Solving the Phosphorus Pollution Puzzle:
Synthesis and Directions for Future Research

Cloé Garnache[a], Scott M. Swinton[a], Joseph A. Herriges[a,b], Frank Lupi[a,c] and R. Jan Stevenson[d]

[a] Department of Agricultural, Food, and Resource Economics, Michigan State University
[b] Department of Economics, Michigan State University
[c] Department of Fisheries and Wildlife, Michigan State University
[d] Department of Zoology, Michigan State University

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Abstract

Despite the success of efforts to reduce phosphorus (P) pollution from point sources, P from non-point agricultural sources remains a vexing problem with many U.S. water bodies having impairments. Key to solving the P pollution puzzle is to take stock of progress to date, the puzzle pieces available, and the gaps to be filled. In this paper, we synthesize the state of knowledge on P pollution, discuss the state of existing public programs, and review economists' contributions to informing P pollution policies. We review the water quality valuation literature, identifying limitations in the linkages to policy-relevant environmental quality metrics. We examine how and why P is used agriculturally, along with recent advances in market-based policy design and field testing. We survey new knowledge in biology and engineering, including improved understanding of the fate and transport of P. In light of recent learning and persistent knowledge gaps, we recommend directions for economic research to add needed pieces to the puzzle of how to protect our water bodies. Puzzle gaps meriting attention include mechanisms to target public funds more effectively in voluntary abatement programs, policy design for emerging mitigation technologies, new ways to implement performance-based policies, means to leverage social norms and behavioral cues, changes in the "pay-the-polluter" paradigm, and application of state-of-the-art evaluation methods to conservation programs. Beyond the realm of public policy lies that of private supply chains, where establishment of environmental standards holds additional promise. Rich research opportunities exist for economists in tandem with biologists, engineers, and others.

Keywords: Agriculture, Ecosystem services, Nonpoint source pollution control, Phosphorus pollution, Valuation, Water quality

JEL codes: Q18; Q51; Q53; Q58

Phosphorus (P) is one of the most widespread pollutants of inland waters and its main source is agriculture (Carpenter et al. 1998; Smith 2003; Bennett et al. 2001). Phosphorus affects 42% of the lakes and 66% of the river and stream miles in the United States and is largely responsible for harmful algal blooms in many lakes and rivers of the world (USEPA 2009a). Assessing the costs of this pollution requires an understanding of the complex interplay between the farm production process, the fate and transport of P once it is applied, and the biophysical processes that determined how P ultimately impacts those ecosystem services that consumers value. For example, recent advances in our knowledge about P transport and fate indicates that P not taken up by crops not only attaches to soil particles, but also runs off the field in the form of dissolved reactive P (DRP) (Tesoriero et al. 2009; Daloğlu et al. 2012). Attached P accumulates in the soil or bottom of streams and lakes, and creates legacies that can be released over several decades, making it difficult to predict P concentrations and attendant algal blooms (Hamilton 2012; Sharpley et al. 2013). DRP promotes algae growth and, thereby, adversely affects water quality and the value of ecosystem services such as recreation and aesthetic amenities. Estimating the disparate effects of P on the values consumers derive from inland lakes, rivers, and streams is essential to design P pollution policies capable of meeting consumer demand for water quality and ecosystem services.

Under the current U.S. legal framework, point sources are heavily regulated, while nonpoint sources such as residential runoffs, septic systems, and crop agriculture remain largely unregulated. Federally-sponsored voluntary conservation programs compensate agricultural nonpoint sources for the adoption of so-called best management practices (BMPs). Yet, despite annual budgets exceeding billions of dollars, these programs have had limited impacts on reducing agricultural nonpoint source pollution. With about half of the nation's water bodies having impairments (USEPA 2009b) and rapidly degrading water quality conditions in many places, meeting consumer demand for water quality calls for major policy changes to control nonpoint source P pollution. Economists have developed several efficient and compelling theoretical instruments, however, many of these theoretically "first-best" policies have unrealistically high informational or transaction costs. But a new generation of policies designed to incorporate the transaction costs that abound in the real world have the potential to greatly enhance both environmental benefits and cost-savings. Such policies include better targeting of funds and auction mechanisms, adoption of performance-based policies (e.g., via the use of simulation models or environmental proxies), leveraging of social norms and behavioral cues, and changes in the “pay-the-polluter” paradigm. Beyond the realm of public policy lies that of private supply chains, where establishment of environmental standards holds additional promise.

We argue that addressing the P pollution challenge requires solving a puzzle with many moving pieces, some of them currently missing. Using a systems framework approach, the present review paper examines the P puzzle broadly, as illustrated in Figure 1. The broad puzzle framework highlights pieces that represent intervention points for policy, pieces that represent recent advances that have changed traditional views, and other puzzle holes where knowledge gaps still remain. We identify five areas of focus: institutions and economic drivers, economic agents, biophysical processes, environmental responses (intermediate ecosystem services), and end-point ecosystem services that consumers value. Jarvie et al. (2012) introduce the P puzzle metaphor, identifying the P legacy as a key puzzle piece for biological processes. We broaden the puzzle metaphor to frame the main challenges and potential solutions to P pollution.

In this paper we describe the P pollution puzzle, synthesize existing knowledge, highlight knowledge gaps and suggest future directions for research. First we discuss the problems P pollution causes in freshwater ecosystems and how it links to farmer P management decisions. We then describe the contributions of economic research on informing water quality policy and highlight a mismatch between traditional water quality measures reported in the valuation literature and measures useful to policy makers. We present recent advances in knowledge on the science of P and new information on the performance of policy instruments in the real world. We close by identifying new research gaps with special focus on economics where finding new pieces promises to bring us closer to solving the P puzzle.

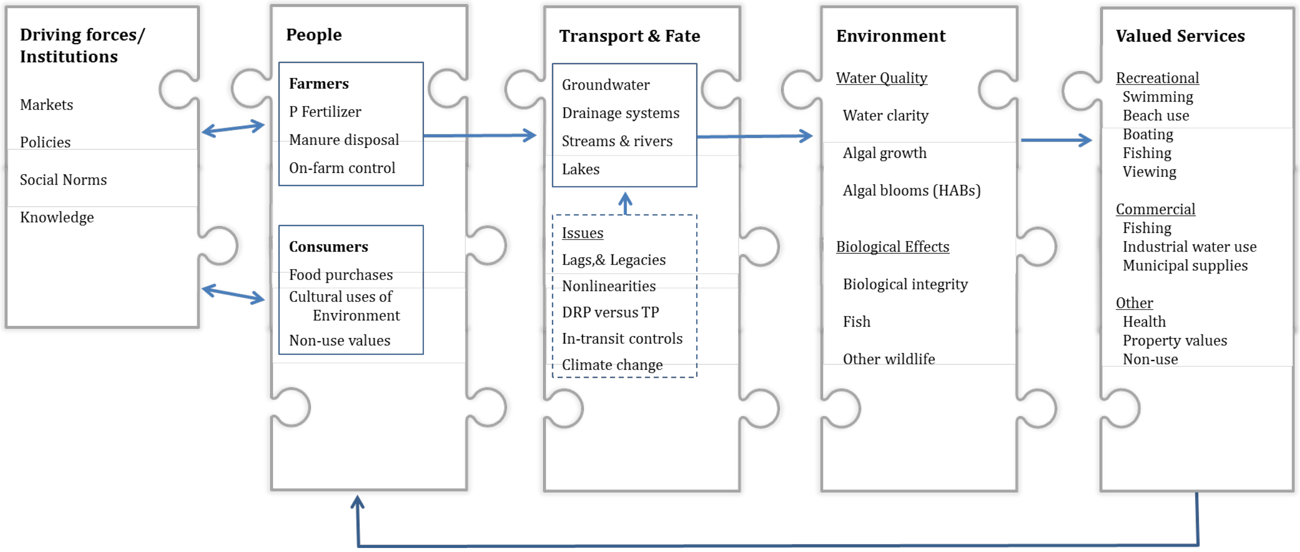

Figure 1: Systems framework illustrating the P puzzle

The P Puzzle

Phosphorus concentrations vary naturally in freshwater ecosystems, and agriculture is the largest source of P runoffs in the United States. Next, we assess the effects of P on freshwater ecosystems, which although often largely adverse may sometimes be positive.

Effect of P on Ecosystems

Phosphorus is the most limiting nutrient for algae and bacteria in most freshwater ecosystems, whereas nitrogen is usually most limiting in coastal marine ecosystems. So when P becomes abundant, algal populations can explode in algal blooms. Excessive growth of algae and bacteria resulting from P pollution affects many ecosystem services. The ability to treat water for drinking is threatened directly by toxic, bad tasting, or malodorous chemicals produced by abundant algae (Arruda and Fromm 1989; Falconer 1999; Watson 2004). In excess, planktonic algae can make lake water turbid, and benthic algae can make beach lake bottoms slimy, repelling recreational users (Poor et al. 2001; Suplee et al. 2008). Worse yet, masses of algae on beaches and in water can harbor and increase persistence of pathogenic bacteria (Ishii et al. 2006). Abundant growth of algae and bacteria also deplete biodiversity by reducing oxygen concentrations to anoxic levels and by physically altering habitat structure (Lasenby 1975; Wetzel 2001; Stevenson et al. 2012, Figure 2).

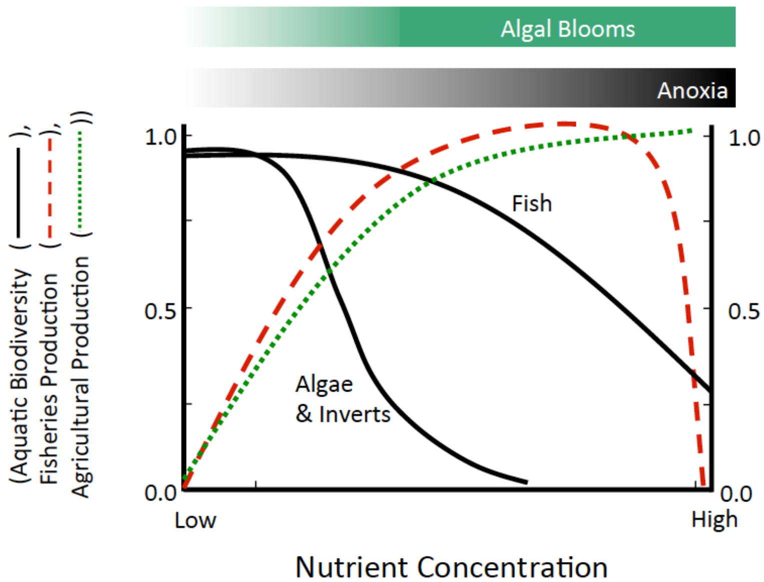

Figure 2: Tradeoffs between nutrient concentration and freshwater ecosystem services: algal, invertebrate, and fish biodiversity, fisheries production, and agricultural production (modified from Stevenson and Sabater 2010).

Tradeoffs among aquatic ecosystem services occur along P gradients, which are common for most ecosystems and human disturbances (Ayensu et al. 1999). Low levels of P pollution are usually below benchmarks that threaten treatability of water for drinking, aesthetics, recreational use, and biological condition (Figure 2). Intermediate levels of P pollution can support high fisheries production and related provisioning services of aquatic ecosystems (e.g., Vaux et al. 1995; Esselman et al. 2015). High levels of P pollution cause harmful anoxic conditions in streams and lakes.

Phosphorus has a complex effect on fisheries. Up to moderate concentrations of P, P has positive effects on fisheries production via enhanced primary production and algal biomass, but negative effects on fish biodiversity (Miltner and Rankin 1998). For example, game species such as trout are much more sensitive to higher P levels that other species (Esselman et al. 2015). Beyond moderate concentrations, oxygen levels fall too low for survival of high quality game and food fish. Phosphorus pollution also harms biological condition, a measure of the similarity of a biological assemblage to one with biological integrity (Davies and Jackson 2006). Many natural or minimally disturbed habitats are home to few species because scarce nutrients limit primary production and food supplies throughout the food web (Stevenson et al. 2008). As P levels increase, so does habitat productivity, enabling new species of algae, bacteria, and animals to invade and potentially change habitat conditions so that sensitive native species can no longer survive. Thus as P pollution increases, new species can invade and potentially cause problems, the total number of species that can live in a habitat increases, but sensitive native species can be lost and ecosystem functions can change from a low-P natural state.

Phosphorus concentrations do vary naturally in streams, rivers and lakes, as a function of geology and climate (Olson and Hawkins 2013). Rainfall complexly affects P concentrations because rainfall can dilute P concentrations in streams, but high rainfall can cause soil erosion (Stevenson et al. 2013).

Phosphorus from agricultural sources is largely caused by runoff from fields with inorganic P adsorbed onto eroding sediment particles (Daniel et al. 1994; Sharpley et al. 1994). Some organic P, either in dissolved or particulate form, also runs off fields as organic matter from decomposing crops. Thus riparian buffer strips have been proposed as an effective means of controlling P runoff in particulate form. Recent research has shown other important routes of P into streams and downstream waters via dissolved inorganic and organic forms of P (Tesoriero et al. 2009). With rainfall, this dissolved P can flush through shallow surface soils to natural or man-made groundwater channels. Groundwater P transport is also a problem with tile drains in fields (Gentry et al. 2007; Kleinman et al 2015b). No-till agriculture may exacerbate the problem of P loading by keeping organic matter in surface soils, increasing their decomposition rates, and releasing dissolved P into the groundwater (Daloğlu et al. 2012).

P in Agriculture

Farmers apply P to the land in two forms: as mineral fertilizer and as livestock manure (Figure 3). The objective of fertilizer application is to boost crop yield. Manure application may be motivated by one or both of two objectives: boosting crop yield and/or disposing of a waste product of livestock operations.

Phosphorus can accumulate in agricultural soils when the rate of application exceeds that of plant uptake (Brady and Weil 2002). Where soil P is deficient, either form of P can boost yields, but when soil P reserves exceed crop needs, no yield gain will result. At high levels, accumulated soil P becomes a source of P pollution. Phosphorus accumulation in agricultural soils can occur anywhere that application rates exceed uptake, but it tends to be most serious where concentrated livestock production generates large quantities of manure. Manure contains P (as well as N and organic matter). It is bulky and costly to transport, creating an incentive for disposal by spreading nearby. Many states have regulations that limit manure disposal when soil P levels exceed a designated threshold (USEPA 2012). The fact that soil P is an essential plant nutrient that becomes a pollutant at high levels accounts for the manure value paradox: that farmers will pay to acquire manure where soil P is deficient, while they pay to dispose of it where soil P is overabundant (Hoag and Roka 1995).

The demand for mineral fertilizer P arises from the biological crop yield response function and input and output prices, as well as farmer attitudes toward risk and environmental stewardship, and other factors (Figure 3). Crop yield response to P has generally been found to reach a plateau, beyond which there is no yield response (Ackello-Ogutu, Paris, and Williams 1985; Frank, Beattie, and Embleton 1990). These yield response studies, which were conducted with long-term, agronomic trial data, suggest little or no input substitution with nitrogen. While this finding is unsurprising to plant nutrition scientists, it has been a point of debate due partly to the early history of modeling crop yield response to nutrients with polynomial functions (Hexem and Heady 1978).

Crops are relatively inefficient at taking up P. First, they can take up only P that is biologically available to the crop root. In addition, farmers often apply fertilizer in the fall when crops are either not in the field or using small amounts of nutrients allowing much of the applied fertilizer to be washed away with rainfall via overland flow and leaching. Second, crops take up only a very limited share of the biologically available P (Brady and Weil 2002). Inefficient P update can induce farmers to apply additional P fertilizer even to soils that already contain a significant reserve of P, especially if P in the root zone is mostly not in plant-available form. Nonetheless, in many agricultural regions of the United States, soil P levels have been rising, leading farmers whose fertilization decisions rely on soil tests to limit P applications to replacement of what crops remove.

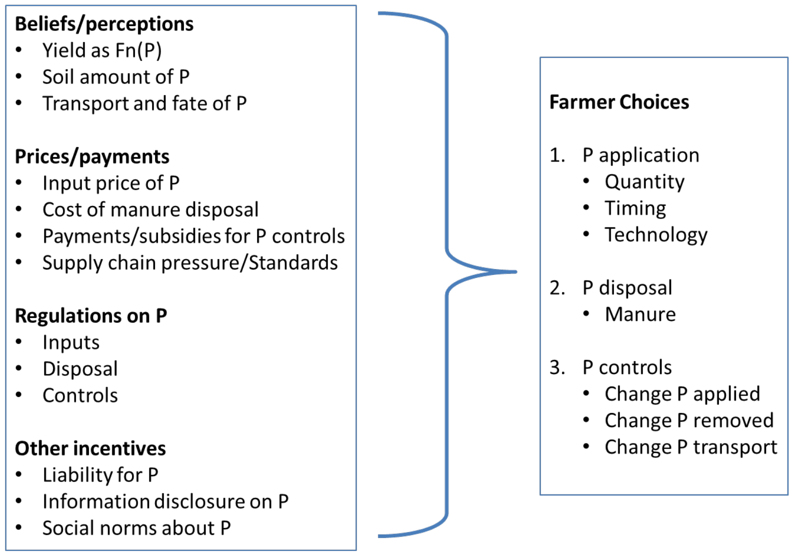

Figure 3: Economic and social drivers affecting farmers' P management decisions

Regulating P from agriculture is fraught with many challenges. In addition to P pollution being unobservable and stochastic due to diffuse and complex pathways, the fate and transport of P outside the field, and therefore the damages associated with P loads, depend on the location of the farm relative to water bodies. In general, the farther the farm from the water body, the more "attenuated" the P load to a given water body or end-point. Consequently, estimating P pollution requires reliance on simulated load responses from process-based or statistical models such as SWAT. Phosphorus loads come from many farms, each with different production costs and P loading processes, so the opportunity costs of abatement are highly heterogeneous. As a result, uniform standards can lead to economic inefficiencies.

Further complicating the regulator's task is the low price elasticity of demand for P as a factor input, particularly in U.S. corn sector over both the short and long term (Denbaly and Vroomen 1993). Earlier research over large regions of the country and multiple crops (Carman 1979) similarly found price inelastic demand for P fertilizer, albeit not as low. Price elasticity of demand for all crop fertilizers declined after the early post World War II period, when synthetic fertilizers were new (Rausser and Moriak 1970). Evidence of price-inelastic demand for P is consistent with the stability of total U.S. consumption of P fertilizer, which has fluctuated in the range of 4.0-4.6 million short tons over the 25 years 1986-2011 (USDA ERS 2015).

Fertilizer decisions are made under uncertainty because the demand of the crop is affected by weather conditions and farmers do not know the amount of P that will be biologically available to the crop root. Risk preferences may augment the demand for fertilizer, beyond what would be expected based on average prices and yield response. While a large body of research has documented risk aversion as a deterrent to fertilizer use by early adopters in developing countries, there is evidence that Iowa corn farmers over-apply nitrogen fertilizer as insurance against the risk of nutrient shortage causing yield to fall below optimal levels (Babcock and Blackmer 1992). We are aware of no studies that have tested the hypothesis that risk aversion increases P fertilization rates.

There is evidence that environmental stewardship motives are associated with adoption of BMPs, including reduced fertilizer rates and targeted fertilizer application (Ma et al. 2012; Prokopy 2008). However, while environmental attitudes do affect farm stewardship behavior, there is little evidence connecting farmer attitudes to fertilizer demand.

Several recent changes in the U.S. agricultural scene could change demand for P fertilizers. First, beginning in 2008, fertilizer prices jumped and became more volatile as row crop acreage expanded in the United States in response to high cereal and oilseed prices during 2007-13 (USDA ERS 2015). Crop prices have subsequently dropped proportionately more than fertilizer prices, which could reduce P use in the short run. Second, there is rising concern about depletion of global rock phosphate deposits, which could sustain recent increases in P fertilizer prices (Vaccari 2002). Third, as noted above, scientists are becoming aware that DRP drained from farm fields into drainage tile is becoming a significant source of P in water bodies (Kleinman et al. 2015b). Farmer awareness and information are important drivers of BMP adoption (Baumgart-Getz, Prokopy, and Floress 2012), so information about P loss from drainage tile could affect P demand. Nonetheless, the long-term inelasticity of P demand suggests that such effects will be slight.

Current U.S. Water Quality Policy

The Clean Water Act (CWA) provides the broad legal framework for controlling water pollution in the nation's waterways (1972, and as amended, U.S. Code title 33, sections 1251–1387). The interim goal of the CWA "provides for the protection and propagation of fish, shellfish and wildlife and provides for recreation in and on the water" (CWA 1972 as amended, section 1251(a.2)). The ultimate goal of the CWA is "restoration and maintenance of chemical, physical, and biological integrity of Nation's waters" (CWA 1972 as amended, section 1251(a)). The USEPA provides guidance for states to implement the Clean Water Act within approved boundaries. Water quality standards are one of the key mechanisms for managing waters. Water quality standards are composed of three parts: a designated use, criteria for waters to meet those uses, and an anti-degradation policy. Examples of designated uses are drinking water, recreation, fisheries, water supply for agriculture and industry, navigation, and aquatic life use. The anti-degradation policy is intended to prevent further degradation of surface water uses. When water quality criteria are violated and a water body is declared to be "impaired", a restoration plan that includes pollution management targets is usually required. Pollution management targets are commonly defined as total maximum daily loads (TMDLs). Management options to reach TMDLs in most states are limited to managing point sources.

Point sources must comply with effluent technology standards and must obtain a permit through the National Pollution Discharge Elimination System (NPDES) in order to discharge effluent to surface waters. Since 2003, waste handling and disposal by large confined animal feeding operations (CAFOs) have also been subject to federal regulations and also require an NPDES permit (Claassen, Cattaneo, and Johansson 2008). Nonpoint sources such as agricultural fields remain largely unregulated and are responsible for the large majority of agricultural nutrient pollution (Shortle et al. 2012).

The current framework for achieving water quality goals has been to increasingly tighten regulations on point sources pollution but to exempt nonpoint sources from regulation. Clearly, marginal abatement costs are not being equated across point and nonpoint sources, and this lack of policy coordination across sectors puts a heavy burden on point sources while failing to capture lower-cost abatement from agricultural nonpoint sources. To make matters worse, point source pollution abatements are achieved through uniform standards (e.g., Freeman 2000; Shortle and Horan 2013), which tend to be costly and provide little flexibility to achieve environmental goals. The Clean Water Act exempts agricultural nonpoint sources from regulation (unless a water body is declared to be "impaired"), in effect granting farmers the right to pollute. As a result, farmer efforts to abate P discharges are voluntary, although restrictions may apply in watersheds with a TMDL. Overall, the effectiveness of voluntary conservation programs has been limited, and agricultural nonpoint sources remain largely unregulated (Kling 2011; Shortle et al. 2012).

Voluntary conservation programs supported by financial and technical assistance from federal and state sources are the main policies to promote agricultural nonpoint pollution control. The Conservation Reserve Program (CRP) and Environmental Quality Incentives Program (EQIP) are the two largest federally-sponsored voluntary conservation programs. They provide cost-share or direct payments for conservation actions on agricultural lands, with annual budgets of several billion dollars (Claassen, Cattaneo, and Johansson 2008). While land retirement through the CRP often provides greater environmental benefit per acre, larger environmental gains per program dollar may be achievable through the adoption of conservation practices working lands under EQIP. In 2002, funding for conservation effort on working lands increased sharply (Claassen, Cattaneo, and Johansson 2008). However, several changes in EQIP may have lowered its cost effectiveness, notably reducing emphasis on benefit-cost targeting, reducing targeting at farm and watershed levels, eliminating bidding for financial assistance, and focusing more resources on assisting producers with regulatory compliance.

Economic Research Contributions

Economic research in the period since initial passage of the CWA has made major advances that contribute to solving the P puzzle, both in measuring the value of ecosystem services to society and in designing efficient pollution control policy. We assess the advances in valuation methods and highlight shortcomings that limit the utility of many valuation studies for policy implementation and evaluation. We then review progress on the design of environmental policy, including important advances in performance-based instruments such as pollution limits and taxes, ambient taxes, and pollution permit markets.

Valuation of Ecosystem Services

The valuation of ecosystem services plays a critical role connecting pieces in the P puzzle (Figure 1). Phosphorus loadings from agricultural and other sources move through the environment impacting ecosystem services (e.g., fish stocks, water quality, etc.) that consumers ultimately care about. Understanding the economic value of these changes is essential for policymakers as they seek to balance the costs of competing regulatory policies against the potential benefits to society. The measurement of environmental values may also influence markets by influencing social activism and supply chain standards.

Significant strides have been made in the past twenty-five years in valuing environmental amenities. Revealed preference techniques, including recreation demand and property value models, have become increasingly sophisticated in terms of both their economic and econometric foundations. As Phaneuf and Smith (2005, p. 673) note, "…[t]oday, economic analyses of recreation choices are among the most advanced examples of the microeconometric model of consumer behavior in the literature." The same can be said of recent advances in the property value and equilibrium sorting literatures (e.g., Kuminoff, Smith, and Timmins 2013). Stated preference approaches, including contingent valuation and choice experiments, have undergone even more scrutiny than their revealed preference counterparts, with Carson (2011) identifying over 7500 contingent valuation studies alone. While not universally held (e.g., Hausman 2012), the general consensus in the literature is that a well-designed stated preference study provides a useful starting point in assessing the use and non-use values associated with environmental improvements (e.g., Kling, Phaneuf, and Zhao 2012; and Carson 2012).

Despite this progress in both revealed and stated preference methods, their application in the policy arena remains challenging, particularly in the context of water quality regulations. The difficulties partly reflect the fact that much of the published economic literature exists to examine theoretical or methodological issues, such as convergent validity or the efficacy of alternative elicitation formats. It places much less emphasis on isolating the value of changes in a specific ecosystem service, such as fish stock or water quality, stemming from a given change in, say, nutrient loadings. Other studies focus on measuring the total value of changes in specific water quality measures (e.g., Secchi transparency), but without identifying the corresponding changes in the ecosystem services that people value. This makes it difficult to understand what precisely is being valued. There is, however, a growing awareness in the literature of the need for a more integrated understanding of how ecosystem services are provided and valued (e.g., Keeler et al. 2012 and van Houtven et al. 2014). These frameworks envision tracing changes in environmental inputs (such as nutrient loads) driven by a policy initiative through to changes in water quality and ultimately to changes in ecosystem endpoints that individuals understand and value.

Griffith et al. (2012, p. 131) identify three specific challenges in valuing changes to water quality changes: “standardizing measures of water quality to facilitate benefits transfer, estimating the benefits from ecological improvements, and strengthening methods for estimating nonuse values.” In most settings, time and budget considerations preclude the development of a nonmarket valuation study specific to the regulatory policy being evaluated, hence the need for benefits transfer. The transfer task, however, is hampered by the lack of a consistent metric for quantifying both the water quality changes and the ultimate changes to ecosystem services. Revealed preference approaches, such as hedonic property value and recreation demand studies, are particularly hampered in this regard, typically limited to one or two readily available water quality measures, such as Secchi transparency (Michael, Boyle, and Bouchard 2014), water toxin levels, or fish catch rates (e.g., Phaneuf et al. 2000). These are rarely tied to the ultimate ecosystem services individuals are presumably valuing.[1]

Stated preference methods, such as contingent valuation and choice experiments have considerably more flexibility in this regard. The challenge here is often balancing the benefits of additional detail regarding the policy scenario being valued against the potential risk of losing the respondent’s attention. In some settings, and for some segments of the population, it is critical to disaggregate the ecosystem services provided by the policy scenario (e.g., changes in walleye versus general fish stock), whereas for others more aggregate measures suffice (See, e.g., Boyd and Krupnick 2013; Boyd et al. 2015). In many stated preference studies, the water quality changes are characterized in terms of discrete increments in broad activities they allow for, using for example the water quality ladder, which distinguishes water bodies suitable for boating, fishing, swimming and drinking. The problem here is that the use of categories themselves are vague, with interpretations varying across individuals, and are often only loosely tied to the underlying biophysical changes induced by a proposed policy. In other cases, a continuous water quality index (WQI) is used, as in the case of the National Sanitation Foundations WQI (Horton 1965; McClelland 1974). Van Houtven et al. (2014) use a discrete

[1] While fish catch rates may seem to be tied to an ecosystem service, they confound the service endpoint itself (fish stock) and the effort put into fishing by the individual. A recent exception links river angler choices to fish biomass (Melstrom et al. 2015), which is independent of fishing effect. The fish biomass estimate was also explicitly linked to P concentrations (Esselman et al. 2015), thereby connecting recreational values to the service endpoint and back to P via an ecological production function.

five-point eutrophic index, explicitly linked to a lake's water clarity, color, algae, odor, and aquatic health. Both of these metrics have the advantage of linking the water quality changes proposed to the physical changes induced by a policy. However, such indices also suggest that all ecosystem services increase monotonically with the index, which need not be the case (Figure 2). While both fish stocks and drinking water services provided by a lake may initially improve with an increment in a given water quality index, fish stocks may actually decline if a water body becomes too clear, no longer providing necessary food sources for the fish. Moreover, it is not clear how the particular metric used to measure and communicate water quality changes impacts the benefit estimates themselves. Meta-analyses, such as in Johnston et al. (2003) and Brouwer et al. (1999), provide some insight into this question, but the cross-study nature of meta-analysis makes it difficult to isolate the causal effect of individual water quality measures on subsequent benefit estimates. What is needed is a controlled comparison of these alternative metrics within a single study.

The second problem identified by Griffith et al. (2012) is the challenge of estimating the benefits from ecological improvements. While regulatory programs may yield measurable changes in the physical and biochemical characteristics of a water body, what ultimately matters to consumers are the perceived changes in the ecosystem services provided. Linking the water quality changes to a range of observable and relevant ecosystem service endpoints requires an understanding of the biophysical processes involved and the spatial scale at which they operate. It is a task that is necessarily interdisciplinary in nature. It also requires an understanding of how consumers perceive these linkages. As Boyd and Krupnick (2013) suggest, some individuals asked to value a specific improvement in water quality may simply value the change in terms of its direct impact on aesthetics or the consumptive use of the water. Others, however, may bring an understanding of the broader implications of water quality changes in terms of species habitat, etc. Even if their understanding is incomplete, the set of ecosystem services they are ultimately valuing is likely different. These factors can also impact the spatial scale at which a valuation exercise is effectively operating. Some survey respondents may understand that water quality improvements in the headwaters of a stream have implications for downstream waters. Their valuations, use and nonuse, will reflect the broader changes in ecosystem services, while a less informed respondent's valuation will be more narrowly focused. Understanding the information and experiences that a respondent brings to a valuation exercise is critical to understanding (and potentially transferring) the resulting benefit estimates.

Finally, the total value of water quality improvements includes both the use and nonuse values held by individuals. While some stated preference valuation techniques provide a measure of the total value, disentangling the use and nonuse values remains an area of ongoing research. A variety of techniques have been employed in the literature. Johnston, Besedin, and Wardwell (2003) identify five approaches: "(1) nonuse values identified as the total WTP for nonusers; (2) responses to separate nonuse value questions in the survey; (3) apportionment of total WTP among categories of value by survey respondents; (4) nonuse values estimated as the total WTP of a survey sample who were asked to assume they would not use the resource being valued; and (5) total WTP of the sample of users minus estimated WTP for the direct use of the resource estimated based on revealed preference data." While the first approach is the most common in the literature, it implicitly assumes that nonuse values are the same for users and nonusers, which need not be the case. At a minimum, as noted above, users and nonusers may bring to the valuation task a different understanding of the scope and spatial scale of ecosystem services being provided by a regulatory program and, hence, be valuing different services and over different scales. The problem with the fifth approach is that is assumes that use values can be uniquely identified from revealed preference data, which is not the case (See Herriges, Kling, and Phaneuf 2004). It also implicitly assumes that nonuse values are the same for users and nonusers. An alternative approach is to simultaneously model the valuations of users and non-users in an integrated framework, an approach explored in Eom and Larson (2006), Egan, et al. (2007), and Egan (2011). In the latter case, a Kuhn-Tucker framework is used to model observed use of a recreation resource, together with contingent behavior and contingent valuation, in a unified structural model. Such models, of course, are complex and unlikely to see widespread application. What is needed is a comparison of the resulting use and non-use valuations for both the complex modeling approach and the simpler methods currently employed in the literature.

The CWA and the need for information on the economic benefits of water quality has been one of the driving forces for innovation in the field of environmental valuation. Yet critical research needs remain, especially for connecting values to policies within the P puzzle.

Policy Design for P Abatement

Economists have made major advances in designing and testing policy instruments that account for heterogeneity in marginal damages and abatement costs across polluters, and are flexible rather than prescribe a one-for-all action to achieve the environmental goal.

Some market-based instruments such as auctions have been incorporated in current programs to manage agricultural landscapes more cost-effectively. Auction-based conservation programs can minimize abatement costs since only the farmers who submit cost-effective bids are selected. In practice, auctions are almost exclusively used for CRP contracts, according to Claassen, Cattaneo, and Johansson (2008). They find that the introduction of bidding mechanisms to allocate funds to farms with high environmental benefits potential per dollar has modestly reduced program costs. A key factor limiting the ability of auctions to lower program costs is that the ability of bidding mechanisms to reveal a bidders minimum willingness-to-accept crucially relies on the bidder's information relative to their bid's competitiveness. Repeated sign-ups in the CRP over time allow farmers to learn about their relative ranking, allowing them to adjust their bids to extract more economic rent and reducing the cost-effectiveness of the auction mechanism (Claassen, Cattaneo, and Johansson 2008*).* The cost-effectiveness of auction-based programs at achieving environmental benefits may further be hampered by non-environmental targeting criteria such as EQIP's special preferences for limited resource (poor) farmers (Claassen, Cattaneo, and Johansson 2008). Likewise, funds are not being allocated to states based on environmental benefit-cost considerations because Congress prohibits favoring farmers in one region over those in other regions, with the exception of some areas of concern such as the Chesapeake Bay watershed (Claassen, Cattaneo, and Johansson 2008; Shortle et al. 2012).

Existing voluntary programs are practice-based rather than performance-based, even when environmental performance measures are readily available and already in use to rank farmer bids relative to environmental benefits. Indeed, environmental performance tools are used to guide the decisions on which CRP contracts to accept but not to estimate payment levels associated with the environmental benefits. Furthermore, the extent to which the adoption of current best management practices (BMPs) can improve water quality may be limited, because they emphasize reducing the transport of P to the water body by trapping nutrients on the field or in filter strips, rather than nutrient input reduction. Many of the BMPs that are eligible for financial assistance have only been shown to be effective at controlling or trapping nutrients on the field at the plot level but not at the scale of the watershed (Sohngen et al. 2015). Current federal voluntary programs do not rely on available biophysical models to approximate environmental performance based on observable actions (Kling 2011; Shortle et al. 2012). Because agricultural water pollution is considered to be nonpoint source, there is no monitoring of nutrient discharges from drainage tile lines, which are widespread in the Midwest.

Conservation programs have provoked considerable debate over "additionality," the question of whether voluntary programs with payment incentives actually induce farmers to adopt practices they would not have adopted absent the payment. Serious doubts exist on the extent to which voluntary programs have led to additionality (Claassen, Cattaneo, and Johansson 2008; Segerson 2013). A study in Ohio finds that cost-share programs are associated with high levels of additionality for hayfield establishment (above 90%) but low levels for conservation tillage (below 20%) (Mezzatesta, Newburn, and Woodward 2013). Another concern that can undermine the environmental benefits associated with land retirement programs is slippage that leads to land use displacement, whereby farmers enroll land in the CRP, while they expand cropland elsewhere (Wu 2000).

Among environmental policies designed to abate P application, the simplest policy is a fertilizer tax. A theoretically optimal fertilizer tax must be farm-specific to conform to heterogeneous damages. However, given the enormous information requirements and transaction costs of implementing such a tax (Lichtenberg 2004), a uniform input tax is a more practical alternative. Input tax analyses show poor and uneven benefits due to inelastic demand response to P fertilizer price and heterogeneity in nutrient runoffs (Vatn et al. 1997; Claassen and Horan 2001; Shortle and Horan 2013). Recent mathematical programming bioeconomic models have found that extremely high P input taxes would be required to significantly reduce P demand, results consistent with econometric evidence of the price inelasticity of demand for P. A 900% tax on P fertilizer was found to be insufficient to reduce demand by 40% in the Minnesota River valley (Westra 2001), while a 100% fertilizer tax had negligible effect on simulated P runoff in a bioeconomic modeling study of bioenergy cropping systems in southern Michigan (Egbendewe-Mondzozo et al. 2013). In contrast, a recent study in Ohio points to estimated reductions in soluble P loadings in Midwestern agricultural watersheds ranging between 1.7% and 3.4% in response to a 10% increase in fertilizer price, which appear similar in magnitude to price elasticity of P demand. A 25% tax on P fertilizer was found to reduce nutrient loadings from three watersheds into Lake Erie by 6.5%, for a cost of $6 per hectare per year, which is similar to costs of wastewater treatment plants but less costly than commonly used agricultural best management practices (Sohngen et al. 2015).

Pollution taxes cannot be implemented directly because P loads from nonpoint sources cannot be observed, although they can be achieved with the use of proxies and biophysical models (Kling 2011; Shortle and Horan 2013). Another instrument that is theoretically compelling is an ambient tax, since it does not rely on monitoring P loads from individual sources. However, for this instrument to be efficient farmers must understand the pollution generation process on their farm and understand and predict the aggregate response of other polluters, which is likely unrealistic (Segerson 1988).

Markets for tradable pollution permits can enable meeting pollution abatement targets, while minimizing total abatement costs. Designing efficient tradable permit markets requires defining the trading ratio between point and nonpoint sources, and capping the aggregate supply of permits such that the market can achieve the target water quality goals (Shortle and Horan 2013). Permit trading programs have been introduced to supplement point source regulation in impaired watersheds to facilitate compliance with TMDL restrictions, e.g., in the Minnesota River Basin and in the Chesapeake Bay (Shortle et al. 2012; Fisher-Vanden and Olmstead 2013).

Economists have helped develop the conditions under which strict liability and negligence rules are likely to succeed as policy tools for pollution control (Segerson 2007). Liability and common law remedies have seen little success for nutrient pollution of water from agriculture, with the exception of cases of negligence for concentrated sources such as CAFOs that fail to comply with a discharge permits or suffer failures that lead to releases (Ogishi, Zilberman, and Metcalfe 2003; Laitos and Ruckriegle 2013). However, most economic assessments of liability as a policy tool for agricultural pollution have suggested that the legal standards for negligence and the nature of agricultural nonpoint source pollution pose many challenges for the use of liability (Ribaudo, Horan, and Smith 1999). As such, cases involving liability over P are limited. Many cases that do exist address more concentrated P sources from agriculture. For example, Kleinman et al. (2015a) provide details of a natural resource damage case involving Oklahoma water users and poultry farms in Oklahoma and Arkansas. In 2001 and 2004, the City of Tulsa and State of Oklahoma filed law suits claiming that P from upstream poultry farms was contributing to eutrophication of reservoirs along the Illinois River that the city uses for water supplies. To date, there is no final judgement on the case, but the case led to P management standards that restrict application of P to soils below a soil test threshold. In part due to changes spurred by the lawsuit, P concentrations have decreased by one-third and poultry farms have adjusted to the regulations. A key adaptation was the export of poultry litter outside the watershed as a marketed agricultural commodity.

In another case, Waterkeeper v. Hudson, a CWA citizen suit was brought against a Delaware poultry farm for degradation of water quality in Chesapeake Bay (Egan and Duke 2015). As in the Oklahoma case, the poultry farm had characteristics of both point and nonpoint source pollution. A judge ruled that the group bringing the suit did not prove the farm polluted the Chesapeake Bay. An analysis of the case suggests that citizen suits under the CWA are poorly suited to achieve water protection goals, but that future citizen suits are likely absent legislative or other institutional changes (Egan and Duke 2015).

For more dispersed sources of P where tracing pollution to likely sources is challenging, liability is recognized as less likely to be effective than other incentives (Ribaudo, Horan, and Smith 1999). Under the CWA, states have the responsibility for regulating nonpoint sources of P, and most states make assignment of liability to agricultural producers difficult. For example, in Ohio the state can hold farmers liable for failure to use BMPs that result in pollution of the state's waters, but the process is cumbersome with many steps and few mandated practices (Kilbert, Tisler, and Hohl 2012). Thus, before an agricultural source is held liable in Ohio, a complaint must be made against a landowner, the state must then issue an order, and the owner must then be found out of compliance with the order. Though at present liability plays a limited role, it has more promise as a policy tool to help solve the P puzzle as future technological innovations improve capacity to link P consequences to sources.

Recent Pieces Added to the P Puzzle

During the past decade, new pieces of information have been added to the P pollution puzzle, filling some holes while also revealing other holes of unexpected shapes. The new information adds economic, political, and biological pieces to the puzzle. The new economic pieces involve empirical tests of theoretically efficient policy instruments. A common finding has been that these instruments have fallen short of their theoretical efficiency for a variety of reasons.

High transaction costs, regulatory uncertainty, and scant participation by unregulated farmers leading to "thin" markets are impediments to well-functioning nutrient pollution markets (Stephenson, Norris, and Shabman 1999; King 2005; Ribaudo and Nickerson 2009). Few nonpoint-to-point source trades have occurred in practice in existing tradable markets because of the absence of caps for nonpoint sources, resulting in a lack of incentives for agricultural nonpoint sources to participate (Shortle et al. 2012; Fisher-Vanden and Olmstead 2013). As a result, trading activity remains low and markets have failed to deliver large efficiency gains.

Experiments with reverse auctions to allocate payments for environmental services in a cost-effective manner point to transaction costs as a major hindrance to efficient allocation. Low participation in such auctions reduces the population sample of bidders (Palm-Forster et al. 2016; DePiper 2015; Peterson et al. 2014; Alston, Andersson, and Smith 2013). Auctions that require detailed applications to enable modeling of environmental outcomes from alternative conservation practices (like P delivery to streams) attract particularly low participation rates. Thin participation results in payments for lower impact practices than would occur with high participation (Palm-Forster et al. 2016). As auctions are costly to implement, the failure to find off-setting cost efficiencies can explain why the USDA Natural Resource Conservation Service has continued to rely on cost shares and fixed stewardship payments (under the EQIP and the CRP).

The recent blossoming of economic research into program evaluation in health care, education and international development invites examination of the effectiveness of conservation programs in the United States. Although the program evaluation toolkit for establishing causality has seen some applications to agricultural conservations programs, existing applications tend to examine effects on acreages enrolled, additionally of enrolled acreages, and outcomes related to farmer wellbeing. However, there are critical unmet needs for applying the suite of program evaluation tools to conservation programs to assess their ability to deliver changes at the watershed scale and to generate downstream improvements in valuable ecosystem services. While it may be politically challenging to perform watershed-scale randomized controlled trials of BMPs, this type of evidence is vital for our understanding of efficacy where we know the least and where it often matters most, downstream.

Outside the research sphere, the U.S. political environment has become less and less receptive to environmental policy initiatives. Yet in the private sector, perhaps stimulated by the booming market for organic products (National Research Council 2010), food company initiatives abound to mainstream and market environmental stewardship. A broad focus is to establish criteria for certification of good stewardship. A coalition of major food companies under the Field-to-Market umbrella (https://www.fieldtomarket.org/members/) has developed a Fieldprint Calculator as a metric for establishing minimum standards for environmental protection (https://www.fieldtomarket.org/fieldprint-calculator/). A separate coalition of fertilizer companies and agricultural input suppliers has developed the 4R's program to shift the business model of agricultural input suppliers from commodity agrochemical sales to value-added crop consulting. The 4 R's program centers around soil testing and precision application methods (http://www.nutrientstewardship.com/what-are-4rs), enabling participants to certify good stewardship as defined by avoiding unjustified agrochemical applications. To date, these efforts have attracted scant academic research into their potential effectiveness.

The past three decades have yielded new advances in biological and engineering research that change the parameters for P policy. First, while it has long been known that P moves slowly through the environment, only recently has the importance of legacy P become apparent. Phosphorus in surface soils, lake sediments, and groundwater are long-term contaminants of watersheds that reflect the legacy of past human activities (Reed-Anderson, Carpenter, and Lathrop 2013; Hamilton 2012). This legacy P makes watershed restoration more challenging because lag times will occur between changing rates of using P in watersheds and lowering of stream and lake P concentrations; and these lag times are difficult to predict (Jarvie et al. 2013; Sharpley et al. 2013). Jarvie et al. (2013) estimate that only 20 to 30% of mined P reaches the food supply, while 70 to 80% remains in landscapes including soils, groundwater, inland freshwater bodies, and estuaries. These legacies could provide enough P that crop growth would be sustained for several years without adding any new P (Sattari et al. 2012). Both lags and uncertainly in recovery rates will affect commitment of decision makers to restoration. However, the threats of long-term contamination and uncertain restoration should encourage decision makers to curtail further contamination of watershed soils as well as river and lake sediments. Second, and partly as a function of high levels of legacy P in many agricultural soils, attention is focusing on increased levels of DRP in water samples from agricultural drainage tile lines that empty into streams (Kleinman et al. 2015b).

But if evidence of DRP coming out of tile lines presents a new problem, advances in engineering research suggest a new category of solution. Technologies for removing DRP from water have been developed (Safferman, Henderson, and Helferich 2007) and are being commercialized. The advent of P mitigation technology opens the door for policy research beyond source abatement by farmers. Indeed, the durability of legacy P suggests that mitigation may be important if abatement of new P applications will be slow to take effect on algal blooms and other impacts of P pollution.

Recent evidence suggests that many algal responses to P increases are non-linear and are manifested in threshold changes in algal biomass along P gradients (Downing, Watson, and McCauley 2001). These threshold responses of ecosystem services to pollution can be valuable for developing stakeholder consensus for pollution management targets (Muradian 2001). Many relationships between elements in the coupled human and natural system are non-linear relationships. For example, ecological responses to nutrient pollution are non-linear (Stevenson et al. 2008), and so are relationships between public perceptions of recreational value and algal biomass. When changes in ecosystems are sufficiently great, they cross thresholds that trigger responses that propagate through the coupled human and natural systems, changing the state of the systems. The coastal toxic algal blooms of the mid-1990s likely played a key role in stimulating the Clean Water Action Plan. Of course, other factors can dampen and disconnect couplings in coupled human and natural systems, but these non-linearities can produce sweet spots for management, and theoretically take the forms of alternative stable states (Carpenter et al. 1998; Stevenson et al. 2008).

The predicted effects of climate change on P movement represents the last new piece of biological information (Paerl and Huisman 2008). Warming temperatures triggered by climate change will favor the growth of cyanobacteria, the active ingredient in most freshwater harmful algal blooms (Michalak et al. 2013). More intense and frequent storms will accelerate erosion of surface and stream bank soils, flushing DRP into streams directly or via drainage tile. The imminent threat that climate change presents will exacerbate the P problem and points to the urgency of solving the P puzzle.

New Gaps for Economics Research to Solve the P Puzzle

As P pollution returns to the policy limelight after a 30-year hiatus, the pieces of the P puzzle that have been quietly assembled through economic, biological, and engineering research enable us to discern new gaps for economic research contributions ranging from the mundane to the cutting edge. Filling these gaps will call for multi-dimensional research, including economic research efforts that range across benefit-cost analysis of P mitigation technology, redesign of voluntary payment programs (including social and behavioral nudges), private supply-chain standards, reallocation of property rights, and rigorous impact analysis of extant conservation programs.

Mitigation opportunities

New engineering technologies for mitigating the impact of P already in the environment will require benefit-cost analysis. Two such technologies stand out: controlled drainage structures that delay the release of DRP-laden drainage water into streams and iron-particle technologies that remove DRP from water. Drainage control structures are already commercial and are marketed for their ability to retain soil moisture against the risk of subsequent drought. But such technologies have been shown to prevent pulses of nitrogen that can trigger hypoxia in marine waters, and they may have similar potential for P movement into fresh waters. Benefit-cost analysis would require partnership with aquatic ecologists and engineers to evaluate the probability of abating harmful algal blooms, while economic research explores both private costs (such as increased risk to farmers of waterlogging that can make fields inaccessible or curtail crop yields) and benefits as well as the valuation of public benefits. At a broader level, *ex ante* benefit-cost analysis is merited for targeted projects to restore wetlands for biological filtering of P in highly sensitive settings where tile drainage has short-cut natural systems for P removal (Blann, Anderson, and Sands 2009).

Emerging technologies for P removal from water (e.g., Safferman, Henderson, and Helferich 2007; Kleinman et al. 2015b) have the potential for direct reduction of the P stock in the environment. Early versions target manure treatment from concentrated animal feeding operations, a growing private market in the one area of agricultural water pollution covered by the Clean Water Act. Manure slurries contain P at much higher concentrations than water exiting drainage tile lines, but the technical feasibility of P removal from tile lines deserves evaluation along with a breakeven price analysis for DRP effluent to establish a baseline for the marginal cost of mitigating the algal bloom problem via removal of P at identifiable collection points where tile lines or private drainage ditches open into public streams.

Valuation of water quality tied to P

Advances in the valuation of ecosystem services and continued refinement of valuation methods provide the economic tools needed to generate the valuation information that fills pieces of the P puzzle and potentially drive policy forward. Moreover, the growth and continued refinement of ecosystem service modelling and ecological production functions will open additional opportunities to connect P from agriculture to public demand and values for water quality. The critical opportunity here lies in empirical applications that produce results directly relevant to policy by adequately connecting values to P.

Redesign of voluntary payment programs

Recent economic research that highlights informational and transaction costs associated with real-world implementation of conservation auctions presents policy mechanism design opportunities. The challenge is clear: transaction costs that deter participation in conservation auctions can seriously undermine their cost-effectiveness (DePiper 2015; Peterson et al. 2015; Palm-Forster et al. 2016). But innovations in linking GIS databases on terrain and soil type to weather simulators and biophysical models of P fate and transport enable *ex ante* classification of agricultural lands by vulnerability to P loss—and associated expected benefits from agricultural P abatement practices. Such information can inform the design of targeted payment-for-practice offers (Palm-Forster et al. 2016). A rich agenda in experimental and constrained optimization modeling methods can be developed that explicitly accounts for transaction costs in designing and testing voluntary policy approaches that aim to optimize the cost-effectiveness of P abatement when nonparticipation is an explicit option.

The continuing improvement of biophysical simulation models and opportunities to link them to spatial databases creates new opportunities to design environmental policy around simulated outcomes. Bioeconomic models have been developed that couple economic objective functions to biophysical simulation models to analyze P management decisions and outcomes (Vatn et al. 1997; Westra 2001). Future advances in such models will need to incorporate new knowledge about legacy P and DRP, meaning they will need to be explicitly dynamic—a feature that has not been needed in nitrogen-oriented agricultural nutrient models. Further development of such models in ways that build in market feedbacks (Egbendewe-Mondzozo et al. 2015) and that wed heterogeneous human preferences to varied landscapes could enable better targeted, outcome-based environmental policy design.

Social norms and behavioral nudges

Voluntary conservation programs need not rely solely on monetary payments to attract participants. Social norms and insights from behavioral economics are of increasing interest as possible policy tools in areas such as health, nutrition and resource conservation. For agricultural practices, understanding of the role of social norms and potential for using insights from behavioral economics are an evolving part of the P puzzle. Social norms in the form of normative expectations and approval of one's behavior by others who are considered important have been linked to agricultural conservation behavior, usually using self-reported data on the importance of these factors and often performed outside the U.S. (Willy and Holm-Müller 2013; Beedell and Rehman 1999; Mzoughi 2011; Fielding et al. 2005). However, a recent study of filter strip adoption in a Great Lakes watershed, Yeboah, Lupi, and Kaplowitz (2015) did not find that normative expectations of others influenced adoption. Similarly, a recent study suggests normative expectations may not be as important a factor in U.S. farmers' decision making compared to Swiss farmers (Celio et al. 2014). Social norms can also take the form of peer effects, that is, the effect that peers and neighbors' behavior can have on one's own behavior. Though peer effects are recognized as difficult to identify (Manski 2000), they have been shown to influence conservation adoption (Chen et al. 2009). A recent study finds that peer effects in the Maumee watershed, which affects P loads in Lake Erie, are significant in explaining farmer use of no till (Konar, Roe, and Irwin 2014). Nevertheless, a causal understanding of the role of social norms regarding P use remains a missing piece of the puzzle where economists have much to offer.

Public disclosure of information about environmental behavior has prompted more responsible stewardship in the corporate world (Anton, Deltas, and Khanna 2004; Khanna and Anton 2002). In particular, information disclosure about toxic releases (Dasgupta,Wheeler, and Wang 2007; Khanna and Anton 2002) and violations of safe drinking water standards (Bennear and Olmstead 2008) have led to abatement of such behavior. Disclosure policies have seen little application to nonpoint source pollution. However, the advent of social networking programs, mobile devices, and apps to run environmental simulation models could enable research in this arena that was not previously practical.

The growing field of behavioral economics has led to interest in the potential for soft policy tools such as behavioral nudges and socially normative messaging to influence conservation (Akerlof and Kennedy 2013). In part due to the body of evidence that people make decisions based on their beliefs about social norms (Cialdini and Goldstein 2004), the provision of social comparison messages has been shown to have effects on pro-environmental behavior by consumers (Ferraro and Price 2013; Goldstein, Cialdini, and Griskevicius 2008) and has been widely studied for energy use. Though much less is known about the effects social messaging and other behavioral economic insights can have on farmers, these approaches have potential as a policy tools. Recently farmers with expiring CRP contracts were assigned to experimental informational treatments, or information nudges, which included information on other farmer's interest in CRP (Wallander, Higgins, and Ferraro 2014). Although the nudges only resulted in slight increases in re-enrollment of 1.7%, the treatments were very inexpensive and suggest further research is warranted. Similarly, social priming may alter the cost-effectiveness of conservation program. In a field experiment of nutrient management auctions, Messer, Ferraro, and Allen (2015) find that switching the default cost share starting bid from 0% to full cost share led to a 17% increase in participant's cost sharing. The USDA has recognized the importance of this area of behavioral research though its internal research and through the Center for Behavioral and Experimental Agri-Environmental Research (http://centerbear.org/). Much remains to be learned about how social norms and behavioral nudges can encourage conservation behavior, ranging from increased participation to reductions in minimum acceptable payments.

Supply chain standard

Increasing consumer interest in where their food comes from and how it is made, coupled with growing market power of large retailers, has the potential exert pressure on farmer production practices. The development of supply chain standards by large retailers such as Wal-Mart and McDonald's has become an important driver of production practices, including food safety certification and pollution reduction (Reardon et al. 2000). In the past couple of decades, private certification and supply chain standards have become prerequisites for gaining access to specific markets in many cases (Waldman and Kerr 2014). In response to pressure from nongovernmental organizations, many food manufacturers and retailers have begun steps toward extending supply chain standards into the realm of agricultural environmental stewardship. Expanding certification initiatives, such as Field to Market and the 4 R's program described earlier, create fertile ground for research collaborations between business market researchers and environmental economists not only to expand what is known about consumer willingness to pay for stewardship labels (Waldman and Kerr 2014), but also to explore the industry structure conditions under which firms will cooperate (or compete) in establishing industry standards for environmental stewardship in supply chains.

Reassigning property rights

There is growing evidence that the "pay-the-polluter" paradigm that underpins voluntary agricultural conservation programs has not been effective at improving water quality in many of our nation's waterways (Kling 2011; Shortle et al. 2012; Segerson 2013; Shortle and Horan 2013; Sohngen et al. 2015). Even though changes in the design of current conservation programs could lead to cost-effectiveness improvement, whether substantial water quality gains could be achieved under voluntary approaches remains questionable (Kling 2011; Shortle et al. 2012; Segerson 2013; Shortle and Horan 2013). As noted above, the voluntary nature of current laws has been implicated in the failure of pilot markets for tradable water pollution permits involving farmers. Segerson (2013) suggests that voluntary approaches could be effective if associated with a credible threat of regulation in case the environmental goal is not met. However, in practice, no such threat typically exists. Kling (2011) proposes a paradigm switch to the "polluter-pay-principle," via an Abatement Action Permit System (AAPS) where each producer would be required to hold abatement permits that would be obtained either by contributing to abatement efforts or by purchasing permits on the market. While a brusque, across-the-board reversal of the property right to pollute is unlikely in foreseeable future, Shortle et al. (2012) propose to start by applying this concept in impaired watersheds with a TMDL in place. Rather than capping only point sources, nonpoint sources would also be capped to meet the TMDL requirements. In impaired watersheds, permit allowances for nonpoint sources might be stricter than historical loadings, thereby increasing the incentives for trading.

Program evaluations

The program evaluation literature has been booming in recent years (e.g., Imbens and Wooldridge 2009; Angrist and Krueger 1999), with applications in labor, education, health and the environment, among others. However, there are relatively few such studies evaluating the impact of agricultural policies and none that we are aware of that seek to directly assess the impact of such policies on P loadings. Most of the existing studies seek to assess the impact of regulations or policies on land use or the adoption of individual management practices. For example, Liu and Lynch (2011) evaluate the impact of land-use policies on the loss of farmland, employing matching techniques. González-Ramírez and Arbuckle (2015) using propensity score matching to study the impact of cost-share programs on the use of cover crops in Iowa. Cooper (2005) examines the role of incentive payments on the adoption of BMP.[2] In many of these papers, the focus is on the additionality of the farmland program (e.g., Mezzatesta, Newburn, and Woodward 2013). These are valuable exercises in their own right. What is missing, however, is an effort tying these impacts to the changes in P loadings and ultimate changes in ecosystem services that consumers value.

The renewed prominence of harmful algal blooms has refocused public attention on the P pollution puzzle after a hiatus of some 30 years. Over that period, new knowledge about P biology and mitigation engineering in parallel with important developments in policy mechanism design and nonmarket valuation have brought new pieces to the table for tackling the puzzle. Our understanding of social-ecological systems has also improved, enabling better identification of potential points and approaches for policy interventions. The research directions recommended above will tap skills of many stripes of economist in partnership with biologists, engineers and others. New puzzle pieces are needed to fill in areas with benefit-cost analysis, bioeconomic modeling, social and behavioral economics, supply chain standards, institutional economics, and program evaluation. The puzzle is daunting, but solution pieces are at hand and the task is urgent.

[2] Other related program evaluation studies include Jacobson (2015) and Chabet-Ferret and Subervie (2013) evaluating individual ag land programs and research evaluating other conservation and PES programs (Andam et al. 2008, Andam et al. 2010; Baylis et al 2015)

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