (Garnache and Mérel 2015). In addition, PMP models can be calibrated against information on yield responses to input use (e.g., yield response to P), an important advantage for the development of IAM and the *ex ante* evaluation of agri-environmental policies (Mérel and Howitt 2014; Garnache 2015; Garnache, Mérel, Howitt, and Lee 2017; Garnache, Mérel, Lee, and Six 2017). In agricultural PMP applications, yield response information may come from field data or biophysical models. In our case, we use a crop growth model called Systems Approach for Land Use Sustainability (SALUS, (described in more detail below) that we calibrated to a sample of crop farm fields in Michigan to generate yield information for alternative cropping inputs such as P use. The yield outputs from the crop growth model feed into the farmer behavior model. In return, the decision variables of the farmer behavior model feed back into the crop growth model to quantify the impact of crop and inputs choices on nutrient dynamics (e.g., the transport and fate of P).

The farmer behavior model is a work in progress, although several major components of the model development and data collection have been completed. Our specific application assumes farmers maximize profits subject to the yield technology, with the possibility for including shadow benefits to account for nonmonetary factors such as preferences for the environment (Ma et al. 2012), social norms (Chen et al. 2009; Yeboah, Lupi, and Kaplowitz 2015) and other factors as in Garnache et al. (Garnache, Mérel, Lee, and Six 2017). Our data sources include a mix of existing data and field-specific data we collected through a farmer survey. For general data on statewide and regional crop production, and especially for row crops, we drew upon data collected through the Michigan Agricultural Statistical Service. Data to instantiate the PMP calibration for supply elasticities were taken from existing literature and studies. Data on input costs were collected from published crop budgets from Michigan State University Extension.

An especially important component is yield and yield response to P and other management inputs such as tillage and conservation practices. In our case, the yield data were derived from the crop model in Linkage 2 for farm fields from our farmer survey. Our farmer survey was a combined push-to-web and mail survey of Michigan crop producers conducted in 2017 (unpublished data, Garnache et al.). The survey received a 30% response rate resulting in data for 1,730 crop farms in Michigan. Farmers provided data and locations for a specific field. Then, for each field in our survey, the crop model used externally available physical data such as weather, soil types, and slopes. Yield also depends on input choices by farmers regarding crop rotations, tillage, P and nitrogen inputs, cover crops, conservation program enrollments, among others, which we collected in the survey. The crop input data from the survey were used to predict yields at field levels under baseline conditions, and these data were then used in field-specific simulations under different weather scenarios and with differing input levels such as P applications. The resulting field-level yields and yield responses were then aggregated into average yield relationships and average farms for each of 12 crop regions in Michigan. Our PMP approach is being applied and developed for each of these regions. Once completed, we will then downscale the regional behavioral responses to policies to field levels for all fields in statewide crop data layers (and not just our sample fields) for the prediction of P leaching (Linkage 2).

Linkage 2: Connecting Changes in P-related Management Practices to Changes in P Transport from Fields

In our IAM, yield responses to P fertilizer and soil P availability are modeled with the SALUS model, which has been thoroughly tested against field measurements for soil carbon dynamics (e.g., Senthilkumar et al. 2009), crop yield (e.g., Basso et al. 2007; Asseng et al. 2013; Rosenzweig et al. 2013; Dzotsi, Basso, and Jones 2013), plant nitrogen uptake and phenology (e.g., Basso et al. 2010; Basso, Ritchie, et al. 2011; Basso, Sartori, et al. 2011; Basso, Kendall, and Hyndman 2013), nitrate leaching (Giola et al. 2012; Syswerda et al. 2012), water use efficiency (Basso and Ritchie 2012), and P dynamics (Daroub et al. 2003). SALUS is derived from the validated CERES models with added ability to quantify the impact of management strategies and their interactions with the soil-plant-atmosphere system on yield along with carbon, nitrogen, and P dynamics. It simulates continuous crop growth and soil, water, and nutrient conditions during growing seasons and fallow periods. SALUS requires input information on soil properties, climate, genotype, and agronomic management practices. As mentioned above, we used SALUS to create the yield responses to P in the farmer behavioral modeling, and for policy evaluations it will also be used to predict changes in nutrient leaching that feed into the hydrologic model.

Linkage 3: Connecting Changes in P Transport from Fields to Changes in P Concentration in Streams and Lakes

The Landscape Hydrology Model (LHM) is a fully distributed process-based code for high-resolution flow simulations over large domains (e.g., Hyndman, Kendall, and Welty 2007; Kendall 2009; Wiley et al. 2010; Luscz, Kendall, and Hyndman 2017). It couples vegetation with surface and subsurface hydrologic processes in an efficient and scalable fashion. Surface and subsurface modules in this code are optimized to simulate large domains at fine resolution with modest computational resources at grid resolutions such as 120 m (Wiley et al. 2010) and 90 m (Luscz, Kendall, and Hyndman 2017). LHM simulates (1) surface water storage and routing, (2) canopy and root zone, (3) deep unsaturated zone, and (4) saturated zone groundwater. Within these domains, process modules simulate hydrologic fluxes from evaporation and transpiration to infiltration, groundwater recharge, overland runoff, and groundwater flow. Recharge fluxes to groundwater are coupled to MODFLOW (Harbaugh et al. 2000), which is the most commonly used groundwater flow code. LHM inputs include hourly climate data (precipitation, temperature, solar radiation, relative humidity, and windspeed), soil hydraulic properties, aquifer properties (hydraulic conductivity and specific storage), Leaf Area Index from the MODIS satellite every eight days, land cover, topography, and boundary conditions (such as locations and elevations of large lakes). The code, which has been validated in multiple domains including Michigan's Lower Peninsula (Luscz, Kendall, and Hyndman 2017) and the High Plains Aquifer, has been shown to accurately predict stream flows within ~6% of annual precipitation with no calibration of parameters except the hydraulic conductivity of aquifer materials. As such, the model is well suited to predict the influence of changes in climate and land use on water fluxes.

The nutrient transport model consists of a high-resolution (30 m) source loading model and a paired statistical and process transport model (Luscz, Kendall, and Hyndman 2015, 2017). Surface- or subsurface-applied nutrients are transported based on the simulated hourly distribution of water that takes various paths, with uptake by plants simulated using transfer functions from SALUS, and sorption to sediments. LHM has been coupled with SALUS to adopt its plant growth, root water, and nutrient uptake algorithms (SALUS-LHM). Flows of water and nutrients feed from this model into the subsequent ecological models.

Because P from manure contributes approximately 34% of P loadings in row crop agriculture and nearly 100% of P loading from pastures in Michigan's Lower Peninsula (Luscz, Kendall, and Hyndman 2015, 2017), we may augment future versions of the farmer behavior model with a livestock economic model to capture major manure production sources and waste management practices. Although our baseline modeling efforts will not model livestock decisions, it will account for livestock loadings with a model that was already developed (Luscz, Kendall, and Hyndman 2015), allowing us to tease out the portion of P loading due to crop production.

Linkage 4: Connecting Changes in P (and Nitrogen) Concentrations in Streams and Lakes to Changes in Ecosystem Services

In this step we relate P concentrations to a wide array of measures of ecosystem services in inland lakes, streams and rivers, and the coastal zone of the Great Lakes (Appendix Table A1). The ecosystem services include biological condition, water clarity, beach algae, and species-specific fish biomasses. We selected these ecosystems services because they are known to generate economic value, and we expect different responses of these services to P pollution. For example, as P pollution is reduced, most measures of ecosystems services will improve, although some fisheries production may decrease for nutrient-tolerant game fish species (Esselman et al. 2015).

Where possible, we rely on existing models when they have been characterized fully in the literature or are available from authors. In other cases, we are developing or have developed new statistical models using data from a variety of sources, including extensive data from U.S. Environmental Protection Agency (USEPA) national and regional surveys as well as other personal and publicly available sources. In the case of *Cladophora* in the Great Lakes, we are using existing process-based models as well as statistical models to relate P concentrations to algal fouling of beaches. Parameters describing climate, geology, hydrology, lake morphometry, and bathymetry have and will be included in models to account for natural variability in ecological responses to phosphorus.

Biological Condition and Water Quality Metrics

Biological integrity of water bodies was measured by biological condition (sensu Davies and Jackson 2006 and U.S. Environmental Protection Agency 2016) and as called for in the Clean Water Act. Statistical models of biological condition response to P concentration were developed using data from the USEPA National Rivers and Streams Assessment (NRSA), the USEPA National Lakes Assessment (NLA), and the data from the USEPA Coastal Condition Assessment. These models use indicators of ecological conditions and site-specific modeling approaches in which natural variation in expected values of metrics in the reference condition are predicted and compared to measured conditions at sites (Cao et al. 2007; Tang, Stevenson, and Infante 2016; Liu and Stevenson 2017). Natural variation in metric values can be great among reference sites within an ecoregion (Tang, Stevenson, and Grace 2020). Specifically, using the sites where biological condition data were available, we first developed models of condition as a function of natural and anthropogenic factors (e.g., climate, soils, geology, habitat geomorphology, and land use) that would be readily available for any water body by using existing databases. We then applied these models to existing databases for streams, rivers, and lakes for Michigan's Lower Peninsula to predict condition for all waters in our IAM.

Metrics of biological condition for diatoms were used to characterize biological condition for rivers and lakes because they are available in both the NRSA and NLA, are highly sensitive to phosphorus, reflect both structure and function of biological condition, and are routinely used around the world to assess ecological condition (Stevenson 2014). Using NRSA and NLA data from the Great Lakes Region, we constructed two statistical models, one for streams and rivers and another for lakes. The models predict biological condition as a function of water body and watershed-scale factors. We then applied these models to predict biological condition in streams, rivers, and lakes of the Lower Peninsula of Michigan. For biological condition of the Great Lakes nearshore zones, we used the Oligochaete Trophic Index that was calculated for samples in the Great Lakes collected during the USEPA's National Coastal Condition Assessment.

Statistical models for water clarity were constructed and tested using data from the USEPA's National Lakes dataset using available data from the Lower Peninsula of Michigan. In these models, natural features such as size and depth of lakes regulate effects of P loading and concentration on Secchi disk depths. These estimated relationships provided model predictions and underlying linkages to P for all water bodies in our IAM. In contrast to lakes, modeling Secchi depth in rivers and streams required a transformation of turbidity to Secchi depth because turbidity is what was measured in the NRSA.

We also developed statistical models for fecal coliform that are used in some of the valuation models in Linkage 5, even though fecal coliform is not presently linked to P management. Because it might potentially be affected by livestock management options, we included fecal coliform in the valuation to preserve the option for additional linkages in future efforts.

Beach Algae

To address the effects of fouling of beaches by algae, the filamentous algae *Cladophora*[1] provides a good example of how multiple models are being used to characterize the response of ecosystem services to P concentration management. Beach fouling by algae is being related to P loading to nearshore zones of the Great Lakes surrounding the Lower Peninsula of Michigan using three models (some existing and some under development). The first model will use an ensemble of results from existing hydrodynamic nearshore zone mixing models to predict P concentrations based on P load to the lakes, which will be estimated using LHM (Linkage 3). The nearshore mixing models are available from NOAA (www.glerl.noaa.gov/res/glcfs). Phosphorus concentrations will then be related to *Cladophora* growth and nearshore accumulation while accounting for area of nearshore zones with enough light and suitable habitat for *Cladophora*. The second model will predict *Cladophora* accumulation (biomass/area) in the nearshore zone as a result of its growth rate (biomass/time), P concentration, water transparency, and lake depth in the nearshore areas with enough light and suitable habitat for *Cladophora*. There are two commonly used, and related, *Cladophora* accumulation models: the more detailed Auer and Canale model (Auer and Canale 1982; Auer et al. 1982) and the simplified version called the *Cladophora* growth model (CGM; Higgins et al. 2005; Higgins, Hecky, and Guildford, 2005). These models have been applied successfully in multiple regions of the Great Lakes (Malkin, Guildford, and Hecky 2008), particularly in Lakes Huron and Erie, and they are both candidates for our modeling. The third model will relate *Cladophora* accumulation to beach fouling using modeled nearshore accumulation, satellite maps of macroalgal habitat (Shuchman, Sayers, and Brooks 2013; Brooks et al. 2013), and data for beach fouling from the Michigan BeachGuard System (Michigan Department of Environmental Quality, www.deq.state.mi.us/beach/). These models will allow us to connect levels of algae in the water and on the shore of beaches to P loadings and classify beach algae in a way that matches the categories used in EPA's beach surveys and links directly to the measures of beach algae in our valuation models in Linkage 5.

Game Fish Biomass

For game fish, we developed models that relate P concentrations to biomass. For lakes, these models build upon the models for river species reported in Esselman et al. (2015). Specifically, game fish abundance in lakes and rivers has been measured through biological sampling by the Michigan Department of Natural Resources. Because the biomass data are not available for all water bodies, the measured biomass data were used to estimate ecological models that were then used to predict biomass at our water bodies of in-

[1] Beach fouling in the Great Lakes by *Cladophora* was a major problem for recreation in the middle to late 1900s (Taft and Kishler 1973; Konasewich, Traversy, and Zar 1975). Phosphorus pollution was identified as the major cause of *Cladophora* growth on the Great Lakes bottoms and the resulting beach fouling. As a result, P regulation reduced P pollution of the Great Lakes and reduced beach fouling by algae (Painter and Kamaitis 1987). In the late 1900s, invasive species changed the Great Lakes ecosystem and caused a resurgence of beach fouling by benthic algae (Higgins et al. 2005; Auer et al. 2010). Invasive dreissenid mussels are filter feeders that have changed the Great Lakes in ways that favor bottom-dwelling algae that foul beaches (see review in Auer et al. 2010). Despite the great changes in the Great Lakes caused by these mussels, P management continues to be recommended to reduce *Cladophora* accumulation and beach fouling (Kuczynski et al. 2016).

terest. For both lakes and for rivers, the estimation used boosted regression trees to define nonlinear functions that predicted biomass at a water body as a function of its characteristics (e.g., temperature, size, morphology, P concentrations) and of landscape-scale characteristics for the areas around the lake (e.g., geology, forest cover). The boosted regression trees allowed us to compute partial dependencies of the estimated biomasses on P and provide predictive relationships for our modeling. All boosted regression models were trained (i.e., estimated) on subsets of the data and cross-validated against hold-out samples. For inland streams and rivers, the game fish species modeled include brook trout, brown trout, bass, panfish, and walleye (Esselman et al. 2015), and for inland lakes species, models were developed for bass, panfish, yellow perch, and walleye (unpublished data, Esselman et al.). The species modeled were selected because they are key targets for recreational anglers and had sufficient data for modeling. For river and streams, species-specific biomass estimates were predicted for each individual river or stream segment and were then aggregated to the hydrologic unit code (HUC) 10 level for Linkage 5 below. For inland lakes, species-specific biomass estimates were predicted for 1,615 individual inland lakes in Michigan (described further in Linkage 5).

Linkage 5: Connecting Changes in Fish Species Composition and Fish Biomass to Changes in the Value of Recreational Fishing, Changes in Algae at Beaches to Changes in the Value of Beach Recreation, and Changes in Metrics such as Biological Condition to Nonuse Values

Recreational Fishing

For recreational fishing, the above-mentioned species-specific biomass estimates that were linked to P have been linked to recreational site choices and values at rivers (Melstrom et al. 2015) and lakes (Klatt et al. 2017). For rivers, the Melstrom et al. (2015) demand model includes 232 watershed sites defined at the HUC 10 level and relates angler site choices to biomass for brook trout, brown trout, walleye, bass, and panfish. For inland lakes, the Klatt et al. (2017) model links angler site choices to species-specific biomass of panfish, bass, walleye, and yellow perch for 1,615 inland lakes in Michigan (for all of the 1,157 lakes 100 acres or larger and for the 458 lakes 10–100 acres visited by at least one angler in the over 10,000 records of angler lake trips from our surveys). In both cases, the data on anglers' fishing locations and species targeted were from monthly last-trip mail surveys conducted from 2008–2013 that achieved a 40%–48% response rate (Melstrom and Lupi 2013; Melstrom et al. 2015). Thus, for both inland lake and river fishing, angler behavior is linked to P because the biomasses were intentionally modeled as a function of river and lake P. For the small portions of watersheds outside of Michigan, the existing statistical relationships for the Michigan models will be applied to anglers in non-Michigan watershed areas, a form of valuation via benefit function transfer (Johnston et al. 2015).

To provide an example of how the models are coupled, we provide more detail on the inland lake recreation demand model of Klatt et al. (2017). The model was specified as a nested logit random utility site choice model (Parsons 2017). Some key specification details follow recent approaches in the literature. For example, the data included both single and multiple day trips that were pooled into a common demand specification following English et al. (2018). Only the 8,245 trips where the angler indicated that fishing was the main purpose of the trip were included in the modeling, although all 10,660 trips regardless of purpose were used to identify the 1,615 individual lakes that were included in the universal choice sets. For the round-trip travel costs, driving costs were calculated following the convention of only including marginal costs and excluding fixed costs such as annual depreciation and insurance (English et al. 2018; Lupi, Phaneuf, and von Haefen 2020), and the time costs were computed using one-third of annual household income converted to hours by dividing by 2000.

Due to the large number of individual lakes in the choice set and the fact that a large share of lakes received only one trip in the data, the model was not estimated with a full set

Table 2
Inland Lake Fishing Demand Estimates from Nested Logit Site Choice Model

Site Choice Variable	Estimated Coefficient	Clustered Standard Error
Travel cost	–0.021*	0.001
Lake size (×1,000 acres)	0.110*	0.006
Targeted species biomass (kg/ha)		
Bass	0.769*	0.149
Panfish	0.333*	0.028
Walleye	2.955*	0.199
Yellow perch	1.947*	0.358
Ecoregional nests, θn	0.646*	0.028
Trips	8,245	

* Significant at 0.001 level.

of site fixed effects as suggested by Lupi, Phaneuf, and von Haefen (in press), but it did follow their recommendation to include a set of regional fixed effects (for the HUC 4 ecoregions). The sites were nested by these ecoregions to account for possible regional correlations among the error terms. In addition, because there is a skewed distribution of lakes sizes with many smaller lakes and a large upper tail and it is likely that lake size affects visitation, the site variables included a measure of lake size along with the fish biomass variables. Table 2 reports key model estimates (the ecoregional dummies are omitted for convenience). As expected, the travel cost has a significant negative effect on site choice and there is significant support for nesting the lakes by region. Each of the game fish biomass measures is significant and positive, and walleye is the most desirable inland lake species. These biomass parameters provide the linkage between P-induced changes in biomass and changes in recreational fishing demand and value.

Beaches

Recreational use of beaches is another key ecosystem service affected by P-related algal problems. To quantify the changes in the value of this service, we adapt a spatially explicit beach recreation demand model that links algal presence in the water and algae on beaches to Great Lakes beach visitation using an economic demand system (Cheng and Lupi 2016). The model combines the revealed preference data of Chen (2013) with the stated preference data of Wiecksel (2012) to connect beach visits to beach algae (i.e., the amount of algae on shorelines and the amount of algae in the water along beaches). The model of Cheng and Lupi (2016) was specifically designed to connect beach use to the available algal data—the same data being used in the *Cladophora* ecological model from Linkage 4. The economic models use survey data with trip details (e.g., trip locations, lengths, month) for over 5,500 randomly selected Michigan residents from a statewide general population mail survey with a web-based follow-up that collected trip data. The data have beach locations for over 8,000 trips and contain a full panel of seasonal trips for each beachgoer. As with the lake fishing demand model described above, only main-purpose recreation trips were included; single and multiple day trips were pooled into a common demand structure, and travel costs were specified as above. The model was estimated as a repeated nested logit with three nesting levels: a participation level, a lake region level (with lakes divided into zones), and a beach level with the 451 individual public Great Lakes beaches.

The resulting beach demand model can predict changes in beach visitation and associated economic benefits or costs to beachgoers that result from changes in beach algae, including the location and severity of algae due to P loadings. For example, the model predicts that if half the beaches in a region experienced a 25% increase in algae on the shore and in the water, then annual recreational losses amount to almost $50 million in the relatively degraded southeastern beaches and almost $80 million in the more pristine northwestern beaches on Lake Michigan (Cheng and Lupi 2016). For areas outside of Michigan, the existing statistical relationships for the Michigan models will be applied to adult populations in non-Michigan watershed areas using benefit function transfer in a manner similar to Palm-Forster, Lupi, and Chen (2016).

Nonuse Values

Total values for changes in nutrient loads in Michigan will be modeled using a contingent valuation model being developed by Herriges, Lupi, and Stevenson (unpublished) that was purposefully designed to connect within our integrated modeling framework. A pilot version of the total value model was estimated using contingent valuation survey data collected from an online panel of over 3,000 Michigan residents during the fall of 2018 (Lupi et al. 2020). The final model will be based on results of a survey fielded in early 2020 that was an address-based, push-to-web survey of the general population of Michigan. A single referendum question was used to elicit preferences. The water quality scenarios utilize four distinct water quality metrics that were modeled for individual water bodies and shown to respondents using maps and summary statistics at the HUC 8 scale. The four water quality metrics are a water contact metric (for water safety related to fecal coliform counts), a water clarity metric, a game fish metric, and a metric of aquatic biological integrity. The game fish metric was constructed using the same underlying ecological production functions described in Linkage 4 that relate game fish biomasses to P concentrations. The other metrics were developed following a similar procedure that involved statistically modeling the relationships between each metric and the features of the water body (including P concentrations) and surrounding landscape, and then the models were used to create predictions for all water bodies. These predicted values were summarized at the HUC 8 level and reported in the survey via maps, graphics, and summary statistics. The four metrics were independently varied across the survey versions and allowed for scenarios in which decreases in P concentrations might reduce the game fish index as suggested for some species by Esselman et al. (2015). The survey also included information on recreational use that will allow us to separate use and nonuse values to avoid double-counting. The valuation model, funded by the USEPA, was designed to be able to measure values for small changes in water quality in a manner suitable for benefits transfer. The resulting valuation model will be explicitly linked to fine scale changes in P concentrations throughout the state and by construction fits directly within our IAM.

3. Discussion

This paper describes modeling work that has been underway for many years with the goal of developing an IAM for agricultural nutrient pollution. Our approach differs from some because we model at a statewide scale rather than for an individual watershed, and we use a variety of physical models that may be of interest to other practitioners (e.g., LHM and SALUS). Moreover, rather than relying on benefits transfer, all our valuation efforts were explicitly developed and designed to connect to ecological production functions linked to nutrients. Presenting our approach at this stage of development makes these models and our valuation strategies available to other practitioners and may improve the ability of some future valuation studies to connect their results to agricultural nutrient pollution.

As we have summarized, our IAM components contain a mix of completed work and work in progress. Once all components are completed and integrated, our systems assessment model can be used to evaluate the effectiveness of potential agri-environmental policies and changes in farmer behavior on the value of key freshwater ecosystem services. It provides a unique platform to evaluate conservation policies because the economic model of farmer behavior is explicitly linked to economic models of ecosystem service demand and value via the biophysical models. With our integrated model, we perform evaluations of existing and potential P-management policies.

For example, one model application to policy we plan to conduct will evaluate investments in voluntary conservation programs. Federally sponsored voluntary conservation programs have been criticized for being ineffective. Yet, few studies have directly estimated the benefits of conservation programs on the value of freshwater ecosystem services. Our IAM can assess the effects of direct (e.g., nutrient management plans) and indirect (e.g., conservation tillage and cover crops) programs on the value of key freshwater ecosystem services. This will be done by constructing a "without conservation programs" counterfactual scenario within the agricultural production model, and then feeding the changes through our IAM to evaluate changes in values. In another policy evaluation, we plan to examine the expected implications of recent lawsuits that claim farmers are responsible for nutrient pollution, and of Wisconsin's Nutrient Reduction Strategy that imposes fertilizer application limits based on mandated soil tests on all agricultural fields. For our study region, we will quantify how potential mandatory adoption of soil tests, combined with fertilizer limits or a tax on soil P in excess of a given threshold would likely affect farmer decision-making, crop yields, P transport and fate, and the values for freshwater ecosystem services of Michigan's inland lakes, streams, and coastal shores.

Among the many challenges in constructing an IAM for agricultural nutrient pollution, two of the most daunting are those that involve linking human systems with the biophysical process models describing the fate and transport of pollution in the environment. The first of these links farmer behavior with the fate and transport of pollution. Whereas the latter works at a fine spatial scale (e.g., 10 m × 10 m cells), most farmer behavior models are developed at a county or regional level, reflecting in part the available data on land use and management practices. Passing both baseline conditions (e.g., P usage, crop choices, and tillage practice), and how these conditions change in the face of policy scenarios, to the next link in the IAM requires distributing aggregate choices down to a finer spatial scale. How this is done can impact the outcomes of any policy assessment. One approach is to assume that conditions measured for the broad spatial scale apply in miniature for the individual farm or land segments. Luscz, Kendall, and Hyndman (2015), for example, assume a proportionate adjustment to recommended application rates in distributing county level fertilizer levels to disaggregate (i.e., finer) individual cells. While this is a reasonable starting point, it does not allow for farmers to respond optimally to policy changes. For example, in the face of restrictions to P use, a farmer might choose to disproportionately cut back P usage for fields with high level of P in the soil. Modeling the relative costs of adjusting P at a fine spatial scale by incorporating farmer behavior (e.g., by drawing on individual farm level usages patterns) and minimizing the aggregate costs of responding to a phosphorus constraint would provide a more accurate assessment of a policy's impact, both on the farmer and on the environment.

The second challenging linkage is between the models that characterize how ecosystem services are impacted and the modules that value the changes in those services. The former, again, operates at a fine temporal and spatial scale, describing how ecosystem services are impacted by a policy. However, most valuation exercises characterize environmental condition in broad scales or terms, for example, describing how water quality conditions have changed from "boatable" to "fishable" or "swimmable" over the course of a seasonal or annual timescale. While such simple characterizations are driven by the desire to convey the impact of a policy initiative in a concise fashion, they can hide important implications of a policy initiative. For example, reducing P loadings in a river or lake can improve the clarity of the waterway, making it more suitable for swimming, yet it may at the same time hinder the ability of the waterway to support productive fisheries. Although our work makes strides on this front, more is needed to provide consumers with a clear characterization of the ecosystem services along all the dimensions that impact the values they derive from water-based recreation.

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