Linking Agricultural Nutrient Pollution to the Value of Freshwater Ecosystem Services

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Abstract:
This paper describes our efforts to integrate economic and biophysical models to evaluate the effect that agri-environmental policies have on the value of freshwater ecosystem services. We developed an integrated assessment model that links changes in P-related management practices on farm fields to changes in the value of key freshwater ecosystem services including fish species composition, fish biomass, and beach quality (e.g., as captured by changing *Cladophora* growth rates). Our IAM will enable us to examine the effects of historical and new conservation programs on ecosystem services and valuation. Historical programs include P fertilizer use, cover crops, and conservation tillage; while new policies include the effects of soil test adoption with restrictions on P application beyond recommended levels or a tax on soil P beyond recommended levels. Results will help policy makers better allocate conservation dollars to improve water quality, enhance ecosystem services, and promote more sustainable agricultural production.

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INTRODUCTION

The agricultural sector in the U.S. 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; IJC 2014). These excess nutrients, in turn, lead to water quality degradation, including nitrate and pesticide contamination of groundwater, algal blooms, hypoxic or 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 their environmental costs. Policy instruments should be designed to encourage producers to consider such tradeoffs (Garnache et al. 2016a).

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 source pollution 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 (BMPs) 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 et al. 2012).

Many studies have examined the effect of conservation programs on land use and farmer adoption of BMPs (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 et al. 2000; Phaneuf et al. 2000). To the best of our knowledge, there have been 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 policy that induces the loading scenario to arise from farmer behavior. Both of these papers represent valuable contributions to the literature. 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 a major cause of stream and lake impairment in the U.S. and conservation programs incentivize farmers to change behavior, little is known about the relationship between such behavior change on the farms and the change in the value of endpoint ecosystem services (Smith and Weinberg 2006). Research connecting policies to outcomes of interest is essential to comprehensively evaluate the effectiveness of agri-environmental policies, allowing for direct comparison of a conservation program's costs and the corresponding benefits accrued to the consumers of ecosystem services. Yet, the task is not trivial (Garnache et al. 2016a; Smith and Weinberg 2006).

Consider policies designed to mitigate the environmental impact of excess phosphorus (P) loadings, a limiting nutrient and common pollutant in freshwater pollutant (USEPA 2009).

Program evaluation requires understanding not only the farm level response to a given policy, but also the ability to trace resulting P loadings from the farm to surface water and groundwater, in the form of dissolved reactive P (DRP) or attached to particulates. Phosphorus then moves into streams and lakes, where it affects algal growth and fish species composition and biomass. The growth of filamentous algae can lead to muck formation, which covers beaches and releases a foul smell when it decomposes, while the growth of toxic algae such as *Microcystis* can make water unfit for drinking and/or bathing and cause beach closures, ultimately diminishing valued ecosystem services. Linking the various pieces of what Garnache et al. (2016a) refer to as the "P puzzle" (Figure 1, below) is essential to evaluate the effectiveness of agri-environmental policies. Linking these pieces is an area ripe for development of an integrated assessment model (IAM) with the incumbent challenges associated with developing IAMs (Hamilton et al 2015; Kling et al 2017; Keiser and Muller 2017).

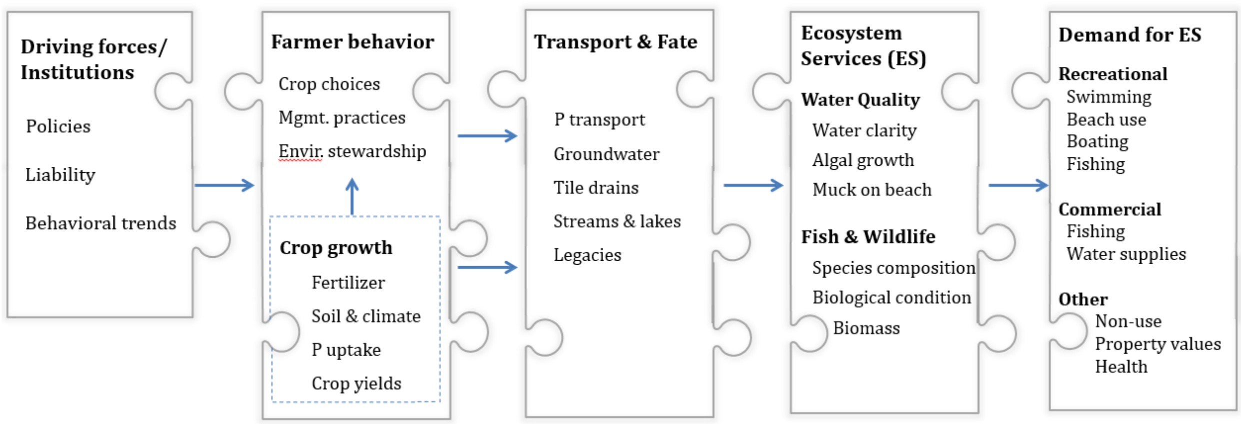


Figure 1. *Components of the P puzzle (adapted from Garnache et al. 2016a).*

The Great Lakes are the largest freshwater ecosystem in the world, and are a major area for recreation activities. Inland lakes and streams of Michigan's Lower Peninsula are home to a large recreational fishing community with game fish species sensitive to P concentrations. Extensive areas of the Great Lakes have *Cladophora* as a major problem with beach fouling. Studies implicate nonpoint nutrient pollution as a driver of these ecosystem service impairments, yet existing piecemeal approaches are insufficient for understanding the economic costs and benefits of changing nutrient loads.

The goal of this paper is to address this gap by developing an integrated assessment model of 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 river basins that drain to the Great Lakes from Michigan's Lower Peninsula, which feed into Lakes Michigan, Huron, and Erie. Our integrated assessment model is designed to capture farmer responses to alternative policy scenarios, that in turn dictate excess P leaving the farm, and consumer demand models that link algal growth to demand for beach visits and changes in fish biomass to demand for recreational fishing. Linking these two end pieces of the P-puzzle (Figure 1) 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 integrated models can evaluate the effect of existing conservation programs that incentivize changes in P use, conservation tillage, and cover crop adoption, on changes in the value of both beach and fishing recreation. They can also quantify the likely effectiveness of management options such as mandatory soil tests combined with restrictions on P application or a tax on excess P in the soil, and changes in farmer attitude (e.g., related to changes in social norms) affect the value of beach and fishing recreation. Evaluating the benefits of agri-environmental policies will help policymakers and regulatory agencies identify and design cost-effective programs to address P pollution.

Our approach focuses on the management of agricultural resources and the effect of agri-environmental policies on the value of environmental goods and services. We discuss the development of interdisciplinary methods, tools, and applications in the areas of: (1) water resources management economics; (2) economic incentive mechanisms and policies designed to promote resource conservation; (3) economic impacts or implications of agriculture, resource conservation and management; and (4) methodological advances in non-market valuation.

Our research will form a basis to improve the long-term sustainability of US agriculture and environmental quality that is essential to human well-being. On the one hand, fertilizer use ensures high crop yield and contributes to national and global food security. On the other hand, the excessive application of nutrients on agricultural soils causes water quality degradation and loss of ecosystem services valued by society. Designing incentives that improve the tradeoff between crop yields and water quality degradation will enable more sustainable agricultural practices, while still meeting the global demand for food crops. Our findings will help regulatory agencies evaluate the effectiveness of existing conservation programs and design the cost-effective policies of the future.

Methods

We outline approaches to: 1) understand relationships between changes in P-related management practices (e.g., fertilizer use, cover crops, and conservation tillage) and key endpoint ecosystem services people value such as recreational fishing and beach use, and 2) evaluate how existing conservation programs, changes in policies, the legal framework, and/or farmer behavior impact the value of freshwater ecosystem services. 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, and linking these two endpoints with biophysical models of P transport and fate that impact the ecosystem services available to consumers.

The methods we present are designed to quantify relationships between changes in P-related management practices and key endpoint ecosystem services that people value. The introduction of P-related management practices puts into motion a complex series of effects connecting the response of farmers to a policy initiative and the ultimate change in ecosystem services that consumers value. Many relationships among natural and social systems that regulate P fertilization and transport, as well as algae and fish response to P, are not sufficiently understood to manage freshwater eutrophication. Understanding each piece of the P puzzle, illustrated in Figure 1, is key to developing a complete picture of the efficacy of alternative P-management programs. Our first objective is to identify and characterize each component in this puzzle.

A key piece in the P puzzle is to understand farmer behavior in response to a given policy initiative. Typically, farmers are assumed to maximize profit or the utility of profit to take into account risk aversion. In doing so, they choose which crops to plant, rotations to employ, and nutrients to apply, as well as whether to participate in available conservation programs. These decisions not only affect crop growth, but also feed into another piece of this puzzle affecting the transport and fate of P beyond the farm.

To explain P transport, we first need to understand crop growth and yield, along with farmer behaviors that maintain yield. The relationship between P fertilization and crop yield is asymptotic, such that large increases in yield occur when P levels in the soil are low, while yield response to P becomes low when P levels are sufficient. In contrast, algal biomass often responds exponentially to P loading beyond some assimilative capacity (Downing et al. 2001).

Given farmer behavior, and its implications for P use, the next piece of the puzzle is characterizing the transport and fate of any excess P beyond the edge of field. New understanding of the forms of P (total P, or TP, consists of attached particulate P and dissolved reactive P, DRP) and their transport is shedding new light on the efficacy of historical management practices prescribed to control P runoff. For example, conservation tillage adoption has reduced attached P runoff from the edges of fields but has contributed to greater DRP movements via subsurface flows. Given that DRP is the main driver of algal blooms, the net effect of conservation tillage on freshwater ecosystems is unclear. Furthermore, the increased prevalence of agricultural tile drains that empty into streams exacerbated the transport of DRP (Kleinman et al. 2015). Yet, how this new knowledge about the effect of tillage conservation and tile drains on the transport of DRP relates to changes in the value of recreational services remains unanswered.

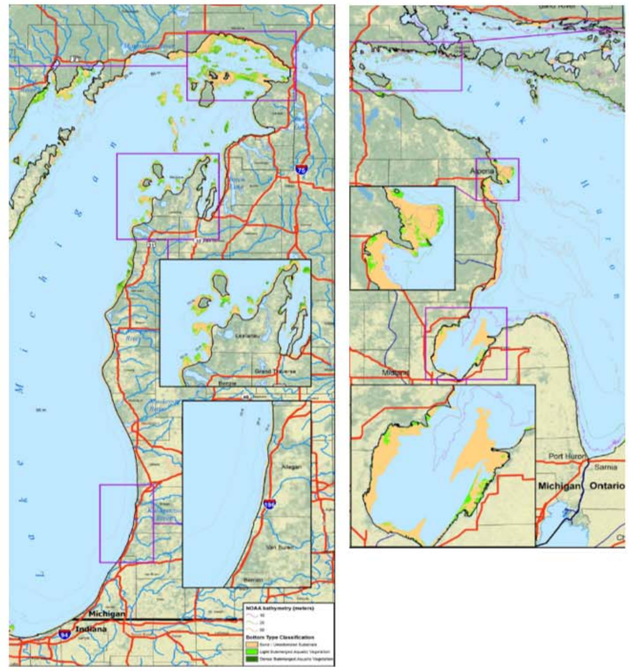
The importance of land use legacies, which is the result of past human activities that can take decades to move through the environment, is also important to consider (Reed-Anderson et al. 2013; Hamilton 2012). For example, Jarvie et al. (2013) estimate that 70 to 80% of the P applied to agricultural fields remains in the environment, including soils, groundwater, inland freshwater bodies, and estuaries. Legacy P has important implications for P management and policymaking due to the presence of lag times that occur between when actions are taken and the start of P concentration decreases in lakes and streams. In addition, these lag times depend on ecosystem characteristics such as soil and hydrologic properties (Jarvie et al. 2012, 2013; Sharpley et al. 2013). Legacy P may further influence our understanding of changes in ecosystem service values and the design of cost-effective policies. Research quantifying the legacy effects of land use on water quality (e.g., Ray et al. 2012; Martin et al. 2011, 2015, 2016; Wiley et al. 2008, 2010; Pijanowski et al. 2007; Wayland et al. 2003) can provide more accurate estimates of how historical landscape management techniques affect nutrient concentrations in surface water ecosystems. Such research provides a basis for understanding the consequences of past human activities, which can in turn be used in an integrated assessment model to test the likely effectiveness of potential future policies under projected changes in climate.

Understanding P transport and its legacies enables us to link farm behavior through to its implications in terms of ecosystem services, the next piece in the P puzzle, including its impacts on key ecosystem services such as beach quality and fish biomass and species composition. Algae and aquatic plants foul beaches where they reduce aesthetics and safety for beach use (Shear and Konasewich 1975; Bhappanahalli et al. 2003; Verhougstraete et al. 2010). Different species of algae foul beaches and contaminate drinking water, and they typically grow in different places in lakes. In the Great Lakes, cyanobacteria growing in the water column cause most of the problems with drinking water contamination, whereas the most common problem for beach fouling is the filamentous green algae *Cladophora* that grows on the bottom of lakes. Beach fouling commonly occurs after storms with waves that disturb the lake bottom and wash scoured algae and aquatic plants ashore. Aesthetics are clearly impaired when beaches are fouled as algae accumulate in piles that rot on the beach, causing foul smells. These algae accumulations can also harbor high levels of bacterial contaminants from wastewater, which can threaten human health (Whitman et al. 2003; Ishii et al. 2006; Olapede et al. 2006). There have been numerous stories in the popular press documenting the consequences of harmful algal blooms (HABs), including interrupting drinking water supplies in Toledo, Ohio, to recent beach closures in Florida due to toxic algal blooms.

Beach fouling in the Great Lakes by *Cladophora* was a major problem for recreation in the middle to late 1900s (Taft and Kishler 1973; GLWQB 1975). Phosphorus pollution was identified as the major cause of great *Cladophora* growth on the bottoms of the Great Lakes and the resulting beach fouling. As a result, laws were passed that regulated the amount of P used in detergents. Phosphorus regulation reduced P pollution of the Great Lakes and reduced beach fouling by algae (Painter and Kamaitis 1987). Then in the late 1900s, invasive species changed the Great Lakes ecosystem and caused a resurgence of beach fouling by benthic algae (Higgins et al. 2005a; Auer et al. 2010). Dreissenid mussels, which are actually multiple invasive species, are filter feeders that live on the bottom of the Great Lakes. They filter algae and other suspended microscopic organisms from large volumes of water and have changed the Great Lakes in two key ways that favor bottom dwelling algae that foul beaches (see review in Auer et al. 2010). First, they clarify the water such that light can reach deeper into the water and provide enough light for algae to grow over larger areas of the lake bottoms. Second, they translocate the nutrients in algae and microbes from the water column to the lake bottom, where the algae and microbes decompose and enrich bottom water with nutrients. Thus Dreissenid mussels increase light and nutrient availability to the lake bottom, which facilitates the growth of *Cladophora*. 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 (Figure 2; Kuczynski et al. 2016). Multiple models have been developed to relate reductions in P concentrations to *Cladophora* growth, resulting mass on lake bottoms, and beach fouling (Auer and Canale 1982; Auer et al. 1982; Canale and Auer 1982; Higgins et al. 2005b). Little is known about P relationships with growth of other nuisance, bottom-dwelling algae and macrophytes in the Great Lakes, but these problems are primarily associated with low P conditions and are less frequent, extensive, and severe.

Climate change may further influence P management and the effects of P on the value of ecosystem services. One of the most concerning issues associated with climate change is the projected increase in the variance of precipitation (Melillo et al. 2014). This increase is expected to cause stronger precipitation events, which in turn will likely increase P transport due to overland flow and flow through agricultural drains. Agri-environmental policy analysis should thus account for changes in climate to generate adequate effects. For *Cladophora*, we expect little direct effect of temperature because predicted Great Lakes temperature changes with climate change are not sufficient to change *Cladophora* growth rates (Malkin et al. 2008). Game fish productivity may increase a small amount due to increased algal productivity with rising temperatures, but such temperature effects will be outweighed by changing thermal regimes that

Figure 2: *Co-author Stevenson on a Lake Michigan beach with submerged aquatic vegetation (SAV) on beach and in water (**left**). Maps of remotely sensed SAV along coastlines of the lower peninsula of Michigan (**right**, Michigan Tech Research Institute, www.mtri.org/cladophora.html).*



strongly affect distributions of cold and warm water fish. Eaton and Scheller (2006) predict that climate change will result in a 50% loss in habitat for cold- and cool-water fish.

The last piece in the P-puzzle is to value changes in ecosystem services implied by the sequence of effects characterized in the other puzzle pieces (Figure 1). In this paper, we focus on those ecosystem services with values that derive from recreation activities, particularly beach recreation and fishing. Recreation demand has 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 (e.g., Bockstael et al. 1987; Egan et al. 2009; Hicks and Strand 2000). An individual reveals the use value they receive from a site or its amenities through how much time and money they are willing to expend to reach that site or amenity. 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 an underlying policy 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).

Although many studies examined pieces of the P puzzle in isolation, few have attempted to connect all these pieces to evaluate the effect of agri-environmental policies on the demand for and value of ecosystem services. Here we discuss methods to link economic models of farmer and consumer behavior through the primary biophysical processes driving P transport and fate in a comprehensive manner, including relationships between: plant uptake and runoff, the transport and fate of P through groundwater and surface water, and between P concentrations, algal growth, and fish biomass, with farmer behavior and economic models of consumer demand for ecosystem services. The resulting integrated model will enable us to estimate the effects of changes in agricultural practices on the key ecosystem services that people value.

Our research also evaluates how existing policies, as well as changes in policy, legal framework, and farmer behavior, affect the value of freshwater ecosystem services. Many conservation programs provide incentives to adopt BMPs that directly (e.g., nutrient management plans) or indirectly affect P runoff from the edge of field (e.g., filter strips, cover crops, conservation tillage). A rich literature examines the effects of conservation programs on the adoption of BMPs and land use (e.g., Cooper 2003; Liu and Lynch 2011; González-Ramírez and Arbuckle 2015). Conservation programs have come under scrutiny for providing payments to farmers who would have adopted a BMP absent financial incentives. This question of "additionality" of conservation programs has been the subject of many studies (e.g., Mezzatesta et al. 2013). Yet, few studies have gone further and attempted to link these conservation programs to the resulting changes in ecosystem services that consumers value (Garnache et al. 2016a). Sohngen et al. (2016) evaluated the effects of federally-sponsored voluntary conservation programs on P concentrations in streams and lakes (by seeking to statistically relate puzzle piece *one* directly to piece *four*, Figure 1). Yet, their study does not link the effects of agri-environmental policies all the way to the ecosystem services that consumers value, and it does not take advantage of process-based models to capture essential features of P transport and fate including legacies. Although some studies (Feather and Hellerstein 1997; Crutchfield et al. 1995; Ribaudo 1989) have used recreation demand models to value water quality that was statistically related to agricultural soil erosion (thus connecting pieces *two* and *five* in Figure 1). They do so without key pieces of this puzzle, such as the detailed biophysical models that we discuss in this paper. Here, we describe methods to integrate economic, crop, hydrology, fish, and algae models to quantify interactions amongst these coupled systems.

The Clean Water Act exempts agricultural nonpoint source nutrient pollution from regulation and relegates its supervision to states. In general, states have opted to reduce agricultural nonpoint source pollution through voluntary programs. Yet, a number of prominent lawsuits may help change the current legal framework and farmer liability. In Iowa, for example, the Des Moines Water Works sued three county drainage districts for agricultural nitrate pollution contaminating the city of Des Moines' drinking water supply. Nitrate runoff from farms currently costs the water utility ~$1 million a year for additional treatment of the water, with future costs expected to increase substantially. This lawsuit put pressure on the state to reconsider whether farmers can be found liable for nutrient pollution (Jacobs and Weninger 2015). In Ohio, farmers are liable for failing to implement BMPs that would prevent nutrient pollution. Yet, the process is cumbersome with many steps and few mandated practices, thus reducing its effectiveness (Kilbert et al. 2012).

Unsurprisingly, lawsuits have been more common for concentrated nutrient sources where an individual is involved such as CAFOs, specifically when they fail to comply with discharge permits or suffer failures that lead to releases, rather than when many farmers contribute to diffuse pollution that cannot be traced back to specific individuals (Ogishi et al. 2003; Laitos and Ruckriegle 2013). For example, in 2001 and 2004 the City of Tulsa and State of Oklahoma lawsuits against upstream poultry farms in Oklahoma and Arkansas claimed that the farms were contaminating reservoirs along the Illinois River, which is Tulsa's drinking water source (Kleinman et al. 2015). The case resulted in management standards that restrict application of P to soils that test above a threshold. Due in part to changes spurred by the lawsuit, P concentrations have decreased by one-third (Kleinman et al. 2015). In another case, a citizen-suit claimed that a Delaware poultry farm was responsible for polluting Chesapeake Bay, though the poultry farm was not found liable (Egan and Duke 2015).

In Wisconsin, the State Legislature adopted a nutrient management rule (Wisconsin Administrative Code 2013; NR 151.07) that regulates the application of manure, commercial fertilizer, and other nutrients to meet federal Gulf of Mexico hypoxia nutrient reduction goals and improve water quality in Wisconsin lakes, streams, and groundwater. The rule requires all crop and livestock producers who apply manure or other nutrients directly or through contract to agricultural fields to comply with a nutrient management plan. All farm nutrient management plans include complete soil tests conducted by certified laboratories. All crop fields must be tested or have been tested within the last four years, and results give base fertilizer recommendations for each field. In addition, to participate in some federal and state farm programs involving cost-sharing, farmers may have to comply with code 590 nutrient management standards from the USDA-Natural Resources Conservation Service (NRCS); these define the minimum requirements and components of acceptable nutrient management plans.

Such changes in farmer liability and in nutrient management rules may contribute to the establishment of mandatory soil tests. Limits on fertilizer application may be enforced when soil tests exceed some P threshold, or a tax may be imposed on excess P. Here, we describe approaches to examine how widespread adoption of soil tests combined with limits on P applications, or a tax on P applied beyond the recommended level, would affect the value of freshwater ecosystem services.

In addition, changes in social norms, farmer attitudes toward environmental stewardship and new P retention or removal technologies, may affect P use, adoption of BMPs, and/or P concentrations in streams (Garnache et al. 2016a). Social norms are constantly evolving and reflect current societal changes. With growing awareness about environmental degradation, including air and water quality, social norms may pressure farmers to adopt more sustainable agricultural practices contributing to the well-being of future generations. Farmer attitudes towards P use and conservation programs may change for a variety of reasons. Technologies for P removal from water are emerging, in particular for manure slurries, and could directly reduce P stocks in the environment (e.g., Safferman et al. 2007). We discuss methods to explore how such changes will likely affect the value of endpoint ecosystem services.

Constructing an integrated model of the P puzzle depicted in Figure 1 requires an interdisciplinary team with expertise in both the economics driving farmer behavior and consumer demand for ecosystem services, as well as in the biophysical process models that connect these two end pieces of the puzzle. The integrated economic and biophysical model will also enable the linkage to be made to non-use values.

APPROACH

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 encompasses the Lower Peninsula of Michigan and its coastal shores with Lakes Michigan, Huron, and Erie depicted on Figure 3, though our IMA can be applied to solve similar issues to other sites where model input is available. Our research has two primary objectives:

i) Link changes in P-related management practices to changes in the value of freshwater ecosystem services,

ii) Model the effects of current policies, and changes in policies, in the legal framework, and/or in farmer behavior on changes in the value of freshwater ecosystem services.

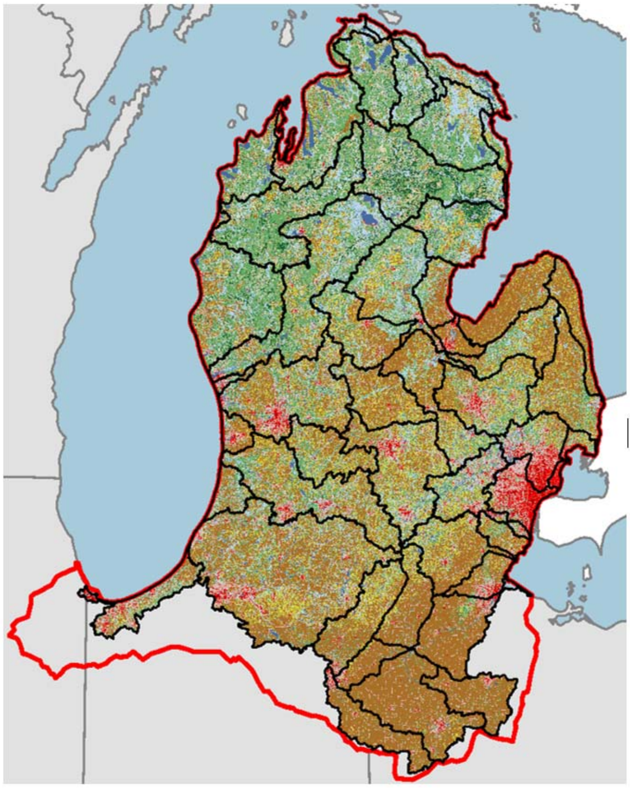


Figure 3: *Research focus area: River basins (in black) that drain to Great Lakes from Michigan's Lower Peninsula and feed into Lakes Michigan, Huron, and Erie. The red outline is existing LHM model boundary. Basemap is land use from the National Land Cover Dataset (NLCD) with agriculture in brown to yellow colors, urban in red, forest in green and water in blue.*

Objective 1. Linking changes in P-related management practices to changes in the value of freshwater ecosystem services.

Change in farmer P management can be related to P loading into streams and P loads to nearshore zones of the Lower Peninsula of Michigan, encompassing Lakes Michigan, Erie and Huron. We leverage existing models developed by our team and others to link P from farms to algal growth and fish biomass and to the value of these services. The models will be 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. Our model will capture climate and biophysical uncertainties and will contribute to identifying potential thresholds or tipping points. Nearshore zone mixing models will be used to relate P loading to P concentrations, which will then be related to *Cladophora* growth, accumulation, and beach fouling while accounting for area of nearshore zones with enough light and suitable habitat for *Cladophora* growth. To connect people to ecological implications of P fertilization, we focus on two affected ecosystem services: recreational fishing and swimming/beach use, which both have over 1 million participants annually in Michigan alone (USFWS 2013; Chen 2013).

We are developing an integrated assessment model that links P from the farms to the value of ecosystem services and to the evaluation of agri-environmental policies . The key steps in this process are broken down into seven steps, five of which are depicted in Figure 4 and address pieces of the P puzzle (Figure 1), while the last two focus on model integration and the sensitivity of our results to climate change scenarios.

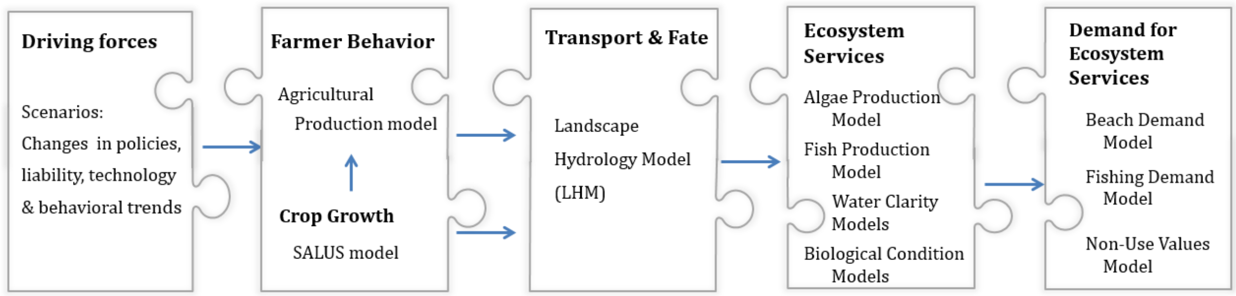


Figure 4: *Linkages between the models to trace P from farm to the value of ecosystem services*

Step1: Link farmer behavior to changes in P-related management practices.

To understand how farmer behavior affects freshwater ecosystems, we will model farmer decisions that influence P dynamics in soils and the environment. Our model will capture farmer crop choices, P fertilization, and soil management practices, including tillage choices.

We will build a positive mathematical programming (PMP) model of crop production for the river basins that drain to the Great Lakes from Michigan's Lower Peninsula. PMP is a nonlinear programming method that is widely used for agricultural and environmental policy analysis (Johansson et al. 2007; Jenkins et al. 2001; Jansson and Heckelei 2011). It allows exact calibration of agricultural production models against available economic information with minimal data requirements, e.g., 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: a) exactly replicate the reference allocation without using artificial constraints that may impede the model's ability to respond to policy shocks, and b) yield smooth responses to policy changes, rather than step-wise responses from one corner solution to the next. PMP models can accommodate a variety of constraints relevant to farmer decision-making such as regional resource limitations (e.g., land, water). Such models can readily accommodate decreasing marginal returns to specific inputs, e.g., to account for heterogeneity in land quality, and/or decreasing returns-to-scale. 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 et al. 2016b, c). Such yield response information may come from field data or biophysical models, allowing PMP models to be coupled closely with process-based models such as SALUS (described below) as the yield information from the crop growth model feeds 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 fate and transport of P).

Land use and nutrient source data have been compiled for all the watersheds in Michigan's Lower Peninsula that drain to the Great Lakes for 2003-2013 (Luscz et al. 2015, 2017). Cost and other data for current practices for major crops are available from USDA National Agricultural Statistical Service and MSU Extension's costs of production and crop budgets. In addition, cost estimates for adoption of conservation practices in Michigan (Ma et al. 2012; Yeboah et al. 2015) will be combined with data from USDA programs such as Conservation Reserve

Enhancement Program (CREP). Agronomic yield responses to P and nitrogen can be derived using the SALUS model (discussed in Step 2 below).

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 (Figure 5; Luscz et al. 2015, 2017), we augment the crop production model with a livestock economic model to capture major manure production sources and waste management practices. The livestock loading model we developed (Luscz et al. 2015) predicts the location of manure sources and the spatial variability in P loading to the ground surface.

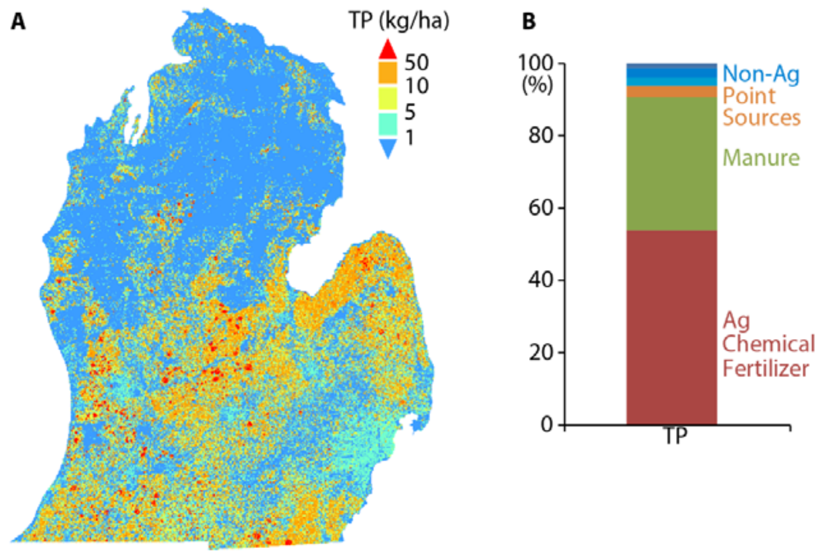


Figure 5. (a) *Total phosphorus (TP) loading to Michigan's Lower Peninsula (LP) from all point and non-point sources (kg/ha/year). "Non-Ag" includes septic systems, atmospheric deposition, and non-ag fertilizer,* **b)** *proportion of P contributed by different sources for the entire LP (Adapted from Luscz et al. 2015).*

Step 2: Link changes in P-related management practices to P legacy and changes in P transport from fields.

In our IAM, yield responses to P fertilizer and soil P availability is modeled with the validated Systems Approach for Land Use Sustainability (SALUS) model. SALUS has been well 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 et al. 2013), plant N uptake and phenology (e.g. Basso et al. 2010, 2011a, 2011b, 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 and C, N, 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, agronomic management practices. In case of missing information, SALUS has internal intelligence of proxies to provide estimates of missing input based on domain and geographic knowledge.

SALUS is capable of capturing new genotypes (GMO, and other morpho-physiological traits) including the impacts of heterogeneous plant stand (prolificacy and barrenness) on grain yield. This is a unique feature of this model, compared to all the other models. SALUS can be executed spatially on areas within field that have been preclassified on low yield, high yield, or unstable yield using a novel procedure developed by Basso et al. (2019). This procedure is able to identify areas within the field that are not responsive to N or P fertilizer applications. Since SALUS quantifies the dynamic changes of soil nutrient availability throughout the season, it can use that information in combination with historical simulations of outcome probabilities in relation to local climate to evaluate strategic management strategies that tactically apply N and P fertilizers to better tailor soil nutrient supply with plant demand to increase nutrient use efficiency. Recently, Basso et al (2019) showed that constantly low yields in a field could occupy up to 30% of the field areas. Addressing spatially variable N and P application rates using this approach and SALUS recommended rates, nutrient loss could be drastically reduced and water quality improved, along with greater profit for the farmers.

Step 3: Link P legacy and 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 et al. 2007; Kendall 2009; Wiley et al. 2010; Luscz et al. 2017). It couples vegetation, 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. 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, windspeed), soil hydraulic properties, aquifer properties (hydraulic conductivity and specific storage), Leaf Area Index from the MODIS satellite every 8 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 et al. 2017) and the High Plains Aquifer, has been shown to accurately predict streamflows within ~6% of annual precipitation with no calibration of parameters except 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 source and transport model consists of a high-resolution (30 m) source loading model and a paired statistical and process transport model (Luscz et al. 2015, 2017). Sources are described using land use, census, agricultural census, aerial imagery, and remote sensing data. Surface or subsurface-applied nutrients are then 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). Growth, decay, and loss of algae respond to nutrient concentrations and water temperatures along with the flow and transport processes described by LHM. Flows of water and nutrients feed from this model into the algal growth model in Step 4.

The LHM code offers significant advantages compared to other commonly used codes such as SWAT (Arnold et al, 2012), because it is process-based and fully-distributed with measurable properties, and thus can better represent conditions outside of the range that have been calibrated. For example, once it has been validated, it can be used to explore likely changes in fluxes for scenarios of changing climate and land use.

Step 4: Link changes in P (and N) concentrations in streams and lakes to changes ecosystem services

In this step we will relate P (and N) concentrations to measures of ecosystem services in inland lakes, streams and rivers, and the coastal zone of the Great Lakes (Appendix Table 1). The ecosystem services include potential for recreation based on safe water contact and aesthetics, fisheries production, and biological condition as measures of ecosystem services. We selected these ecosystems services because they are known to generate economic value and and we expect different responses of these services to P (and N) pollution. For example, along nutrient pollution gradients we expect differing magnitudes of change, peak sensitivities to pollution in different ranges of pollution, and discordant responses to pollution. As P pollution is reduced, most measures of ecosystems services will improve, except fisheries production will decrease for many fish species.

We plan to use existing and new models to characterize responses of ecosystem service parameters to decreases in P concentrations resulting from P management. Where possible, we will use existing models when they have been characterized fully in the literature or are available from authors. When existing models are not available, we will develop new statistical models using data from a variety of sources, including extensive data from USEPA national and regional surveys as well as other personal and publicly available sources of data. In a few cases, e.g. *Cladophora* in the Great Lakes, we will use 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 will be included in models to account for natural variability in ecological responses to phosphorus. When we have results of more than one model for parameters, we will compare results and utilize multi-modeling techniques to evaluate consistency in results and integration of those results (Wiley et al. 2010).

In general, most models of lake phytoplankton will be taken from the literature. A number of models for planktonic chlorophyll a and cyanobacteria have been developed, commonly at larger scales than the lower peninsula of Michigan (Downing et al. 2001, Hakansson et al. 2007, Beaulieu et al. 2013, Yuan and Pollard 2014). These models or revisions of these models will be tested with data from Michigan from personal and state datasets, and then applied to all lakes in lower peninsula. Models for water clarity and turbidity can also be constructed and tested using data from the USEPA's National Lakes dataset using available parameters, and 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 algal and cyanobacterial abundance, water clarity, and turbidity.

Models of biological condition response to P concentration will be developed using data from the USEPA National Rivers and Streams Assessment, the USEPA National Lakes Assessment, and the data from the USEPA-sponsored Great Lakes Ecological Indicators Project. We will develop multimetric indicators (MMIs) of ecological conditions using site specific modeling approaches in which natural variation in expected values of metrics in reference condition are predicted and compared to measured conditions at sites (Cao et al 2007, Tang et al. 2016, Liu et al. 2017). Natural variation in metric values can be great among reference sites within an ecoregion. Site specific models have performed better than traditional MMIs that do not account for natural variation among sites within ecoregions. Effects of phosphorus condition will be related to MMIs and predicted for lakes, streams, rivers, and coastal zones of the Great Lakes. We expect substantial differences in the responses of algae, invertebrates and fish to P concentration, and for those effects to differ among habitat types.

Cladophora beach fouling provides a good example of how multiple models will be used to characterize response of ecosystem services to P concentration management. Beach fouling by algae will be related to P loading to nearshores zones of the Great Lakes surrounding the Lower

Peninsula of Michigan using three models. The first model can use an ensemble of results from hydrodynamic nearshore zone mixing models to predict P concentrations based on P load to the lakes, which will be estimated using LHM (Step 3). The nearshore mixing models are available from NOAA (www.glerl.noaa.gov/res/glcfs). P concentrations can 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 can predict Cladophora accumulation (biomass/area) in the nearshore zone as a result of its growth rates (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 (1982a, 1982b) model and the simplified version called the Cladophora growth model (CGM, Higgins et al. 2005b, 2006). These models have been applied successfully in multiple regions of the Great Lakes (Malkin et al. 2008), particularly in Lakes Huron and Erie. The third model will relate Cladophora accumulation to beach fouling using modeled nearshore accumulation, satellite maps of macroalgal habitat (Shuchman et al. 2013; Brooks et al. 2013), and data for beach fouling from the Michigan BeachGuard System (Michigan Department of Environmental Quality (MDEQ), www.deq.state.mi.us/beach/).

For fish, models have been developed that relate land use, climate, and geology of watersheds to P concentrations, game fish biomass, and fish biological integrity in an EPA-funded project (Esselman et al. 2015). Game fish abundance in lakes and rivers have been measured by the Michigan Department of Natural Resources. Models of nutrient concentrations in lakes and rivers have been developed based on watershed land use, geology, soils, and climate using boosted regression trees. For inland streams and rivers, estimates for each stream segment (at the HUC10 level and finer) of species-specific fish biomass and an index of biological integrity from models developed by Esselman et al. (2015). For inland lakes, similar ecological models have been developed (Esselman et al. unpublished data) linking species-specific biomass to nutrient loads for the 1,157 Michigan lakes 100 acres or larger and for 458 lakes 10-100 acres visited by at least one angler in our recreation data (described in Step 5).

Step5 Link changes in fish species composition and fish biomass to changes in the value of recreational fishing, and changes in algae at beaches to changes in the value of beach recreation.

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 2015). 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 biomasses for brook trout, brown trout, walleye, bass and panfishes. For inland lakes, the Klatt (2015) model links angler site choice decisions to species-specific biomasses 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 all the 458 lakes 10-100 acres visited by at least one angler in the over 20,000 records of angler lake trips from the monthly fishing surveys conducted by MSU from 2008-2013). In both cases, the data on anglers' fishing locations and species targeted were from monthly surveys that achieved a 46% response rate (Melstrom and Lupi 2013, Melstrom et al. 2015), with a data set that now includes details on over 30,000 trips. Thus, for both inland lake and river fishing, angler behavior is linked to P because the biomasses were modelled as a function of river and lake total P as well as site characteristics (e.g., temperature, size, morphology, nutrients) and landscape-scale characteristics (e.g., land cover, quaternary geology, hydrologic connectivity measures). For the small portions of watersheds outside of Michigan, the existing statistical relationships for the Michigan models can be applied to anglers in non-Michigan watershed areas, a form of valuation referred to as benefit function transfer (Johnston et al. 2015).

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 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 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 to in the Cladophora growth ecological model from Step 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 web-based survey with a 59% response rate. The data has beach locations for over 8,000 trips. The resulting model can predict changes in beach visitation and associated economic benefits or costs to beach-goers that result from changes in beach quality, 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 (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 et al. (2016).

Non-use values for changes in nutrient loads in Michigan are modelled using a contingent valuation model developed by Herriges, Lupi and Stevenson (unpublished) that was purposefully designed to connect within our integrated modelling framework. The non-use valuation model measures willingness to pay for water quality indices and takes care to separate use and non-use values. The non-use model was estimated using contingent valuation survey data collected from over 3000 Michigan residents during Fall 2018. The non-use valuation model, funded by EPA, was designed to be able to measure values for small changes in water quality in a manner suitable for benefits transfer. The model can value policies with state-wide (the whole of the Lower Peninsula) or regional scales (where regions are individual HUC 4 watersheds), at either scale all water-quality metrics are quantified and shown to respondents at the levels of HUC8 watersheds. The spatially-explicit water quality metrics are extensions of the traditional water quality ladder index, with the accompanying swimmable, fishable, and boatable categories. The extension separates the index into a “water contact” index (e.g. swimming and boating), a “recreational fishing” index (i.e., a preference-weighted index across game fish species whose abundances are quantified as discussed in the ecological models above), and an aquatic “wildlife index” based on the biological condition gradient (USEPA, 2016).

Step 6 Integration of the models.

All the models can be integrated to align the spatial and temporal scales of the key input and output variables. Relationships and key variables are described in Table 1.

Table 1. Processes, relationships, and key variables of the models (See also Figure 6).

Processes (Figs. 1, 4, #)	**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	Agric. Production Model	P application = f(yield, prices, norms, regulations)
Transport and Fate	Landscape Hydrology Model (LHM)	P conc. = f (land use, hydrology, sources) Water temp = f (recharge, air temp, LU)
Ecosystem Services	Algae Production Model	Algae in water and on beach = f(P concentration, nearshore habitat)
	Fish Production Model	Fish biomass = f(water temp, P concentration)
	Biological Condition Model	BC = f(P concentration, natural factors)
	Water Quality Model	WQ = f(P concentration, e-coli, clarity)
Demand for Ecosystem Services	Beach Demand Model	Value = f(algae in water and on beach)
	Fishing Demand Models	Value = f(fish biomass by species)
	Non-use Values	Value = f(water quality, fish, biological cond.)

The crop growth model (SALUS) quantifies the effects of the interactions between soil, weather, and agricultural management practices, including P fertilizer to simulate and predict crop yields. These crop yields feedback into the farmer behavior model as farmers make crop choices and agricultural management practice decisions based on crop yields, crop prices, and inputs costs. The crop growth then determines P uptake by plants and P in the soil from the farmer's P fertilizer. LHM simulates hourly flows of water in a fully spatially distributed manner across the study domain, based on dynamic land cover derived from both the SALUS model and Leaf Area Index product every 8-days from the MODIS satellite, along with hourly climate data from NLDAS. P loads discussed earlier are then converted to P fluxes along stream reaches and at the outlets of all rivers to the Great Lakes, using statistical methods described in Luscz et al. (2015, 2017). The available pools of P decline as plants take up nutrients, based on SALUS simulations across space and time, and increase when farmers apply fertilizer. The

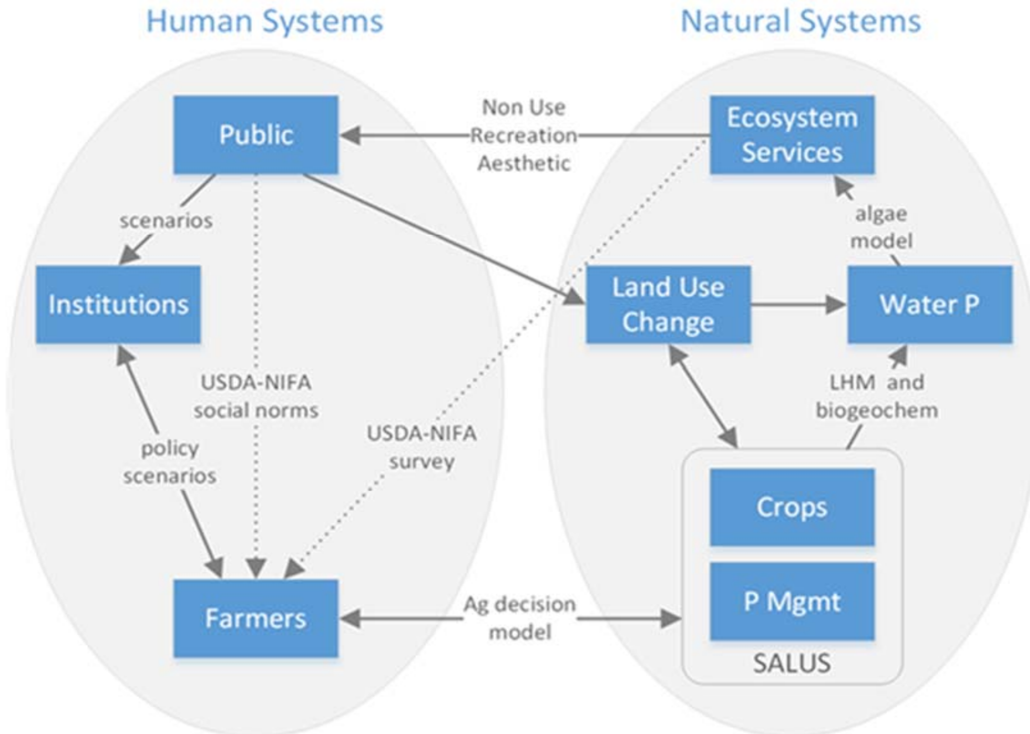


Figure 6. *Conceptual diagram of linked the models.*

concentrations of nutrients at the river outlets to the Great Lakes are then diluted based on results from an ensemble of hydrodynamic mixing models run by NOAA, with outputs available on their web site. Nutrient concentrations in nearshore zones where the macroalga *Cladophora* grows can be related to *Cladophora* accumulation and beach fouling. Nutrient concentrations in rivers and streams can be related to species-specific models of game fish biomass. Changes in beach fouling and game fish biomass can then be valued in the beach and fishing demand models, since they are demand shifters in these models.

Step 7 Quantify sensitivity of the integrated model to climate change.

To account for the effect of climate change on our model's results, we model projected changes in temperatures, precipitation, streamflow, and P transport using the most common IPCC scenarios (RCP 4.5, 6.0 and 8.5). The detailed climate inputs for the hydrology and crop models are derived from a modified change factor statistical downscaling method in which we preserve the monthly and annual variability from historical climate data in projected change scenarios (Cotterman et al. 2017, Basso et al. 2015). Projected changes in land use change across the study region has already been simulated through 2050 through ongoing research by our team based on the land transformation model (Pijanowski et al. 2007; Ray et al. 2012; Smidt et al. 2018). Based on these land use and climate projections, we simulate changes in crop yields using SALUS (Basso et al. 2015, 2013), and changes in hydrology and nutrient fluxes based on LHM. Finally we combine these outputs in statistical models that allow us to characterize the likely sensitivity of P inputs, algae growth, beach closures, and recreational opportunities for each scenario of changing climate and land use.

Model Applications

Our integrated systems model can be used to assess the effectiveness of potential agri-environmental policies and changes in farmer behavior on the value of beach recreation and fishing. Our integrated model provides a unique platform to evaluate conservation policies since 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 will estimate the effect of voluntary conservation programs that incentivize changes in P use, cover crops, and/or conservation tillage, on the consumer surplus associated with beach recreation and fishing trip demand. Using results from existing studies along with USDA data to develop counterfactual scenarios for agricultural management practices, we will estimate a lower bound gain in ecosystem services value. Varying the enrollment numbers and adoption levels, we can estimate associated changes in key ecosystem services benefits and determine the break-even cost that regulatory agencies would be willing to spend to achieve those higher levels of BMP adoption. We also estimate how changes in farmer liability, that may lead to the widespread adoption of soil tests combined with P fertilizer application restrictions or a tax on excessive soil P, affect the benefits that accrue to ecosystem services consumers. Model applications to policy can be illustrated with four policy evaluation examples:

Example 1. Model effects of existing conservation programs on value of ecosystem services.

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. We assess the effects of direct (e.g., nutrient management plans) and indirect programs (e.g., conservation tillage and cover crops) on the value of beach recreation and recreational fishing in Michigan's inland lakes and streams and coastal shores. This will be done by constructing a "without conservation programs" counterfactual scenario within the agricultural production model, and then feeding the changes through our integrated systems model to evaluate changes in beach and fishing use values. The counterfactual will be calibrated to existing data on conservation practices for those in and not in conservation programs (from NRCS data on EQIP and CREP enrollments, as well as data from the NASS Conservation Effects Assessment Program).

Example 2. Model effects of changes in liability with mandatory soil tests combined with P fertilizer application restrictions or a tax on excessive soil P on value of ecosystem services.

We will examine the expected implications of recent lawsuits that claim farmers are responsible for nutrient pollution, and of the 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, algal blooms, fish, and the value of beach recreation and recreational fishing in Michigan's inland lakes and streams and coastal shores.

Example 3. Model effects of changes in farmer behavior on the value of ecosystem services.

We will consider how changes in farmer knowledge about the effect of nutrient management practices on environmental services, social norms, attitude toward environmental stewardship, adoption of supply chain standards by large US retailers may affect the value of beach recreation and recreational fishing in Michigan's inland lakes and streams and coastal shores.

Example 4. Model the effects of targeted farmer incentives (across and within fields) on the value of ecosystem services.

The benefits of spatial targeting of incentive-based policies across fields are well-documented, but scant attention has been paid to addressing targeting by space and time within fields, as well as across fields. Taking advantage of the power of SALUS to dynamically target nutrient applications within fields and across growing seasons to minimize yield losses, we will examine the costs and economic benefits to ecosystem services of incentives to fine-tune fertilizer applications to minimize run-off and yield losses.

Challenges

Among the many challenges in constructing an Integrated Assessment Model, 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., 10m x 10m cells), most farm model are developed at 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 et al. (2015), for example, in distributing county level fertilizer levels to disaggregate (i.e., finer) individual cells, assume a proportionate adjustment to recommended application rates. While this is a reasonable starting point, it does not allow for farmers to optimally respond to policy changes. For example, in the face of restrictions to phosphorus 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 phosphorus 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 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; e.g., describing how water quality conditions have changed from "boatable" to "fishable" or "swimmable" over the course of a year. While such simple characterizations are driven by the desire to convey the impact of a policy initiative in 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 a waterway, making it more suitable for swimming, yet it may at the same time hinder the ability of the waterway to support a desirable fish species or the diversity of species. More work is needed to provide consumers with a clear characterization of the ecosystem services along all of the dimensions that impact the values they derive from water-based recreation.

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Appendix

Table A1: Ecosystem services linked to P concentrations in the ecological models.

Ecosystem Service	Parameter	Inland Lakes	Rivers/ streams	Coastal Zones
Recreation/Safety	Cyanobacteria	X		X
Recreation/Aesthetics	Chlorophyll a	X		X
Recreation/Aesthetics	Turbidity	X		X
Recreation/Aesthetics	Water Clarity	X		X
Recreation/Safety	Toxic Cyanobacteria	X		X
Recreation/Aesthetics	Benthic Algae - Chlorophyll		X	X
Recreation/Aesthetics	Benthic Algae - Cladophora		X	X
Recreation/Aesthetics	Benthic Algae - Lyngbya			X
Recreation/Aesthetics	Beach Algae	X		X
Fisheries	Game fish biomass	X	X	X
Non-use	Biological Condition: Fish	X	X	X
Non-use	Biol. Condition: Invertebrates	X	X	X
Non-use	Biological Condition: Algae	X	X	X