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Valuing economic impact reductions of nutrient pollution from livestock waste

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Abstract

Nutrient pollution from livestock waste impacts both fresh and marine coastal waters. Harmful algae blooms (HABs) are a common ecosystem-level response to such pollution that is detrimental to both aquatic life and human health and that generates economic losses (e.g., property values and lost tourism). Waste treatment and management technologies are not well established practices due, in part, to the difficulty to attribute economic value to associated social and environmental impacts of nutrient pollution. In this work, we propose a computational framework to quantify the economic impacts of HABs. We demonstrate the advantage of quantifying these impacts through a case study on livestock waste management in the Upper Yahara watershed region (in the state of Wisconsin, USA). Our analysis reveals that every excess kilogram of phosphorus runoff from livestock waste results in total economic losses of 74.5 USD. Furthermore, we use a coordinated market analysis to demonstrate that this economic impact provides a strong enough incentive to activate a nutrient management and valorization market that can help balance phosphorus within the study area. The proposed framework can help state, tribes, and federal regulatory agencies develop regulatory and non-regulatory policies to mitigate the impacts of nutrient pollution.

Keywords

Livestock waste; Economics; Eutrophication; Phosphorus

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Apoorva M. Sampat: Methodology, Formal Analysis, Software, Writing - Original Draft, VisualizationAndrea Hicks: Writing Review & Editing.**Gerardo J. Ruiz-Mercado:** Supervision, Writing Review & Editing, Funding acquisition. **Victor M. Zavala:** Conceptualization, Supervision, Writing Review & Editing, Funding acquisition.

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1. Introduction

Agricultural non-point nutrient pollution is the leading cause of water quality impairments to rivers, the second largest cause for wetlands, the third largest for lakes, and is a major contributor to the contamination of groundwater (U.S. Environmental Protection Agency, 2019). When excess nutrients (in the form of chemical fertilizers or manure) are applied to croplands having legacy phosphorus in soils and there is either rain or snowmelt following the application, the nutrients runoff to the waterbodies resulting in ecosystem responses such as excess growth of algae. The rapid growth of algae is known as harmful algae blooms (HABs) and the toxins released during HABs can be detrimental to both aquatic life and human health. HAB events can cause massive fish kills (Woods Hole Oceanographic Institution, 2020), closure of beaches (CNN News, 2019) and shellfish beds (CNN News, 2019), death of marine mammals and sea birds (Sea-Bird Scientific, 2020), coral reefs (Bauman et al., 2010), and alter marine habitats (Stumpf and Tomlinson, 2005). This in turn hurts tourism, recreational and commercial fishing, and valued habitats, that are vital to local economies (National Oceanic and Atmospheric Administration NOOA, 2019). In July 2019, all 21 beaches in the State of Mississippi were closed due to HABs (CNN News, 2019). Dodds et al. (2009) estimate an average annual loss of 770 million USD in recreational activities due to eutrophication of U.S. freshwaters. In the State of Florida, HABs have resulted in a monthly loss of 2.8 and 3.7 million USD corresponding to restaurant and lodging revenue, respectively (Larkin and Adams, 2007). Frequent occurrence of HABs also lowers property values of lakefront properties. The loss in property values are the largest impact bearers of eutrophication with an estimated average economic loss of 1.6 billion USD annually (Dodds et al., 2009).

Estimating the scale of economic loses associated to HABs provides valuable information to determine appropriate counter measures to prevent or mitigate the loses (Hoagland and Scatasta, 2006). Such an estimate can also influence key policy decisions such as nutrient credit trading (Fassbender, 1994; King and Kuch, 2003), incentives (McIsaac, 2012; Stubbs, 2010), and regulations (Burlakova et al., 2018; Levinson, 2005). Unfortunately, not many studies have been conducted to quantify the economic impacts of HABs. Most of the reported studies are driven by impacts of toxins in commercial fisheries (Jin et al., 2008; Pretty et al., 2003; Sanseverino, Conduto, Pozzoli, Dobricic, Lettieri). Hoagland et al. (2002) first estimated an annual expenditure of 20 million USD in public health due to seafood poisoning caused by HABs in the United States. HABs can also cause respiratory illness such as asthma, pneumonia, and bronchitis. In the gulf coast of Florida, wind causes toxins released by HABs to form aerosols and causes damage to the respiratory system (Fleming et al., 2005). For the Saratosa County in the State of Florida alone, the cost of respiratory illnesses associated with HABs is estimated to be 0.5 to 4 million USD annually (Hoagland et al., 2009). Phaneuf et al. (2013) developed a tool that estimates the monetary impact on recreational use of freshwater lakes in the southeast for a change in water quality. As an input, the tool requires the current and desired concentrations of phosphorus, nitrogen, and chlorophyll a. It also requires an estimate on the number of trips to the lake. It then outputs the total economic impact of improving the water quality from a recreational use perspective. As it will become evident later, our work can provide an extension to this tool

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by providing a methodology to estimate the impact on the number of trips to the lake as a function of water clarity, estimating the impact on lakefront property values, and quantifying the clean up expenses. Quantifying the impacts of nutrient pollution can also drive the decision-making for recovery processes. Sena et al. (2020) observe that recovering struvite from a waste water treatment plant in Madison, WI is economically viable if we consider the avoided cost of nutrient pollution.

A number of the economic analyses available in literature rely on survey data for estimating the economic impact of algae blooms (Dodds et al., 2009; Fleming et al., 2005). Dodds et al. (2009) use data on total phosphorus and nitrogen concentrations in different ecoregions to estimate economic damages of eutrophication in U.S. freshwaters. Fleming et al. (2005) estimate health impacts of red tides in the Saratosa county in Florida through a statistical model that correlates the HABs outbreak with the cost of emergency visits to the Saratosa Memorial Hospital for respiratory illnesses. For the impacts of nutrient pollution, extensive research has been done about the amenities and disamenities due to nutrient pollution in the Great Lakes region (Mendelsohn and Olmstead, 2009; Murray et al., 2001; Simons and Saginor, 2006; Stephens and Partridge, 2015a; 2015b). Such methodologies are difficult to scale to different geographical areas facing similar HABs related issues. A model based on easily measurable quantities (e.g., water clarity) can help extend the model to different geographical locations and provide a preliminary estimate towards quantifying the economic impacts of HABs. Vesterinen et al. (2010) propose a hurdle model to quantify the change in demand for recreational activities as a function of water clarity. The hurdle model is proposed for Finland, but it can be useful in estimating the economic impacts in other locations by modifying the parameter values specific to the study area. Dodds et al. (2009) propose a linear relationship between water clarity and the loss in property value. The advantage of such methodologies is that water clarity can be easily linked to the total phosphorus (TP) concentration in the waterbody (Lillie et al., 1993) (thus providing a direct way to quantify the economic impacts of HABs).

The contributions of this work are two-fold. First, we provide a computational framework to integrate models that estimate the socio-economic impact of HABs with a coordinated market framework (Sampat et al., 2019). This integration of models reveals the inherent value of excess phosphorus and connects it to the socio-economic impacts realized due to HABs. Second, we demonstrate the importance of estimating these economic impacts through a case study for the coordinated management of livestock waste in the upper Yahara watershed region. Our analysis reveals that every excess kilogram of phosphorus runoff from livestock waste in the upper Yahara watershed region results in total economic losses of 74.5 USD. As it will become clear later (in Section 3), this economic impact can provide a driving force for waste processing, help in balancing P in the region, and achieve nutrient pollution reduction targets in an environmentally and economically sustainable manner.

2. Economic impacts of algae blooms

The U.S. Environmental Protection Agency (U.S. Environmental Protection Agency, 2015) identifies seven major economic categories that are impacted by the eutrophication of waterbodies. Amongst these categories, the largest economic losses in U.S. freshwater are

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attributed to property values and recreational use (Dodds et al., 2009). For the scope of this work, we quantify the impacts associated with property values, recreational costs, and cleanup expenses. Impacts on commercial fishing and human health are location specific and are difficult to generalize through mathematical modeling. Also, for our study area (Upper Yahara watershed region), the impacts in these categories are negligible (Wisconsin Department of Natural Resources, 2012).

2.1. Property values

The value of lakefront properties depends strongly on water clarity. Dodds et al. (2009) estimate that the property value decreases by 15.6% for every meter decrease in water clarity (measured by Secchi depth). The Secchi depth (SD) is a metric used for water clarity that is calculated by inserting a black and white colored disc in the water and by measuring the maximum depth until which the disc is visible. Algae blooms have a direct impact on water clarity; specifically, a high total phosphorus (TP) concentration during an algae bloom turns the water turbid, reducing the Secchi depth. The relationship between the Secchi depth and total phosphorus concentration is given by Lillie et al. (1993):

$$\ln(\mathrm{SD}) = a + b\ln(\mathrm{TP}) \tag{1}$$

where SD is the Secchi depth in meters and TP is the total phosphorus concentration in $\mu g/L$. Parameters a and b depend on the lake type. For stratified natural lakes (e.g. Lake Mendota, WI), $a = 2.10$ and $b = -0.44$ (Lillie et al., 1993).

2.2. Recreational costs

A decrease in water clarity also reduces the demand for recreation activities such as swimming and fishing (Wisconsin Department of Natural Resources, 2012). Vesterinen et al. (2010) propose a hurdle model to quantify the change in demand for recreational activities as a function of water quality. This hurdle model is proposed for the waterbodies in Finland. Currently, no other relevant studies exist that quantify the impact of water quality on recreational activities. Also, we can apply this model for the State of Wisconsin based on the observation that both Finland and Wisconsin have similar population sizes and similar median household income, and both face problems of eutrophication of water bodies (Wisconsin Department of Natural Resources, 2012). In fact, this model is used by the Wisconsin Department of Natural Resources.

The hurdle model is a two stage model: a logit (or logistic regression) model to estimate the probability of participation in a recreational activity, and a negative binomial model to estimate the frequency of participation. The logit model has the general form:

$$y = \ln(O) = \ln\left(\frac{p}{1-p}\right) = \beta_0 + \beta^T X \tag{2}$$

here, p is the probability of participation and y is the logarithm of the odds $O = \left(\frac{p}{1-p}\right)$. $\beta_0 \in \mathbb{R}$ and $\beta \in \mathbb{R}^n$ are logit coefficients. $X \in \mathbb{R}^n$ is a vector of n characteristics (e.g. water clarity, number of summer days, etc.) that affect the odds of participation in the recreational

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activity. The logit coefficients for different recreational activities (β_1^A) reported by Vesterinen et al. (2010) with respect to change in water clarity are listed in Table 1. Here A represents the activity from the set {swimming, fishing}. Vesterinen et al. (2010) estimate that a decrease in Secchi depth does not have a significant effect on the odds of participation in swimming ($\beta_1^S = -0.006$), but the frequency of participation (days spent swimming) decreases $\left(\gamma_1^S = 0.059\right)$. For boating, they find water clarify has no direct effect either in probability or frequency of participation. For fishing, both the probability and frequency of participation decreases with a reduction in Secchi depth.

As per the logit model, the odds of participation in a recreational activity $A \in$ {swimming, fishing} are:

$$O^A = \exp\left(y^A\right) \tag{3}$$

$$= \exp\left(\beta_0^A + \left(\beta^A\right)^T X\right) \tag{4}$$

For a change in the Secchi depth, the odds ratio (OR^A) of an activity is given by:

$$\mathrm{OR}^A = \frac{O_2^A}{O_1^A} = \exp\left(\beta_1^A \Delta X_1\right) \tag{5}$$

where ΔX_1 is the change in Secchi depth (in meters) and β_1^A is the corresponding logit coefficient. O_1^A and O_2^A are the odds of participation in an activity (A) before and after the Secchi depth decreases respectively.

Next, we quantify the change in frequency of participation in recreational activities using the negative binomial model and the associated coefficients (Table 1) reported in Vesterinen et al. (2010). The ratio of the frequency of participation in an activity A is given by the "Incidence rate ratio" (IRR^A):

$$\mathrm{IRR}^A = \frac{\mu_2^A}{\mu_1^A} = \exp\left(\gamma_1^A \Delta X_1\right) \tag{6}$$

here μ_1^A and μ_2^A are the rates (or frequencies) of participation and γ_1^A is the negative binomial coefficient for an activity with respect to change in Secchi depth.

Once the impacts on the probability and frequency of participation are calculated, we estimate the annual loss in recreation trips (Ω^A) for an activity A, given population size N:

$$\Omega^A = N \times \left(O_1^A - O_2^A\right) \times \left(\mu_1^A - \mu_2^A\right) \tag{7}$$

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Using this information, we estimate the loss in recreational activities by using the cost per trip data (δ^A):

$$\text{Loss in Revenue} = \sum_{a \in \mathscr{A}} \Omega^a \times \delta^a \tag{8}$$

In case of fishing and swimming, Kaval and Loomis (2003) estimate the value of δ^A to be on average 63.27 USD/trip and 57.27 USD/trip respectively (converted to 2018 USD).

2.3. Cleanup expenses

In cases when excess nutrients are already introduced in the waterbodies, mitigation and restoration technologies are required to prevent the manifestation of nutrient problems and algae blooms. Common treatment technologies include aeration systems, alum treatment, bio-manipulation, dredging, herbicide treatment, and hypolimnetic treatment. More details on these technologies and corresponding cost estimates can be found in U.S. Environmental Protection Agency (2015). In areas where the affected waterbody is a source for drinking water, clean up procedures such as alum treatment are required to make the water potable. The alum treatment costs are based on acres of the water surface treated. Wisconsin Department of Natural Resources (Wisconsin Department of Natural Resources, 2012) reports the alum treatment cost to range between 344 and 861 USD/acres plus a fixed cost of 25,000 USD. We note that in some instances the clean up expenses may be higher than it would be worth to the affected community to live with the degraded waterbody. Our methodology to estimate the economic impacts of nutrient runoff provides a systematic way to compare these two expenses.

2.4. Human and pet health

Direct contact with waterbodies that are affected by HABs, either by swimming or other recreational activities, can cause illness in humans and animals. Common symptoms include dermal rashes, respiratory irritation, gastrointestinal distress, and cold/flu-like illness symptoms. Many of the health related costs of HABs are realized in the U.S. coastal states. Hoagland et al. (2009) estimate the cost of respiratory illnesses associated with the red tides in Saratosa County, Florida to range from 0.02 to 0.13 million USD annually. The authors use a statistical exposure-response model that is based on number of hospital emergency department visits for respiratory illness and the occurrence of algae blooms. For our case study, the HABs related health care costs in the State of Wisconsin are minor (Wisconsin Department of Natural Resources, 2012). Thus, the associated costs are not included in our analysis.

2.5. Waste processing

One strategy to prevent phosphorus runoff (and HABs in turn) is by processing livestock waste and recovering phosphorus. We consider three technology variations in our case study (Fig. 1). These technologies capture the different levels of complexity (ranging from simple mechanical separation to the more advanced chemical treatment) commonly employed for waste processing. The first pathway uses a screw press to separate the livestock waste into

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solid and liquid fractions. The solid fraction can be used as a crop fertilizer (Aguirre-Villegas et al., 2019). The second pathway further processes the solid fraction through granulation technology to recover P in the pellet form (Sharara et al., 2018a). The third pathway processes the liquid fraction and recovers P through struvite formation. The economic viability of these waste processing pathways depends strongly on the composition of waste streams (which is highly variable), economies of scale, and transportation costs. Also, the logistical issues associated with waste collection and transportation hinder the large scale deployment of waste treatment technologies. Diverse government regulations and incentives to promote waste treatment have not been able to overcome these techno-economic and logistical issues. As a result, the infrastructure for waste management is at present fragmented and limited, posing a significant obstacle to mitigate the pollution of water, land, and air resources. This obstacle also hinders the sustainable growth of urban, agricultural, and food sectors.

In this complex decision decision-making environment where a large number of stakeholders rely on shared and constrained infrastructures, a coordinated management framework (Sampat et al., 2019) can enable efficient exchange of products. This framework can capture complex spatio-temporal dependencies and externalities (e.g., weather). In this system, the suppliers and consumers provide bids for waste and derived products. The technology and transportation providers also submit bids for their services. An independent system operator (ISO) uses this bid information and runs a dispatch system that finds optimal physico-chemical transformation and transportation pathways that balance demands and supplies in a given geographical region (Fig. 2). This approach also ensures that system-wide transportation and transformation capacities are met by the dispatch solution. The management system functions as a *coordinated market* that generates prices for each waste type and derived product at every location in the study area. This framework helps us determine how economic impacts of HABs can incentivize waste transportation and processing technologies. We provide a brief overview of this coordination markets model in the next section. More details about this framework and the economic properties satisfied by the cleared prices can be found in Sampat et al. (2019).

2.6. Coordinated market model

We consider a system comprising of geographical locations/nodes $\mathcal{N}$ (e.g. dairy farms, croplands, external companies), products $\mathcal{P}$ (e.g. waste, pellets, struvite), suppliers $\mathcal{S}$ (e.g. dairy farms), consumers $\mathcal{D}$ (e.g. croplands, external companies), transportation providers $\mathcal{L}$ (e.g. hauling and shipping companies), and transformation (technology) providers $\mathcal{T}$ (e.g. mechanical separation, granulation, struvite recovery). The market players and the associated set notations are summarized in Fig. 3. Each supplier $i \in \mathcal{S}$ has an associated supply flow $s_i \in \mathbb{R}_+$, location $n(i) \in \mathcal{N}$, product type $p(i) \in \mathcal{P}$, maximum offered capacity $\bar{s}_i \in \mathbb{R}_+$, and bidding cost $\alpha_i^s \in \mathbb{R}_+$. Each consumer $j \in \mathcal{D}$ has an associated demand flow $d_j \in \mathbb{R}_+$, location $n(j) \in \mathcal{N}$, product type $p(j) \in \mathcal{P}$, maximum requested capacity $\bar{d}_j \in \mathbb{R}_+$, and bidding cost $\alpha_j^d \in \mathbb{R}_+$.

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Each transportation provider $\ell \in \mathcal{L}$ has an associated flow $f_\ell \in \mathbb{R}_+$, sending node $n_s(\ell) \in \mathcal{N}$, receiving node $n_r(\ell) \in \mathcal{N}$, product transported $p(\ell) \in \mathcal{P}$, maximum capacity $\bar{f}_\ell \in \mathbb{R}_+$, and bidding cost $\alpha_\ell^f \in \mathbb{R}_+$. The bidding cost is the cost of moving a unit of flow from the source to the destination node. The set of all flows entering node $n \in \mathcal{N}$ is $\mathcal{L}_n^{in} := \left\{\ell \mid n_r(\ell) = n\right\}$ and the set of all flows leaving node $n \in \mathcal{N}$ is $\mathcal{L}_n^{out} := \{\ell \mid n_s(\ell) = n\}$.

Each transformation provider $t \in \mathcal{T}$ has corresponding transformation/yield factors $\gamma_{t,p} \in \mathbb{R}$, location $n(t) \in \mathcal{N}$, a reference product $p(t) \in \mathcal{P}$, processing capacity $\bar{\xi}_t \in \mathbb{R}_+$, and processing cost $\alpha_t^\xi \in \mathbb{R}_+$. Transformation factors capture the units of product p consumed/generated per unit of reference product $p(t)$ consumed/generated by the technology unit. We follow the convention that $\gamma_{t,p} > 0$ if product p is generated, $\gamma_{t,p} < 0$ if product p is consumed, and $\gamma_{t,p} = 0$ if product p is neither produced nor consumed by the technology t. We note that $\gamma_{t,p(t)} = -1$ represents that one unit of reference product is consumed to produce/consume other products. For each technology, $\xi_t \in \mathbb{R}_+$ represents the extent of transformation, which is the total amount of $p(t)$ processed.

The Environment as a Stakeholder: To capture the economic impact of HABs resulting from excess P in the region (denoted by θ_j), we define the environment as one of the stakeholders (represented by set $\mathcal{D}' \subset \mathcal{D}$). The idea being that, if there is excess P in the region, it can be released to the environment but at a cost (λ). This cost can be seen as a tipping cost or a *value of service* (VOS) that the environment charges society. The VOS captures the economic impacts of nutrient pollution (and HABs), which include both external costs borne by local economies and communities impacted by environmental and human effects. As it will become clear later (in Section 3.2), this VOS value acts as an incentive that exerts sufficient socio-economic pressure to activate a waste management market.

The ISO uses bidding information (α^d, α^s, α^f, α^ξ) and capacity limits ($\bar{d}, \bar{s}, \bar{f}, \bar{\xi}$) to solve the clearing problem (9). The model outputs are allocations (d, s, f, ξ) that maximize the social welfare (9a), satisfy the physical conservation laws (9b), and capacity limits (9e)–(9h). Maximizing the social welfare function ensures that the demand served is maximized and the costs of supply, transformation and transportation are minimized. The conservation laws/balancing constraints serve as market clearing constraints that balance demand and supply at every location. The first term in parenthesis in the balancing constraint (9b) is the total input flow of product p into node n (consisting of supply and transportation flows entering the node). The second term in parenthesis is the total output flow of product p from node n (consisting of the demand and transportation flows leaving the node). The third term captures the generation/consumption of product p in all technologies located at node n.

$$\max_{(s, d, f, \xi)} \sum_{j \in \mathcal{D}} \alpha_j^d d_j - \sum_{i \in S} \alpha_i^s s_i - \sum_{\ell \in \mathcal{L}} \alpha_i^f f_\ell - \sum_{t \in T} \alpha_i^\xi \xi_t - \lambda \sum_{j \in \mathcal{D}', p(j) = P} \theta_j \tag{9a}$$

$$\text{s.t.} \left(\sum_{i \in S_{n,p}} s_i + \sum_{\ell \in \mathcal{L}_{n,p}^{in}} f_\ell \right) - \left(\sum_{j \in \mathcal{D}_{n,p}} d_j + \sum_{\ell \in \mathcal{L}_{n,p}^{out}} f_\ell \right) + \sum_{t \in T_n} \gamma_{t,p} \xi_t = 0, (n,p) \in \mathcal{N} \times \mathcal{P}, (\pi_{n,p}) \tag{9b}$$

$$\theta_j \geq d_j - \bar{d}_j,\ j \in \mathcal{D}', p(j) = \boldsymbol{P} \tag{9c}$$

$$\theta_j \geq 0,\ j \in \mathcal{D}', p(j) = \boldsymbol{P} \tag{9d}$$

$$0 \leq d_j \leq \bar{d}_j,\ j \in \mathcal{D} \tag{9e}$$

$$0 \leq s_i \leq \bar{s}_i,\ i \in \mathcal{S} \tag{9f}$$

$$0 \leq f_\ell \leq \bar{f}_\ell,\ \ell \in \mathcal{L} \tag{9g}$$

$$0 \leq \xi_t \leq \bar{\xi}_t,\ t \in \mathcal{T}\ . \tag{9h}$$

We define $\mathcal{C}$ as the set of all possible/feasible allocations (d, s, f, ξ) that satisfy the capacity constraints (9e)–(9h). The dual variables $\boldsymbol{\pi}_{n,p}$ of the conservation laws (9b) set values for products at different geographical locations and act as the *market clearing prices*. Because of this, $\boldsymbol{\pi}_{n,p}$ are also referred as the *locational marginal prices* or *nodal prices*. We use the short-hand notation $\boldsymbol{\pi}$ to denote all dual variables. Prices and allocations derived from the clearing formulation establish fundamental economic properties of the market. These prices and allocations remunerate providers (e.g., dairy farmers) and charge consumers (e.g., croplands and product buyers). Moreover, the prices are also known as *coordination prices* as they can be used as incentives to promote coordination between stakeholders. The allocations and prices generated from the clearing formulation also represent a competitive economic equilibrium that maximizes the collective profit of the market players (Sampat et al., 2019). We note that the coordinated market framework is a dynamic market, which is cleared at fixed temporal frequencies (e.g. every day, month, or year). This allows the stakeholders to update their price bids at regular time intervals as the system evolves. This dynamic property is imperative when the market interacts with the wider economy which causes a change in the demand, supply, and product prices. The dynamic nature in fact drives the existence of coordinated wholesale electricity markets (Bohn et al., 1984; Hogan et al., 1996; Pritchard et al., 2010; Zavala et al., 2017) where prices of electricity is determined in a rapidly evolving market of natural gas, electricity demand, and supply of renewable sources such as wind and solar.

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3. Case study

We consider the upper Yahara watershed region in the State of Wisconsin (Fig. 4) to reduce the occurrence of harmful algae blooms in Lake Mendota. Heavy use of livestock manure and agricultural fertilizers have resulted in excess amounts of phosphorus being accumulated in this area. The accumulated phosphorus (P) is often washed into waterways, due to rain and snow melt. This runoff leads to the blue-green algae blooms in Lake Mendota (University of Wisconsin-Center for Limnology, 2018). In this work, we quantify the economic impacts associated with algae blooms in Lake Mendota.

The study area consists of 203 farms (148 dairy farms and 55 beef farms). These farms account for more than 99% of the P generation associated with livestock waste. Here, we consider that all the farms within the study area spread the waste on the associated croplands. This corresponds to spreading of 1.34 million tons of waste annually, resulting in a P release rate of 917.83 tons/yr. We note that the agricultural non-point sources (such as livestock waste) contribute to 43% of total P loading in the Yahara river basin (Montgomery Assocaites, 2014). Considering similar P loading in the upper Yahara watershed region, the total P runoff in the area is 2135 tons/yr. In this work, we focus on the economic impact of P runoff from livestock waste which contributes 917.83 tons/yr. The croplands in our study area, with a total area of 50, 593 acres (Sharara et al., 2017), have a total P uptake capacity of 629.74 tons/yr (Sampat et al., 2019). There is thus a surplus of 288.09 tons/yr of P. We consider that 10% of this excess P runs off to Lake Mendota. To keep the calculations on a conservative side, we have assumed that 10% of excess P runs off to the lake instead of the 10% of applied P (which is the number used in state-of-the-art LCA methods like ReCiPe (Huijbregts et al., 2016) for mid-point and end-point environmental impact categories). The initial TP concentration of Lake Mendota is considered to be 53 $\mu g/L$ (based on the average TP concentration for the years 2014 – 2017) (Wisconsin Department of Natural Resources, 2017). Due to the P runoff from the overapplication of waste, we estimate (by mass balance calculations for the year 2017) the TP concentration of Lake Mendota increases to 110 $\mu g/L$ (considering the lake volume to be 505 million m^3 (Lathrop and Carpenter, 2011)). This increase in TP concentration acts as a basis for our calculations for quantifying the economic impacts associated with algae blooms.

3.1. Economic impacts of algae blooms in the Upper Yahara watershed region

3.1.1. Property values—When the initial TP concentration in Lake Mendota is 53 $\mu g/L$, the Secchi depth is 0.97 m (by Eq. 1). After P runoff, when the TP concentration of the lake increases to 110 $\mu g/L$, the Secchi depth decreases to 0.64 m. This 0.34 m decrease in Secchi depth corresponds to an estimated 5.3% reduction in all property values on the Lake Mendota shoreline (according to Dodds et al. (2009) 1 m reduction in Secchi depth reduces property values by 15.3%). Lake Mendota has a shoreline length of 33.8 km (North Temperate Lakes, 2018). Assuming an average lot length of 54.64 m (Wisconsin Department of Natural Resources, 2012), there are 619 lots on the lakeshore. We consider 85% (Wisconsin Department of Natural Resources, 2012) of these lots are private properties, and have a median property value of 269,100 USD (Zillow, 2018). The reduction in Secchi depth

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results in a total loss of 7.46 million USD/yr. This is equivalent to 25.9 USD/kg excess P released.

3.1.2. Recreational cost—In our case study for the Upper Yahara watershed region, the Secchi depth decreases by 0.34 m. The impact of a reduction in Secchi depth on the frequency of participation in fishing and swimming is summarized in Table 2. For the current odds of participation, a survey by the Wisconsin Department of Natural Resources (2011) reports that 37.4% of residents participate in freshwater fishing and 41.7% swim in lakes. The current odds for fishing and swimming are thus 0.60 and 0.72, respectively. Using the Eq. 5 and the logit coefficients from Table 1, the new odds for fishing and swimming are 0.58 and 0.72 respectively. This corresponds to a new participation of 36.6% and 41.7% in fishing and swimming respectively. In case of fishing, the participation reduces by 0.8% while there is no change observed in case of swimming. There is no impact on the participation probability in swimming because the logit coefficient $\left(\beta_1^s\right)$ estimated by Vesterinen et al. (2010) is close to zero (Table 1).

Wisconsin anglers participate in 17.3 days (U.S. Fish and Wildlife Service, 2008) of fishing annually, while the frequency of swimming trips (by Wisconsin residents) is considered to be 5 days (Wisconsin Department of Natural Resources, 2012). Using these frequencies of participation and the negative binomial coefficients listed in Table 1, we estimate that a decrease in Secchi Depth of 0.34 m reduces the frequency of participation in fishing and swimming to 16.8 and 4.9 days respectively (by Eq. 6). These results are summarized in Table 3.

After quantifying the impacts on the probabilities and the frequencies of participation, we estimate the corresponding loss in revenue (summarized in Table 4). Kaval and Loomis (2003) estimate the value of a day spent fishing and swimming to be on average of 63.27 USD and 57.27 USD, respectively (converted to 2018 USD). For our study area, we consider that the participants are from the Dane County, WI, which has a population of 536,416 U.S. Census Bureau, 2018). We have not considered participation from non-resident anglers or swimmers in our calculations. For our study area, we estimate a total loss of 11.8 million USD/yr and 1.19 million USD/yr in fishing and swimming respectively (by Eqs. 7 and (8). This translates to a loss in revenue (from recreational activities) of 45.4 USD/kg excess P.

3.1.3. Clean-up expenses—Lake Mendota is not a source of drinking water and thus alum treatment is not performed. However, the excessive amount of phosphorus runoff in the Yahara river waterbodies over the years has resulted in high phosphorus deposition in the bed of the streams that feed into the lake. Thus, even if all the agricultural runoff was successfully prevented from entering the Yahara river waterbodies, Lake Mendota would still be prone to algae blooms for decades to come (Land & Water Resources Department, 2019). The Dane County is implementing a project called Suck the Muck (Wisconsin State Journal, 2019) to pump out phosphorus-laden sludge from the bottom of creeks and streams to combat the toxic algae blooms. The estimated cost of this project is 12 million USD for removing 870,000 pounds of phosphorus (or 30.2 USD/kg P removed) from the streams leading to the Yahara lakes. For our case study, where the excess P is 288 tons/yr and 10% of

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this excess P is assumed to runoff, the annual cost of lake cleanup translates to 3.0 USD/kg excess P.

3.1.4. Summary of economic impacts—We summarize the economic estimates for the impacts due to harmful algae blooms in Lake Mendota in Table 5. From our analysis, the impact on the recreational activities is the highest (45.4 USD/kg excess P) followed by the impact on the property values (25.9 USD/kg excess P). Overall, every excess kg of P results in an economic loss of 74.5 USD. As we demonstrate in the next section, this economic impact can be useful in designing and activating a market that facilitates the coordinated management of organic waste.

3.2. Upper Yahara coordinated market

In this case study, we consider the 203 livestock farms in the Upper Yahara watershed region as the suppliers of waste. The waste is categorized into beef, dairy cow, and heifer manure and initially assumed to be offered for free. Amongst these manures, dairy cow manure has the highest P concentration (Agricultural waste management field handbook, 1992; Nennich et al., 2005). As described in the Waste Processing section, the following products can be derived from manure: granulated compressed pellets, struvite, digestate, and the manure solid fraction. In this coordinated market setting, the agricultural lands are the consumers ($\mathscr{D}$) that demand raw manure, solid fraction of manure, and digested manure. The case study includes 1167 agricultural land nodes that can be used for waste application (to fulfill nutrient requirements). The set of consumers also includes external players (located outside the region in Madison, WI or Sauk County, WI) that accept waste surplus and buy pellets, struvite, and the solid fraction of manure. We consider the demand bidding costs to be similar to the market value of products: 100 USD/tonne for pelleted waste, 800 USD/tonne for struvite, 0.05 USD/tonne for the solid waste fraction, 0.002 USD/tonne for the liquid waste fraction, and 0 USD/tonne for the digestate (Sampat et al., 2018; Sharara et al., 2018b). Location and capacity data for demand and supply nodes are obtained from Sampat et al. (2018); Sharara et al. (2017, 2018b). The supply capacity of waste for dairy farms are based on the number of cows present at the farm and the demand capacity for croplands is based on the land area and the type of crop grown. We consider transportation bidding cost of each route as 0.3 USD/tonne-km for manure and digestate (in case of pellets and struvite this value is 0.15 USD/tonne-km as solids are easier to transport). For simplicity, the transportation paths between nodes are assumed to be linear and that transportation bids exist to move product between all nodes in the network/market. This gives rise to a complex logistical network. The corresponding market clearing problem is a linear program with over 30 million decision variables and 0.5 million constraints. This problem can be solved with modern solution tools such as Gurobi (version 7.5.2) (Gurobi optimizer reference manual, 2018) in 15 mins. All the scripts required to reproduce the results are available at https://github.com/zavalab/JuliaBox/tree/master/EconomicImpacts.

As described earlier in the Waste Processing section, we consider three different processing options for waste treatment (Fig. 1). The processing costs for these technologies are 0.23 USD/tonne raw manure for separation, 4.00 USD/tonne raw manure for granulation, and 38.1 USD/tonne liquid feed for stuvite recovery (Sampat et al., 2018; Sharara et al., 2018b).

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We note that struvite recovery is more expensive as it involves sophisticated technology. A tradeoff exists between the processing cost and the value of the product recovered. Struvite produced is a more concentrated in P and valuable than pellets; while pellets is more concentrated in P and valuable than the manure solid fraction. In this case study, we consider 126 hypothetical technology installations to be located at large farms (having over 500 animal units). The technologies include 61 separation units, 3 granulation units, and 62 struvite recovery units. Only CAFOs (concentrated animal feeding operations) with over 1000 animal units were considered for the installation of granulation technology. The installation locations are randomly selected and shown in Fig. 5.

Under this setting of market players for livestock waste management in the Upper Yahara, we apply the coordination framework (described in the Coordinated Market Model section). Attributing an economic impact (value of service VOS or λ) of 74.5 USD to every kg excess P provides an external driving force to process waste and balance the P in the study area (Table 6). This VOS can be provided by federal or state agencies to the dairy farmers as a part of incentives for processing waste and avoiding nutrient pollution. Under this scheme, the optimal strategy is to use separation and granulation technologies to process waste and transport the excess P (in the form of pellets) out of the watershed region to areas that are deficient in soil P concentration. As a result, the market model predicts that there is no excess P in this scenario.

Since there is uncertainty around the exact value of VOS, we perform a sensitivity analysis to study its impact on the overall P distribution. We observe that, in absence of an external driving force (VOS = 0 USD/kg excess P), no waste is processed and there is 45.6% excess P (Fig. 6). If the VOS value is reduced to 19 USD/kg excess P (25% of the estimated value), there would still be 14.3% excess P in the study area. This VOS value would only be able to activate the use of separation technologies, leaving some waste in the study area untreated. Whereas, when the VOS is 149 USD/kg excess P (twice the estimated value), the external driving force is high enough to balance P in the region by using separation and granulation technologies. A VOS value of 45 USD/kg excess P is the break-even value that completely balances excess P in the upper Yahara watershed region. We note that, in none of these scenarios, struvite recovery technology was selected. Even though struvite has a higher market value, the high processing cost associated to this product prevents it from being economically competitive to separation and granulation technologies.

For an economic impact of 74.5 USD/kg excess P, the clearing prices for beef and dairy cow manure are summarized in Fig. 7. Here, the clearing prices are negative, indicating that the farmers need to pay a monetary amount in order to get rid of their waste. In case of beef and dairy cow manure, the farmers need to pay on average 16.6 USD/tonne and 23.5 USD/tonne respectively. The clearing price of dairy cow manure is higher since it has more P concentration (Agricultural waste management field handbook, 1992; Nennich et al., 2005) compared to the beef cow manure. Moreover, the clearing prices capture the geographical distribution of P in the study area. For the areas with higher concentration of P, the clearing prices are more negative. These values also act as a price signal that can drive more investment in the areas with more negative clearing prices. One strategy to fund these payments can be through federal and state incentives that promote waste management

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practices in areas where phosphorus loading is high. This allocation of environmental cost amongst stakeholders will be analyzed in detail in our future work.

4. Conclusion

We have presented a computational framework to estimate the economic impacts of nutrient pollution from livestock waste. It is difficult to distinguish the economic impact of nutrient pollution from that of HABs. Nonetheless an order of magnitude estimate of this impact can guide federal and state agencies to design policies and tools that reduce nutrient pollution (U.S. Environmental Protection Agency, 2015) which in turn causes HABs. Moreover, our methodology can capture the geographical features of nutrient pollution through the environmental cost (or VOS) and clearing prices (Fig. 7). Our analysis reveals that every excess kilogram of phosphorus in the Upper Yahara watershed region results in an economic loss of 74.5 USD. In addition, we observe that for this case study the environmental cost is higher than the break-even cost to drive processing of livestock waste. Thus justifying the investment in waste processing technologies. This analysis is based on a steady state analysis and does not account the temporal system variations. Our future work will analyze the allocation of the environmental cost i.e. which stakeholder should pay for the economic loss in order to drive waste processing. One strategy to fund these payments can be through federal and state incentive programs that promote waste management practices in areas with high phosphorus imbalances.

Acknowledgments

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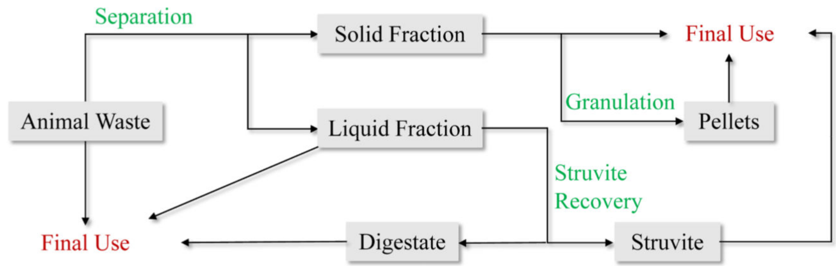

Fig. 1.
Different processing options for livestock waste.

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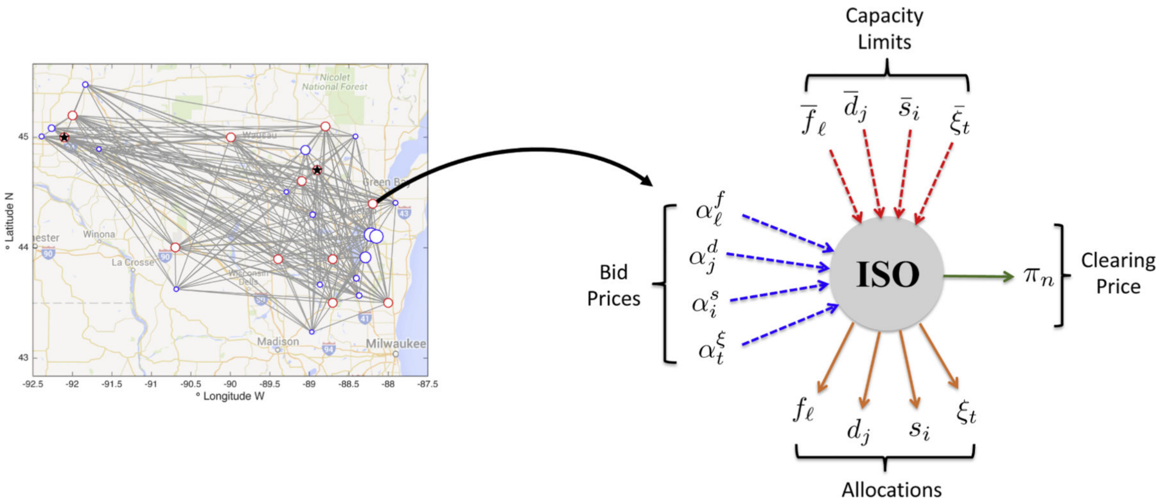

Fig. 2.

For every node in the supply chain network, the ISO (independent system operator) accepts bid prices and capacity limits from the market players (e.g. farmers, fertilizer consumers, federal and state agencies etc.). The ISO then solves the market clearing problem (Eq. 9) to find the clearing prices of the services and the corresponding service allocations.

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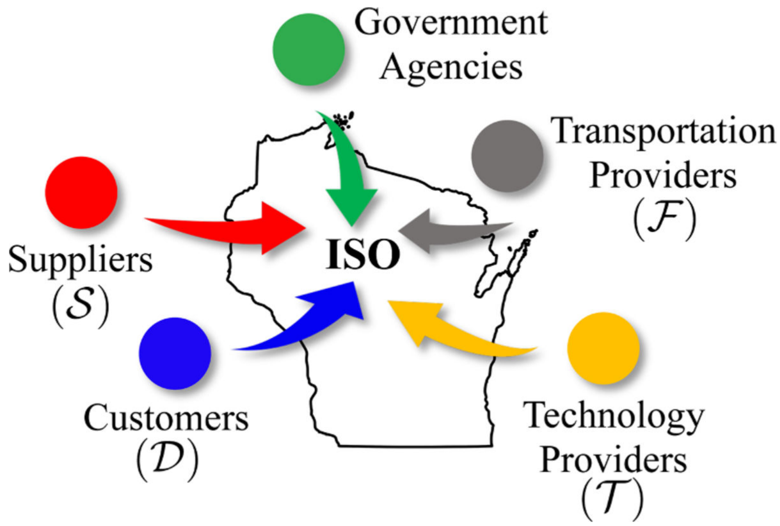

Fig. 3.
Market players and the corresponding mathematical set notations indicated in parenthesis. The market players submit bid prices and capacity limits for their services to the ISO.

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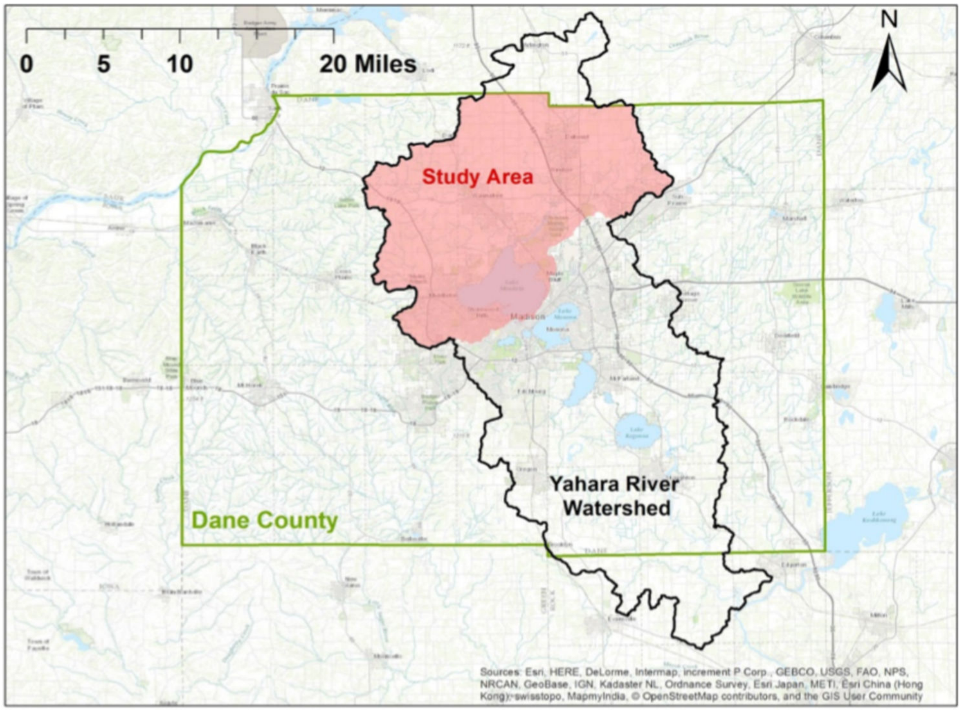

Fig. 4.

Lake Mendota in the Upper Yahara watershed region in Dane County, WI is the study area for quantifying the economic impacts of nutrient runoff.

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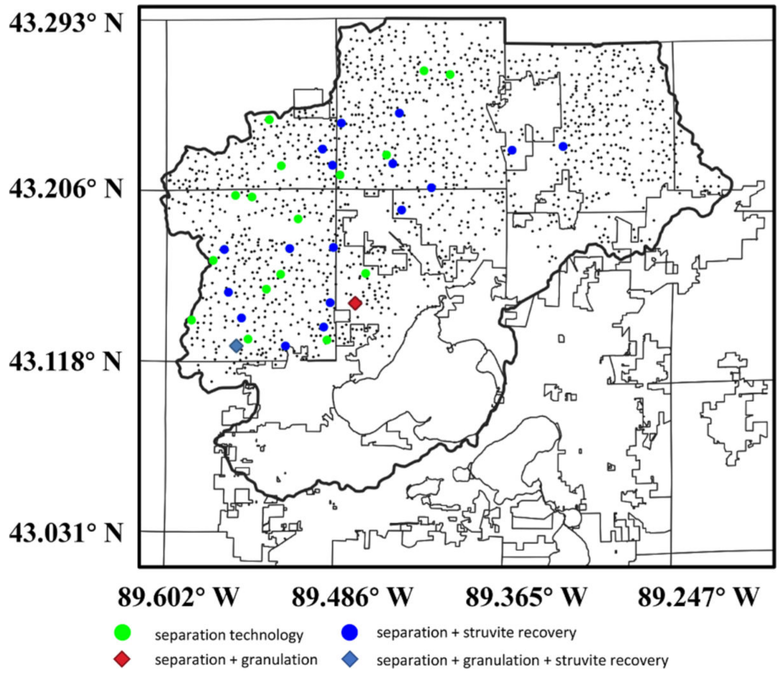

Fig. 5.

Locations considered in the case study for agricultural lands, farms, and waste processing technologies in the Upper Yahara watershed region. Small dots indicate location of farms and agricultural lands.

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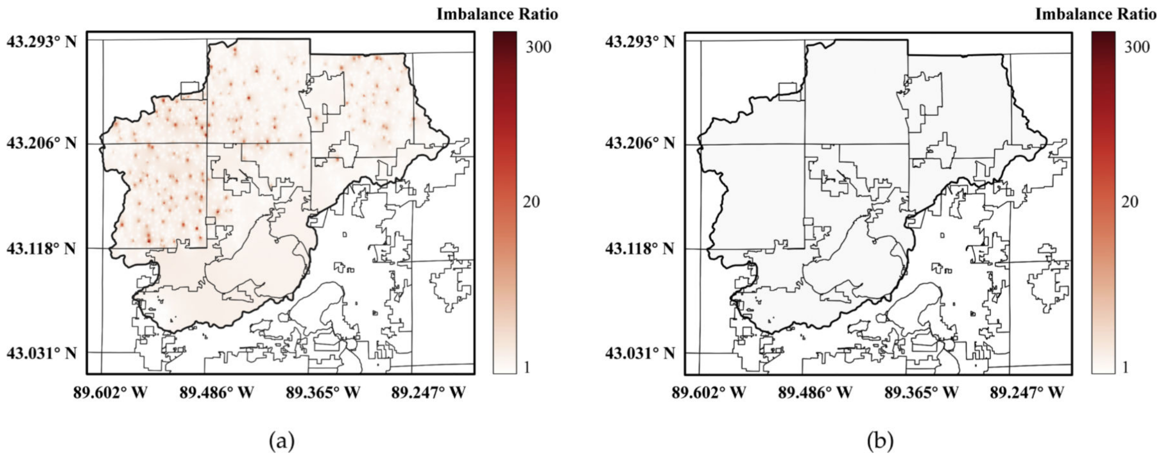

Fig. 6.

Phosphorus (P) imbalance maps for a value of service (VOS) of (a) 0 USD/kg excess P and (b) 74.5 USD/kg excess P in the Upper Yahara watershed region. Note that the imbalance ratio is presented in a logarithmic scale.

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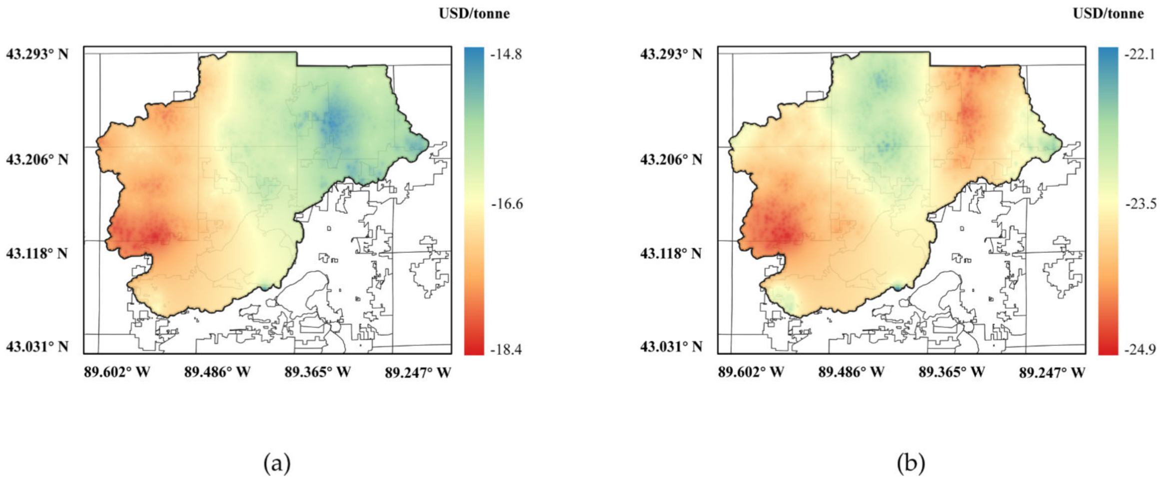

Fig. 7.
Clearing price for (a) beef cow manure and (b) dairy cow manure in the Upper Yahara watershed region (corresponding to a VOS of 74.5 USD/kg excess P).

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Table 1

Logit and negative binomial coefficients for water recreational activities with respect to water clarity (based on the hurdle model by Vesterinen et al. (2010)).

Independent Variable	Swimming Logit $\left(\beta_1^S\right)$	Negbin $\left(\gamma_1^S\right)$	Fishing Logit $\left(\beta_1^F\right)$	Negbin $\left(\gamma_1^F\right)$
Water Clarity	−0.006	0.059	0.107	0.097

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Table 2

Impact of reduction in Secchi depth (of 0.34 m) on the probability of participation in fishing and swimming in Lake Mendota.

Activity	**Fishing**	**Swimming**
Current Participation	37.4%	41.7%
Current Odds $\left(O_1^A\right)$	0.60	0.72
New Odds $\left(O_2^A\right)$	0.58	0.72
New Participation	36.6%	41.7%
Loss in Participation	0.8%	0%

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Table 3

Impact of reduction in Secchi depth (of 0.34 m) on the frequency of participation in fishing and swimming in the Lake Mendota Lake M.

Activity	Fishing (days/yr)	Swimming (days/yr)
Current Frequency $\left(\mu_2^A\right)$	17.3	5
New Frequency $\left(\mu_1^A\right)$	16.8	4.9

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Table 4

Loss in revenue from recreational activities due to a decrease in Secchi depth of 0.34 m in Lake Mendota.

Activity	Fishing	Swimming
Loss in Trips (trips/yr)	1.9×10^5	2.1×10^4
Average Trip Cost (USD/trip)	63.3	57.3
Loss in Revenue (USD/yr)	11.9×10^6	1.2×10^6

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Table 5

Summary of economic impacts of excess phosphorus (resulting in HABs) in the Upper Yahara watershed region.

Impacted Category	Economic Loss (USD/kg excess P)
Property Value	25.9
Recreational Activities	45.4
Lake Cleanup	3.0
Human and Pet Health	–
Total Monetized Loss	**74.5**

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Table 6

Sensitivity analysis for different values of economic impact (or VOS).

Economic Impact (USD/kg P)	Excess P (%)	Technology Selected
0	45.6%	–
19	14.3%	Separation
45 (break-even value)	0%	Separation + Granulation
74.5 (estimated value)	0%	Separation + Granulation
149	0%	Separation + Granulation