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Analysis

Optimizing Spatial Land Management to Balance Water Quality and Economic Returns in a Lake Erie Watershed

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ABSTRACT

Significant reductions in phosphorous (P) inputs from cropland are needed to address the re-eutrophication of Lake Erie. Previous studies aimed at addressing non-point source pollution have primarily analyzed the effectiveness of conservation practices (CPs) as land-management strategies. However, the effectiveness and efficiency of these practices have not been compared to those of possible land-use changes. We develop a spatially explicit integrated modeling approach that compares the effectiveness and economic efficiency of alternative spatially optimal land-use and -management strategies for P abatement in the Sandusky River watershed. Using the Soil and Water Assessment Tool and data on costs and profits from crop and forest production and urban development, we evaluated joint impacts on P reduction and economic-returns for optimized land-use changes and/or implementation of CPs in the watershed. Results showed a combination of both CPs and land-use changes are likely required to meet current abatement targets for dissolved reactive phosphorus. Additionally, the combination of these approaches can generate a positive, synergistic effect on economic efficiency in meeting key policy targets. This is largely because the combined strategy will establish CPs on the most productive cropland, while achieving greater nutrient reduction through land-use change away from corn-soybean rotations on less productive lands.

1. Introduction

Nutrient loading resulting from extensive agricultural land development has been repeatedly linked with declines in surface water quality and related ecosystem and human health problems such as harmful algal blooms (Michalak et al., 2013; Schilling et al., 2010). To improve sustainability in land-use systems, including reducing such environmental impacts, recent studies have highlighted the importance of incorporating ecosystem services into decisions about land use and management (Goldstein et al., 2012; Guerry et al., 2015). At the same time, several regulatory programs for water quality have been put into place in the United States (e.g., Clean Water Act and Total Maximum Daily Loads (TMDL)) (Houck, 2002). However, current implementation of these efforts is far from sufficient to restore watersheds and reduce harmful algal blooms (Hoornbeek et al., 2013; NRC, 2009) largely because there are no mandatory regulations for controlling non-point source (NPS) pollution from agricultural land.

Among the multiple possible strategies to mitigate NPS pollution, many studies have focused on implementation of conservation practices (CPs) (Kalcic et al., 2016; Maringanti et al., 2009; Rabotyagov et al., 2014a; Scavia et al., 2014; Tomer and Locke, 2011; USDA NRCS, 2010). Agricultural CPs are management tools aimed at controlling soil erosion and reducing nitrogen (N) and phosphorus (P) discharge to surface waters (USDA NRCS, 2010). Previous studies have recommended spatial targeting of CPs as a cost-effective approach to reducing NPS pollution (Kalcic et al., 2015; Maringanti et al., 2011; Rabotyagov et al., 2014a), but the P abatement effectiveness of these approaches is limited (Bhattarai et al., 2009; Kleinman et al., 2011; Lemke et al., 2011). Also, various studies have highlighted the need to differentiate dissolved reactive phosphorous (DRP) from TP reductions because TP includes P adsorbed on mobilized sediments, whereas DRP includes only dissolved reactive P. In general, it is much more difficult for agricultural CPs to reduce soluble nutrient loadings, such as DRP, than to reduce sediments (Kleinman et al., 2011; Mueller-Warrant et al., 2012; Osmond et al., 2012). Several watershed level experiments led by USDA (Osmond et al., 2012) show that current agricultural CPs provide improvement of downstream water quality. In watersheds with substantial agricultural land, aggressive implementation of CPs at watershed scales would be

Abbreviations: USGS, United States Geological Survey; USDA NASS, United States Department of Agriculture National Agricultural Statistics Service; USDA ERS, United States Department of Agriculture Economic Research Service; USDA FSA, United States Department of Agriculture Farm Service Agency; USDA NRCS, United States Department of Agriculture Natural Resource Conservation Service

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needed to achieve even moderate (~30%) reductions in nutrient discharges (Bosch et al., 2013; Kalcic et al., 2016, 2015; Rabotyagov et al., 2014a; Volk et al., 2008).

Even if CPs were sufficient to meet NPS reduction targets, we need to consider whether they are the most economically efficient means of achieving those reduction targets. Before making large investments in new conservation actions, it is worthwhile to compare the physical effectiveness and economic efficiency of alternative strategies. In fact, some studies have suggested that diversifying agricultural landscapes with perennial plants should be a fundamental strategy for restoring agroecosystem health (Mueller-Warrant et al., 2012; Schulte et al., 2006). In recent year, "landscape approaches" have been recommended as a promising strategy to reconcile trade-offs in conservation practices (which help to abate P runoff) and agricultural production (which strongly governs economic gain from the land) (Sayer et al., 2013). By re-allocating land uses to suitable locations based on the comparative advantages of each land unit, previous studies have demonstrated that optimizing land-use patterns can be an effective approach to jointly improving regional economic and ecosystem services outcomes (Nelson et al., 2008; Polasky et al., 2008; Ruijs et al., 2015; Seppelt and Voinov, 2002). Although individual landowners may lose profit as a result of spatial optimization, the public-sector costs of providing incentives for land-use modification (e.g., converting corn-soybean rotations to other land-use types, like alfalfa hay, grasslands, or forests) might be lower, per unit of P abatement, than the costs to compensate farmers for implementing CPs.

To make watershed or landscape-scale land use and management choices more effective in lowering NPS pollution, it is essential to understand the comparative advantages of possible strategies in a spatially explicit manner (Ruijs et al., 2015). Various studies have utilized watershed modeling and spatial optimization algorithms to evaluate tradeoffs between agricultural production and environmental services under different policy scenarios (e.g., Lautenbach et al., 2013; Valcu-Lisman et al., 2016). Nonetheless, previous studies have rarely considered the role of changes in the spatial patterns of land use versus land management when assessing cost-effectiveness of CPs for reducing NPS pollution. Land retirement has been occasionally explored as land-use change and compared with CPs (Kling, 2011; Rabotyagov et al., 2014b), probably because it is relatively easy to implement and monitor. However, retiring cropland from production will be significantly more expensive relative to CPs on a field-by-field basis (Rabotyagov et al., 2014a), because government subsidies are often needed to compensate farmer's financial loss. A strategy that includes land-use changes and working land options, such as timber and hay production, could potentially be more cost-competitive than land retirement, depending on land conversion costs, management costs and prevailing economic markets for timber and hay. In this case, whether one strategy would be relatively more economically efficient than the other, given the same nutrient reduction objective, remains an open question. Therefore, an understanding of the tradeoffs and complementarities in physical efficacy and economic net returns and benefits of land-use (e.g. conversion to switchgrass or working forest) versus land-management (e.g. using CPs) approaches to reducing nutrient pollution is needed to fill a knowledge gap and to guide future planning. Previous studies have not compared these two approaches in an integrated modeling framework.

In this study, we aim to close the knowledge gap by developing an integrated modeling framework for Lake Erie's Sandusky River watershed that (i) evaluates the physical P abatement effectiveness of CPs, conversions of cropland to other land use/cover (LUC) types, and combinations of both strategies for reducing NPS pollution, and (ii) compares economically optimal spatial land-use and -management patterns based on different decision-making strategies. The joint ecological and economic performance of alternative land-use and CP options was estimated using an ecohydrological model combined with an economic net-returns component. Using this approach, we identify efficient spatial patterns of land-use and -management changes, given estimated field-level tradeoffs in performance on economic and nutrient-reduction goals. In our analysis, a solution is considered efficient if it maximizes economic returns to landowners for a given nutrient-loading reduction outcome, and vice versa. Instead of providing a single optimal solution, we developed a Pareto efficiency frontier that represents a range of nutrient-reduction objectives and their associated maximum economic returns for landowners.

2. Materials and Methods

2.1. Study Area and Scenario Development

Using data from the watershed of the Sandusky River in northern Ohio, we demonstrate the opportunities for an integrated approach to reducing non-point source pollution [see supplemental information (SI) Fig. S1a]. The watershed covers about 3458 km^2 and is dominated by cropland (> 80%) with some areas of urban development (Fig. S1b). Over 85% of the cropland is maintained as corn-soybean (C-S) rotation (USDA NASS, 2016). The landscape is generally flat, with some gently rolling plains in the central and southern portions (Fig. S1c). About 32% of the watershed's soil (Fig. S1d) is in the poorly or very poorly drained category (USDA NRCS, 2015) and artificial subsurface (tile) drainage is used to support agricultural production by lowering the water table below the crop rooting zone.

The Sandusky watershed is part of the Lake Erie basin, which has received increased public attention due to recurrent massive microcystis blooms. In 2011, Lake Erie experienced its largest recorded bloom (Michalak et al., 2013) and in 2014 a significant bloom rendered drinking water unsafe for the city of Toledo. To restore good water quality, the International Joint Commission (IJC) established the Lake Erie Ecosystem Priority in 2012, and recommended that the United States and Canadian governments take action to significantly reduce P loading to Lake Erie (IJC, 2014). To decrease the hypoxic area in the central Lake Erie Basin by 50% and limit the number of hypoxic days to 10 days per year, the IJC recommended 46% and 78% reductions in the total phosphorous (TP) and DRP loads, respectively, relative to the 2003 to 2011 and the 2005 to 2011 annual averages. Notice that these targets apply to combined TP and DRP loads from the western and central basins. In addition, some studies have suggested that control of DRP loads should be the focus in the study area, because it is more directly related to the algal bloom problem (IJC, 2014; Scavia et al., 2014).

By comparing the relative economic efficiency of three approaches to P reductions, with each scenario resulting in an efficiency frontier, our analysis will help policymakers understand the tradeoffs between land-use and -management approaches, and between agricultural production (e.g. crops, timber, and hay) and P reduction under each scenario. To establish a baseline, we assume farmers will maintain any cropland in the watershed as a C-S rotation. Spatial explicit data on current CP coverage is not publically available for the region (Kalcic et al., 2016). A recent survey for the western Lake Erie basin (USDA NRCS, 2011) shows that about 65% of the soybean acres planted are grown using no tillage, which is helpful for reducing soil erosion. However, only 19% of corn acres were planted using no-till. Using data from 2012 U.S. Agriculture Census (USDA NASS, 2012) and 2016 Crop Acreage Data (USDA FSA, 2017) reported to USDA FSA, it can be estimated that about 11.5% of the cropland in the watershed is currently under CPs (See *SI text: Exiting CP Coverage*). For the baseline simulation, we assume that no CPs on cropland, except no tillage for soybean. The three P reduction scenarios are:

(1) *Optimizing placement of CPs for each level of P abatement*: selected C-S fields receive one of five frequently adopted CPs (Tomer and Locke, 2011), including reduced tillage, no tillage, vegetative filter strips, grassed waterway, and winter cover crops. We added to this list a nutrient management option, assuming that farmers reduce

fertilizer applications by 20%.

(2) *Optimizing land-use conversions for each level of P abatement*: selected cropland could be converted from a C-S rotation to one of the following alternative land-use types: switchgrass hay, switchgrass as biofuel feedstock, alfalfa hay, managed forestry, CRP modeled as grassland, and rural-residential land.

(3) *Combined optimization of CP and land use conversions for each level of P abatement*: each targeted farm field could either implement a CP or be converted to another land-use option such as switchgrass, residential land, or forest.

In each of the three scenarios, we assume C-S rotation is the default option for all fields and only cropland can be converted to other land uses. For a given field, only a single land-use type can be selected, so multiple uses (e.g. C-S and grassland) in the same field are not supported. The amount of cropland that can be converted to other land uses is not limited by quotas, except that any increase in new residential land is limited to 10% of the current urban land area. We further assume market prices are uniform within the watershed and insensitive to changes in land use composition.

2.2. Overall Simulation Framework

We developed an integrated modeling framework (Fig. 1) that links outputs from an ecohydrological model – the Soil and Water Assessment Tool (SWAT) (Arnold et al., 2010, 1998) and an economic net return component to an optimization algorithm. We used the SWAT model to estimate the effects of conservation practices and alternative land-use types on crop yields and nutrient discharges for each field. Field-level options were integrated into a spatial optimization model based on mixed-integer linear programming. Individual components of the framework are described in the following sections.

2.3. Watershed Model Setup and Calibration

The SWAT model is a semi-distributed, hydrologic model that has been widely used to evaluate the effects of alternative management decisions on nonpoint-source pollution in large river basins (Arnold et al., 2010). A baseline SWAT model was developed for the Sandusky River watershed using elevation, stream network, soil type, land use and climate data (Table 1). By default, the SWAT model divides a watershed into several sub-basins. In each sub-basin, discrete land areas that share land use, soil, and slope characteristics are merged into a single hydrological response unit (HRU), which is the basic spatial unit in SWAT. This definition limits the applicability of the SWAT model for optimizing spatial land management because there is a disconnect between model units and landowners' decisions (Kalcic et al., 2015). In this study, we defined HRUs instead as field boundaries to link SWAT units with land management practices, using a method similar to Kalcic et al. (2015). We used the Common Land Unit (CLU) data layer, originally developed by USDA to delineate agricultural field boundaries, together with parcel maps obtained from local governments, to form the spatial unit of analysis. Land use and management were represented over these spatial units defined in the rural and urban areas by CLU and parcel boundaries, respectively. The resulting map divided the Sandusky watershed into 41,233 distinct spatial units. A 2006 land-use/cover map was overlaid with field boundaries to identify an initial land-use pattern. We apply alternative land-use and -management options only to current cropland (27,905 units). Other current uses, such as existing urban land and forest cover, were kept constant. More details about the field boundary map can be found in the Supporting Information (SI).

We ran the field-based SWAT model for 1993–2010, including three years for model spin up (1993–1995), 11 years of calibration (1996–2006) and four years for validation (2007–2010). The long calibration period is necessary to capture both high and low flow patterns: monthly peak flow is generally lower than 100 m^3/s prior to 2004, but there are several high flow months ($>$ 150 m^3/s) between 2004 and 2010 (Fig. S5). Twenty-eight parameters were changed in calibration (Table S4). The initial set of parameters was collected from the literature (Kalcic et al., 2016). A sensitivity analysis based on the Sequential Uncertainty Fitting (SUFI-2) algorithm was conducted to narrow down the list of parameters, using the SWAT-CUP software (Abbaspour et al., 2007). Final parameter values were determined via manual calibration. We used measured streamflow data for the Fremont station (Fig. S1a) and water quality data from Heidelberg University (2015) to evaluate the fit of the model's estimate near the outlet of the watershed. In terms of model evaluation, a wide range of methods exist in the literature and previous studies often use different metrics (Bennett et al., 2013). For this study, calibration and validation results based on three common model evaluation criteria for hydrological modeling (Moriasi, 2015), including goodness-of-fit (R^2), the Nash-Sutcliffe coefficient (NSE), and the percent bias (PBIAS), indicate that the model accurately predicted streamflow and nutrient load (Table 2). We also calibrated model-simulated crop/plant yields against reported values from USDA NASS (Table S6). Assuming the baseline scenario of corn-soybean rotation for all agricultural fields, we simulated the effects of CPs by adding selected CPs to each field. To optimize land uses, we replaced the C-S rotation with alternative crop or land-use options to meet target reductions in P discharges aggregated across the watershed as a whole. Details on

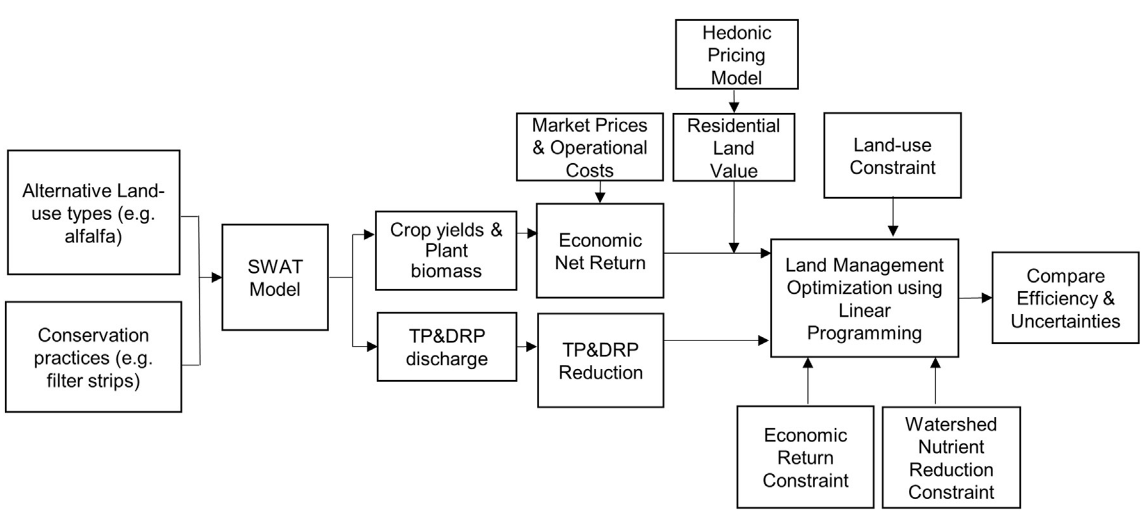


Fig. 1. Modeling framework.

Table 1
Model input data sources for the Sandusky Watershed. USGS, USEPA, USDA-NRCS and NOAA refer to U.S. Geological Survey, U.S. Environmental Protection Agency, U.S. Department of Agriculture-Natural Resource Conservation Service, and National Oceanic and Atmospheric Administration. Multi-Resolution Land Characteristics Consortium (MRLC) is a group of 13 federal programs in 10 agencies.

Data type	Resolution or scale	Source	Data description/properties
Digital Elevation Model (DEM)	30 m	USGS (Gesch et al., 2002)	Elevation and terrain
National Hydrography Dataset Plus Version 2 (NHDPlus V2)	1:24,000	USGS and USEPA (McKay et al., 2012)	Stream network
Soil (SSURGO)	1:12,000	USDA-NRCS (USDA NRCS, 2015)	Soil physical properties such as pH, texture and bulk density
Land use (NLCD 2006)	30 m	MRLC (Fry et al., 2011)	Land use classifications
Climate (Global Historical Climatology Network-Daily (GHCN-DAILY) data)	7 stations	NOAA (Menne et al., 2012)	Daily precipitation, minimum and maximum temperature
Land management information	State	USDA, Ohio State University Extension and literature (see SI SWAT model setup for more details)	Fertilizer application rates and timing, planting and harvesting dates

Table 2
Monthly statistical performance of the SWAT model. Performance rating for goodness-of-fit (R^2), Nash-Sutcliffe Efficiency (NSE), and percent bias (PBIAS) are based on Moriasi (2015).

	R^2	NSE	PBIAS	Performance rating
	Calibration (validation)	Calibration (validation)	Calibration (validation)	Calibration (validation)
Stream flow	0.83 (0.89)	0.82 (0.87)	− 1.66 (− 2.76)	Very good (very good)
Sediment	0.90 (0.77)	0.83 (0.73)	− 1.02 (9.8)	Good (good)
TP	0.73 (0.77)	0.66 (0.76)	− 4.83 (− 2.13)	Very good (very good)
DRP	0.59 (0.69)	0.56 (0.62)	− 4.84 (− 11.35)	Good (good)
TN	0.84 (0.75)	0.83 (0.75)	2.29 (− 0.27)	Very good (very good)

model parameterization, calibration, and simulation of CPs and land-use change are provided in the SI.

2.4. Net Return and Cost Estimation

For field crops and forest, observed 14-yr average (2001–2014) market prices and simulated yields from SWAT were used to calculate revenues. Data from USDA ERS (2015) and Ohio State University (OSU)-Extension (2016, 2015) were used to estimate costs so that net profit could be calculated for each option. Our calculations of economic returns did not include government payments or subsidies to growers, such as direct payments, when they took place on productive agricultural land. Our calculations of economic returns did include government payments to landowners related to CRP enrollment. Economic returns of land enrolled in CRP were calculated as the county average of CRP payments minus maintenance costs. Our calculations of economic returns related to land use also included hedonic price modeling of residential land; we modeled the present value per acre of rural-residential land as a function of geographic and site conditions. Using real estate appraisal and sales data from local tax departments, we estimated a hedonic price function for the value of rural-residential land use at each farm (see *SI Text: Economic Valuation in the Model*). Estimated economic returns for residential land use were converted to a constant annualized equivalent for use in the economic return calculation for residential land use.

2.5. Optimization Algorithm

Given estimates of nutrient discharge and economic return (*ER*) for each option at each field, the goal of optimization is to find the efficient spatial configuration of changes to land use and land management, which we define as the configuration that would maximize economic return for the aggregate of all landowners in the watershed for a certain level of nutrient-load reduction. Since choices made at each land unit are discrete, this problem can be formulated as a mixed integer program involving a large number of fields. Notice that the land-use constraint is not directly included in the equations, but implemented by limiting land-use conversion to occur only from cropland to other land uses, as mentioned above. This means fields with other land uses (e.g. forest, settlement) cannot be changed during the optimization process. By using a series of discrete values for the nutrient (TP and DRP in this study) load constraint (*LC*) and optimizing land use and management separately for each value of LC for each scenario, we were able to trace out Pareto efficiency frontiers for each of the three scenarios. The use of optimization algorithm that can trace Pareto efficiency frontiers combined with scenario analysis is recommended as effective approach to sustainable land management (Seppelt et al., 2013). A Pareto efficiency frontier identifies the maximum gains in economic returns over baseline conditions that can be obtained from the landscape given a range of fixed P reduction objectives. The origin (0, 0) represents the status quo point for the baseline condition. Each point on the frontier corresponds to an optimized spatial allocation of land-use and/or -management changes. A point on the efficiency frontier is found by identifying an optimal pattern of land-use changes and/or CPs that solves the following:

$$ER = Max \sum_{j=1}^{N} \sum_{k=1}^{P} R_j^k A_j (X_{k,j}) \tag{1}$$

SubjectTo:

$$\sum_{j=1}^{N} \sum_{k=1}^{P} NP_j^k X_{k,j} \le LC \tag{2}$$

$$\sum_{k=1}^{P} X_{k,j} = 1 \quad j = 1, \ldots, N \tag{3}$$

$$k_j = \{0, 1\} \quad j = 1, \ldots, N \tag{4}$$

where j indexes fields ($j = 1,\ldots,N$ where $N = 27{,}905$), k indexes land-use or CP options ($k = 1,\ldots,P$), P is the number of available CP/land-use options for each of the three P reduction scenarios. Including keep current C-S rotation unchanged, there are six land-use options for land-use only scenario ($P = 6$), seven CP options for CP-only scenario ($P = 7$), and 12 CP or land-use options for the combined scenario. $X_{k,j} = 1$ if CP or land-use option k is implemented at field j and 0 if not, R_j^k is the annualized average per-hectare net economic returns of applying option k at field j, A_j is field j's area in hectares, and NP_j^k is edge-of-field TP or DRP discharges for option k at field j.

We solved the optimization problem using the A Mathematical Programming Language (AMPL) software (Fourer et al., 1990) and the CPLEX solver. AMPL provides a syntax close to natural algebraic expressions so that optimization problems can be formulated using familiar mathematic notations. The CPLEX solver is an effective solver for mixed integer programming problems.

Assuming a C-S rotation for all baseline agricultural fields, average agricultural TP and DRP loads for the 2001–2010 period were 379.4 metric ton (t)/year and 34.5 t/year, respectively, estimated using the SWAT model. Based on simulated yields and reported market prices and production costs, baseline net return was estimated to be about $122 Million (M)/year.

2.6. TP and DRP Abatement Costs

Effects of changes suggested by the spatial optimizations on individual landowner economic returns may vary substantially, and certain landowners may lose profit as a result of the suggested changes. If a government policy sought to ensure that no individual landowner would be worse off, financial support may be required to compensate for economic costs associated with CP implementation or a change in land use. We calculated the amount needed to compensate landowners for losses in terms of the minimal P-abatement cost. For the CP options, implementation cost is calculated as reduction in revenues as a result of yield decrease, if any, along with CP implementation and maintenance costs. For alternative land-use options, P-abatement cost is calculated as any reductions in net returns due to conversion from agricultural land with corn-soybean rotations to other options (see *SI Text: Economic Returns to Alternative Land Uses* for more details).

3. Results

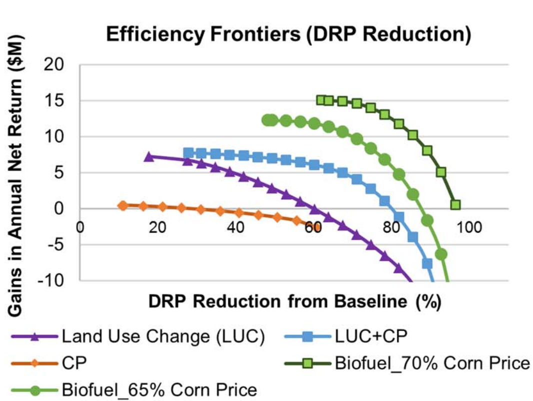
3.1. Phosphorus Reduction Efficiency

For TP and DRP reductions, Pareto efficiency frontiers (Fig. 2) generated based on 10-yr (2001 to 2010) average performance show that there is a clear tradeoff between P reduction outcomes and economic returns. Patterns of these frontiers are also quite different across scenarios. Some points along the frontier show a negative gain in economic returns, i.e., a reduction in economic returns. If the efficiency frontier associated with a P reduction strategy has greater economic returns relative to another strategy for the same set of P reduction targets, we considered it to be as relatively more efficient, or a potential Pareto improvement. Starting from any point on each frontier, the negative slope along the frontier indicates that further reductions in P loading require a loss of economic return from the sale of agricultural products, offset by economic returns from gains in timber harvests or residential land modeled via hedonic pricing. However, when compared to baseline conditions, we found that optimization of changes in either land use or CPs could, in fact, increase watershed-level economic returns for low to medium (< 40%) P-reduction objectives. Reduced fertilizer cost is the major contributor to increasing profits in the CP scenario, because farmers generally tend to over-apply fertilizer (Sheriff, 2005) so reducing 20% fertilizer does not reduce yields in some fields. Positive economic gains associated with the land-use optimization strategy are achieved by replacing C-S rotation with more competitive options in suitable fields.

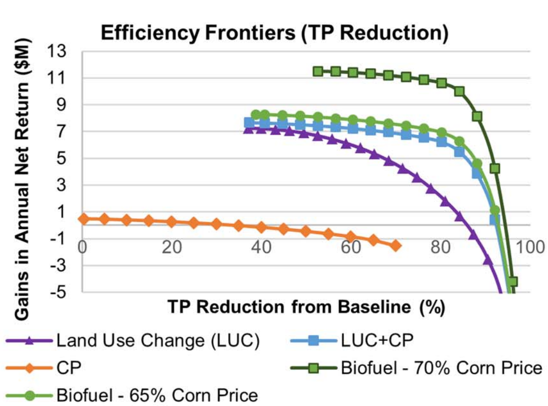
Overall, land-use optimization tended to be relatively more efficient than relying on CPs under goals for either DRP or TP reductions, i.e., land-use changes generated higher economic returns than CPs given the same level of P reduction. Differences in efficiencies between land-use and CP scenarios tended to narrow as the P-reduction target increased. Perhaps more importantly, while CPs can achieve the policy goal on TP reduction (~46%), CPs could only achieve around a 60% reduction of DRP (Fig. 2). Further reductions would require more aggressive fertilizer control and implementation of multiple CPs in the same field (Fig. S2a). However, we included these options in the optimization model, and results indicated that it is still not possible for CPs to reduce DRP > 68% (Fig. S2a). On the other hand, land-use optimization could easily achieve 80% or higher DRP reductions (Fig. 2a), although such levels of reduction come with reduced economic returns because cropland needs to be replaced with less profitable land uses such as working forest and grassland (Fig. 3a).

Over most of the range in P reduction, the gradient of the frontiers based on land-use optimization (LUC and biofuel frontiers in Fig. 2) were generally steeper than those from the CP optimization. Between any two points on the frontier, differences in net returns can be considered as the marginal cost (or incremental cost) of P reduction under our optimal algorithm. As we increase P-reduction targets, marginal costs associated with land-use optimization increased faster than those for CP optimization, which indicates that economic performance of the land-use optimization scenarios is more sensitive to the P-reduction objectives.

Substantial improvements in efficiency were attained when we included both CPs and alternative land uses as optimization options at each field. Improvements in efficiency in this combined scenario were not simply additive. Unlike the differences between the frontiers generated by the optimizations that used only land-use options or only CP options, additional gains achieved via a combined strategy increase with P-reduction targets, and the difference peaks around 70% or 85%



(a)



(b)

Fig. 2. Efficiency frontiers for (a) DRP reduction and (b) TP reduction based on the CP targeting scenario, the land-use change scenario, and a combination of both scenarios. The point of origin (0, 0) represents baseline annual profit ($122M) and P load levels for the 2001–2010 period. Biofuel scenarios assume switchgrass price per metric ton dry matter could reach 65% and 70% of that for corn grain price, respectively. The CP frontiers are not truncated because, in some fields, reducing 20% fertilizer does not reduce yield, so savings on fertilizer cost can improve net returns and P reduction simultaneously. Land-use based frontiers (LUC and biofuel) are not truncated and presented higher initial net returns than the baseline condition because we assume corn-soybean rotation at all cropland as the default, but other less fertilizer intensive land-use types (e.g. alfalfa) can be more profitable at certain fields when government payments to row crops are excluded.

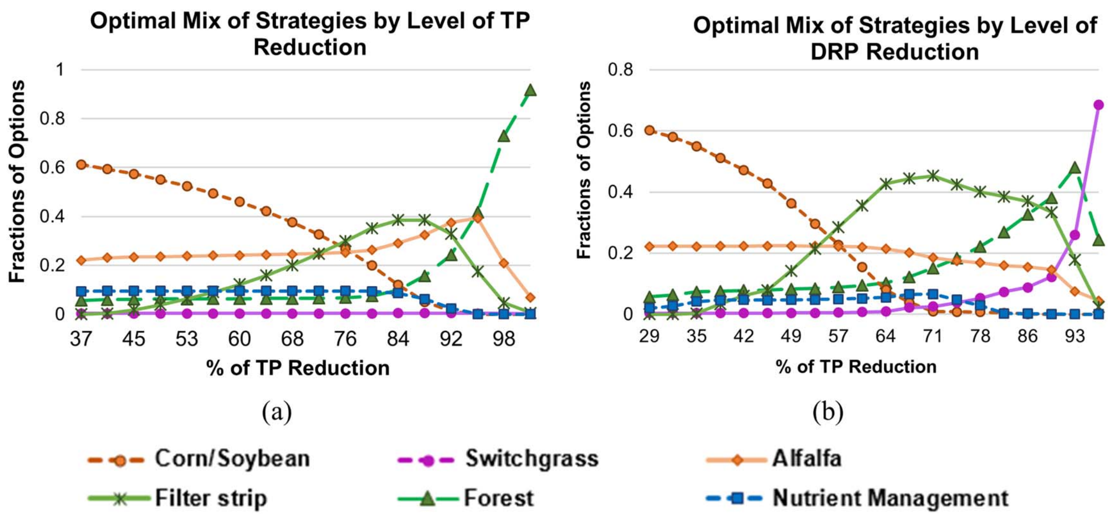


Fig. 3. Optimal mix of strategies by level of (a) DRP reduction and (b) TP reduction based on combined optimization of CP and land-use scenarios. Fraction of option means the fraction of total cropland that adopts a particular land-use or CP type, calculated as the ratio of field land-area in a certain CP or land-use option to total cropland area. Only six CP and land-use options are presented here because all other options have a fraction < 0.01 for TP or DRP reductions.

for DRP or TP reductions. The advantage of the combined scenario over land-use optimization alone diminishes for higher P-reduction targets, because nearly all CPs are replaced by land-use options to achieve the greater nutrient reductions (Fig. 3). The results reveal the importance of adopting multiple strategies for reducing NPS pollution, especially when significant nutrient reductions are needed to restore impaired water bodies. For instance, CPs are not sufficient to meet the 78% DRP-reduction target; land-use optimization could achieve the reduction, but would result in large losses (~$7M) in annual economic return. The combination scenario, on the other hand, can achieve the target level of P reduction while actually increasing annual economic return slightly (~$1M) in our simulations. This is because the combined strategy allows the optimization algorithm to establish CP on agricultural lands that are the most productive and profitable, while achieving greater nutrient reduction through land use change on less productive land or soils with poor drainage capacity where CPs have poor P reduction performance.

Another important finding is that the optimal land-use patterns to achieve targets for TP reduction do not satisfy the targeted level of DRP reduction. Under the plan generated using the combination scenario based on a targeted TP reduction of 46%, DRP would be reduced by 29.3%, which is far from the 78% target for DRP reduction. On the other hand, optimal land-use patterns generated based on the DRP-reduction objective could achieve an 82% reduction in TP. This result confirmed the need to adjust conservation practices to reflect the fact that DRP is the primary concern.

3.2. Biofuel Feedstock Scenarios

Switchgrass is a potential bioenergy feedstock for cellulosic ethanol production (Chung et al., 2014; Schmer et al., 2008). Based on current market prices, switchgrass (marketed as grass hay) is not an economically competitive land-use option, even on marginal land. However, under DRP-reduction targets of at least 80%, switchgrass becomes a competitive option. Switchgrass is a type of cellulosic or second-generation ethanol feedstock. Although industrial-scale cellulosic production technology is not mature at this time, advances in technologies (Chung et al., 2014) indicate that it could be economically competitive in the future. We included two price scenarios for switchgrass as a bioenergy feedstock. Previous studies found that the energy content of switchgrass is about 80% to 90% that of corn grain/stover (Mani et al., 2004; Schmer et al., 2008). If the price of per unit dry matter of switchgrass reaches 65% or 70% of the corn grain price, a substantial improvement in land-use efficiency could be achieved. For instance, assuming a switchgrass price that is 70% that of corn, the watershed could markedly increase annual profits (~$15M) and reduce DRP by 60% simultaneously (Fig. 2a), partly because we assume only nitrogen fertilizer is applied for switchgrass (USDA NRCS, 2009).

3.3. Compositional Change and Spatial Pattern

Fractions of the cropland of the watershed in various CP and land-use options varied significantly and non-linearly across both DRP and TP reduction objectives (Fig. 3). For moderate P-reduction objectives (< 50%), alfalfa hay, vegetative filter strips, nutrient management (20% fertilizer reduction), and replacing agriculture with forest were the top four options for both TP and DRP reductions. For greater TP and DRP reductions, broader forest coverage is necessary to reduce P loss from agricultural land. About half of the fields need either a CP or a land-use alternative to achieve 46% TP reduction. To achieve the 78% DRP reduction target, nearly all fields need CPs. The top four options selected to achieve this target were vegetative filter strips (39.8%), forest (23.3%), alfalfa hay (16.8%), and 20% fertilizer reduction (12.8%). Switchgrass also became a competitive option (5.8%), and played a vital role for DRP reductions of 80% or more (Fig. 2), because filter strips have limited effect for high DRP reductions, and alfalfa hay requires more fertilizer input than switchgrass. Unlike field crops, switchgrass does not require tillage and can grow with little fertilizer application. Residential land has a small fraction because the area of new residential land was limited to be < 10% of current urban land area. This limitation is imposed because of the low rate of growth in urban land area (3%) from 2001 to 2011, signaling weak demand.

The spatial distributions of changes in CPs and land use for both TP and DRP reductions indicated that options were not randomly distributed across the landscape. Instead, fields receiving a given option tended to form spatial clusters, largely determined by similar soil and terrain characteristics. Areas receiving alfalfa hay, forest, switchgrass, or nutrient-reduction options are mostly fields with relatively steep slopes (> 5%) (Fig. S1c) or poor soil drainage capacity (Fig. S1d). For instance, in our optimization simulations, alfalfa hay and nutrient management became concentrated in the southern part of the watershed, and conversion to forest tended to occur in the northwest, because these areas need tile drainage (Fig. S1d) and CPs are not very effective for reducing P loss via tile flow. (Lemke et al., 2011; Smith et al., 2015a) The center of the watershed remained in crop production to reduce TP by 46% (Fig. 4a), but most fields would need vegetative

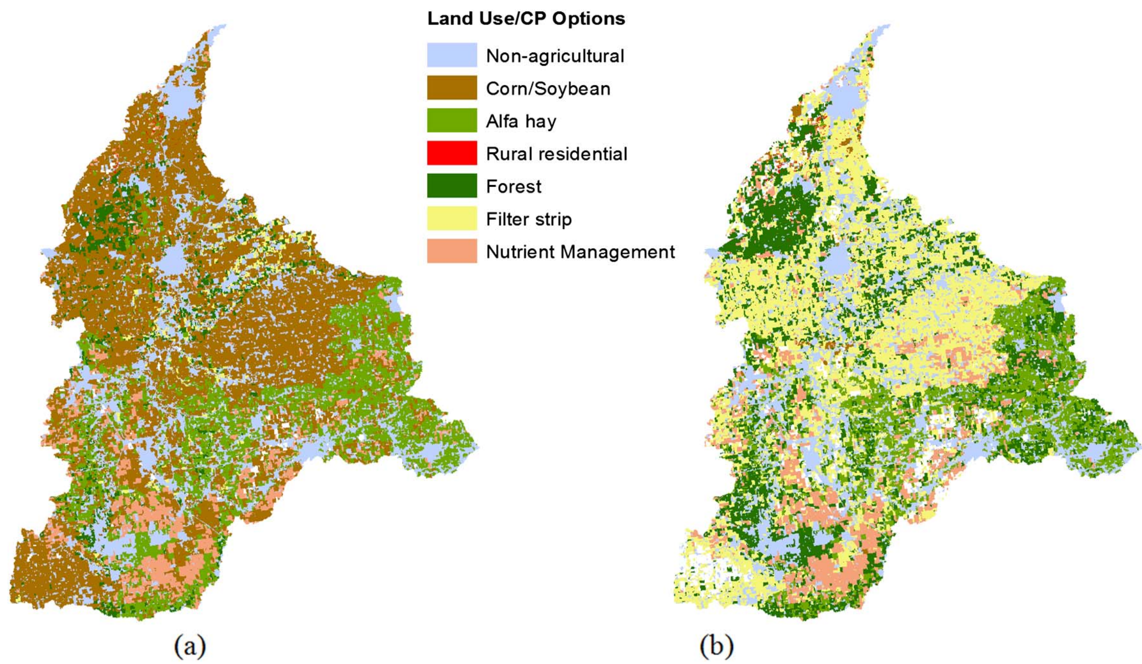


Fig. 4. Spatial distribution of CP and land-use strategies based on combined optimization of CP and land-use change with (a) the 46% TP reduction and (b) the 78% DRP reduction from baseline targets, respectively.

filter strips to reduce DRP by 78% (Fig. 4b). Under the assumption that the price of switchgrass is 70% of corn price, switchgrass tended to replace forest and alfalfa hay in much of the western part of the watershed (Fig. S4), because this scenario makes switchgrass more profitable than timber production.

Efficiency improvements achieved by land-use optimization relied heavily on alfalfa hay and forest to replace corn-soybean rotations. We tested the sensitivity of the land-use optimizations to price changes in these two land-use types. If the price for alfalfa hay or timber decreased by 20%, land-use optimization could remain relatively more efficient than CP optimization, but CP placement would be more efficient for medium DRP reductions when prices for alfalfa hay and forest decrease > 10% simultaneously (Fig. S2b). However, since CPs alone cannot meet the desired DRP reduction target, mixing of both scenarios is still needed. In fact, the combined strategy still improves on land-use efficiency considerably, especially for high DRP reductions (Fig. S2b).

3.4. TP and DRP Abatement Costs

On average, it seems that implementation of CPs does not substantially affect net returns of corn-soybean production, but converting corn-soybean to other uses like timber production may lead to considerable revenue loss (Table 3 and Fig. S3). Based on the combined scenario, we found that total abatement costs for 46% TP and 78% DRP reductions would be \$1.17M and \$5.82M annually, respectively (Table 3). To reduce TP by 46% based on CPs alone would cost \$1.98M. Since CPs cannot reduce DRP by 78%, we calculated abatement costs for a 61% DRP reduction. Abatement costs for 61% reductions under the CP and combined scenarios would be \$3.13M and \$2.32M, respectively. Even if prices for alfalfa hay and timber drop by 20%, the combined strategy saves on abatement costs (Table S1). To achieve 78% DRP reduction based on the switchgrass biofuel feedstock price scenario, if government payments would make up the difference between grass hay price (\$95/t) and 65% of corn grain price (\$115/t), then an additional \$2.3M (Table S2) would be required. Under the 70% price scenario for switchgrass feedstock (\$124/t), an additional \$14.3M (Table S3) would be required.

4. Discussion

Using the Sandusky River watershed in the Lake Erie basin as an example system, our results indicated that relying on traditional agricultural CPs alone is neither sufficient nor the most economically efficient strategy to meet the aggressive DRP reduction targets set by the

Table 3
TP and DRP abatement costs by CP and land use option (78% DRP reduction).

CP or land use option	Total cost (\$Million)		Unit cost (\$/acre)		Area of fields with profit loss (km^2)		Area of fields adopted (km^2)	
	46% TP[a]	78% DRP[b]	46% TP	78% DRP	46% TP	78% DRP	46% TP	78% DRP
Filter strips	0.072	1.4	4.3	5.2	67.6	1083.9	67.6	1083.9
− 20% fertilizer	\$25	0.18	0.04	5.7	2.4	123.9	234.7	347.6
Alfalfa hay	0.017	0.04	1.8	3.5	39.7	50.5	641.6	455.6
Forest	0.006	2.68	2.3	31.9	11.6	340.1	168.1	631.4
Switchgrass hay	\$535	1.43	1.3	39.9	1.6	145.5	11.4	157.1
CRP	0.008	0.08	44.2	44.4	7.3	7.5	7.3	7.5
Cover crop	0.991	–	90.8	–	44.2	–	64.0	–
Subtotal	1.17	5.82	–	–	174.4	1751.4	1194.7	2683.1

Note: abatement costs for each option vary spatially. Reported unit cost is average cost. CP = conservation practice.
[a] Achieving 46% reduction of TP loading from 2001 to 2010 baseline average.
[b] Achieving 78% reduction of DRP loading from 2001 to 2010 baseline average.

IJC to address the hypoxia problem in Lake Erie. Results based on a 10-year (2001 to 2010) simulation demonstrated that integrating land-use optimization into conservation planning can not only help overcome limitations of CPs for improving water quality, but can also improve economic returns when aggregated to the watershed scale and when the gains from timber harvests and residential land are included. In addition, we found that the optimal solutions to meet the 46% TP reduction target are far from sufficient to meet the goal for 78% DRP reduction. Results from this study reinforced the need for conservation planning to differentiate mineral or soluble nutrient from particulate nutrient pollution, especially in the Lake Erie basin where DRP is the primary concern (IJC, 2014; Scavia et al., 2014).

Our analyses illustrate the trade-offs between using CPs versus land-use strategies. Although land-use change appeared to be a Pareto improvement over CPs, note that both strategies have their limitations and performance of each option is subject to changes in local conditions. One of the advantages of CPs is that, unlike land-use conversion, the marginal cost of applying CPs does not change significantly (Fig. 2). However, leaving the current cropping system unchanged has several limitations. First, our results showed that it is generally much more difficult to achieve significant reductions in mineral nutrients with CPs. This finding is consistent with lessons learned from several watershed level experiments conducted in the U.S. (Kalcic et al., 2016; Rabotyagov et al., 2014a; Tomer and Locke, 2011). Also, recent studies found that while CPs like vegetative filter strips can be effective at containing P in the short run, accumulated labile P will gradually become a source of DRP later (Dodd and Sharpley, 2016; Jarvie et al., 2013). Furthermore, in the U.S. Midwest, the subsurface (tile) drainage system is widely adopted to support agricultural production, and recent research found that > 40% of P losses may happen via tile drainage (Smith et al., 2015b). However, traditional CPs were primarily designed to address nutrient loading to surface water and may have little impact, or even negative impact, on reducing soluble nutrients when tile drainage is present (Bhattarai et al., 2009; Lemke et al., 2011; Smith et al., 2015a). Incorporating crops like winter wheat into C-S rotation may lower P loads, but a recent experiment (Smith et al., 2015a) in the region found that multi-crop rotation has no statistically significant impact on TP and DRP loads via tile flow.

On the other hand, converting corn-soybean fields to perennial vegetation and/or less fertilizer-intensive land uses is generally more effective at reducing nutrient discharge due to a significant reduction in fertilizer application and improved soil cover. One key concern with land-use conversion is the cost. We found that, when government crop subsidies are excluded, the C-S rotation could be less profitable than alternative land uses at certain fields (Fig. S3). Therefore, replacing those C-S fields with alfalfa hay and forest would jointly improve economic and nutrient-reduction objectives. In fact, even if we account for financial loss due to land-use change, mixing CPs and land-use change is still more cost effective than CPs alone (Table S1).

Given the tradeoffs and complementarity of CPs and land-use change, a potentially more efficient approach to addressing excessive DRP load in the Sandusky watershed would be to implement CPs and land-use change jointly and in coordination. For instance, spatial distribution patterns of CPs and alternative land-use change placement (Fig. 4) reveal that there are clearly clustered patterns: alternative land-use types were predominately placed on poorly drained soils or fields with steep slope, since CPs are not very effective at these region. In the central basin where CPs like filter strips work well, broad adoption of CPs is more cost-effective. By allocating CPs and land-use change to the most suitable regions, a positive synergistic effect on economic efficiency can be generated. One of the barriers to encouraging this change is possible financial loss to growers. Even though a corn-soybean rotation may not be the most profitable option at all fields based on market price, government payments for row crops provide little incentive for farmers in the Midwest to diversify their cropping systems (Olmstead and Brummer, 2008). A potential approach to mediate this barrier is to provide some payments, such as a payment for ecosystem services program, to compensate potential net return loss caused by alternative land-use types (e.g. alfalfa and switchgrass) (Woodbury et al., 2017).

The integrated modeling approach presented in this study can serve as a flexible and useful decision-supporting tool to inform conservation planning. By integrating eco-hydrological modeling with an economic module, our method is capable of jointly evaluating physical effectiveness and economic efficiency of CPs versus land-use change based strategies for NPS reductions. The use of spatial optimization algorithms including Pareto frontiers can not only support evaluation of the trade-offs between water quality and economic objectives over a range of possible NPS reduction objectives, but also produce maps showing detailed spatial configurations of optimal solutions. When combined with scenario analysis, comparing Pareto frontiers derived across scenarios can help to evaluate trade-offs among NPS reduction efficiency and policy preferences. One limitation of the current approach, however, is that it did not incorporate landowner preferences and perceptions on various management alternatives. Ultimately, it is landowner management choices that reshape agricultural landscape (Nassauer et al., 2011). While integrated modeling approach implicitly assumes farmer's decision is based solely on profitability, many social-economic factors (e.g. farmland ownership, stewardship ethic) will also affect the decision-making process (Nassauer et al., 2011; Osmond et al., 2012). For this reason, it would be interesting for future studies to evaluate how landowner preferences that affect non-economic dimensions of utility may affect efficient solutions in a real world setting.

Moving toward sustainable environmental and land management may depend on stakeholder support (Lange et al., 2015; Osmond et al., 2012). Given the complexity of the spatial optimization approach, a common challenge is communicating to stakeholders about the integrated modeling system as a tool to support spatial planning (McIntosh et al., 2011; Volk et al., 2010). To facilitate stakeholder engagement, spatial optimization approaches need to be flexible enough to support an iterative scenario development process (Volk et al., 2010). The optimization approach developed in this study uses actual field boundaries as the basic spatial unit, which means landowners and decision makers apply their decisions to specific fields in a spatially explicit manner. Maps of specific spatial configurations associated with optimal solutions may serve as an interactive visualization tool to actively engage stakeholders in the planning process.

Similar to previous optimization studies (Chiang et al., 2014; Maringanti et al., 2009; Rodriguez et al., 2011), the nutrient discharges estimated in this study are edge-of-field nutrient loss. In reality, nutrient reductions achieved by CPs are expected to be lower at the outlet of the watershed than the sum of edge-of-field discharges, because in-stream processing may dampen the response (Bosch et al., 2014). Given these processes, our results provide a conservative estimate of the measures needed to reduce nutrient loads, and additional actions might be needed to achieve the same level of nutrient reduction at the watershed scale. We did not model in-stream processing because the current SWAT model has only a limited capacity to model in-stream water quality dynamics (White et al., 2014), and, in addition, it would require a dynamic link between the SWAT model and the optimization model. While a genetic algorithm could be utilized to find an approximate solution (Rabotyagov et al., 2010), tens of thousands of simulations would be needed to trace out efficiency frontiers (Maringanti et al., 2009; Rodriguez et al., 2011). Because a single generation took about 3 h on a 3.9 GZ PC, given the large number of HRUs to be simulated, it would take decades to trace out the efficiency frontier. Insufficient computing power given the current model configuration prohibits use of the dynamic linking strategy. In addition, previous optimization studies found that efficiency frontiers are very similar for SWAT simulations with and without in-stream process (Maringanti et al., 2009).

Success in improving water quality requires long-term commitment and coordinated effort (Osmond et al., 2012). Findings from this study highlighted the need for policy makers and scientists to work together, and they demonstrated the potential gains from innovations beyond current conservation planning strategies. Efficiency frontiers developed in this study provide insight into the efficacy and efficiency of alternative spatial land use and management strategies for addressing NPS pollution abatement. Due to the lack of data on existing CP implementation, simulation studies usually compare scenarios with and without CPs to demonstrate how CPs may help with achieving desired nutrient reduction targets. However, farmers in many watersheds have already adopted CPs on their cropland. For instance, about 12% of cropland in the Sandusky watershed may have implemented CPs beyond conservation tillage. In these cases, achieving the specific reduction targets might be even harder, and more innovative approaches are needed.

5. Conclusions

Using the Sandusky River watershed in Ohio in U.S. as an example, an integrated modeling approach is developed in this study to compare physical effectiveness and economic efficiency of alternative land management strategies aimed at reducing P loading from land. The approach links the SWAT model with a spatial optimization algorithm and an economic returns component to evaluate joint impacts on P reduction and returns for optimized land-use changes and/or implementation of CPs in the watershed. Among the three P reduction strategies evaluated in this study, we found that the strategy relying on CPs alone is neither sufficient nor the most economically efficient strategy to meet the aggressive P abatement targets set by the IJC to address the hypoxia problem in Lake Erie. Simulation results suggest that the CP-only strategy may be adequate for meeting the TP reduction target because CPs are efficient at intercepting sediments and TP includes P adsorbed on mobilized sediments. However, it is much more difficult for CPs to achieve the 78% DRP reduction target. On the other land, optimization results based on the land-use change strategy suggest that converting some corn-soybean fields to other less fertilizer intensive land uses (e.g. alfalfa, forest) simultaneously improve economic returns and water quality in this watershed. Higher economic returns are obtained because, when government subsidies are not included, alternative land uses can be more profitable than corn-soybean rotations on certain fields. Further, we found a combination of CPs and land-use change strategies can generate a positive, synergistic effect on economic efficiency in meeting key policy targets. This effect is achieved by keeping crop production on the most productive land, and replacing less productive land with alternative land uses (e.g., alfalfa, switchgrass) to achieve greater nutrient reductions.

Our results highlight the gains that could be made if policy makers and scientists work together and also demonstrate the potential gains from innovating on current conservation planning strategies. The Pareto efficiency frontiers derived in this study quantified the trade-offs between economic returns and water quality for the feasible range of P abatement targets under different land use and management strategies. Maps associated with spatial configurations of optimal solutions not only provided details on trade-offs, but produced spatial distributions for each scenario based on soil types and topography. This information can serve as a useful tool for promoting more scientifically informed deliberations among policy makers and stakeholders.

Conflict of Interest

Notes: The authors declare no competing financial interest.

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Appendix A. Supplementary Data

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References

Abbaspour, K.C., Vejdani, M., Haghighat, S., Yang, J., 2007. SWAT-CUP calibration and uncertainty programs for SWAT. In: Fourth Int. SWAT Conf. 1596–1602, http://dx.doi.org/10.1007/s00402-009-1032-4.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment part I: model development. J. Am. Water Resour. Assoc. 34, 73–89. http://dx.doi.org/10.1111/j.1752-1688.1998.tb05961.x.

Arnold, J.G., Gassman, P.W., White, M.J., 2010. New developments in the SWAT ecohydrology model. In: 21st Century Watershed Technology: Improving Water Quality and Environment, pp. 21–24.

Bennett, N.D., Croke, B.F.W., Guariso, G., Guillaume, J.H.A., Hamilton, S.H., Jakeman, A.J., Marsili-Libelli, S., Newham, L.T.H., Norton, J.P., Perrin, C., Pierce, S.A., Robson, B., Seppelt, R., Voinov, A.A., Fath, B.D., Andreassian, V., 2013. Characterising performance of environmental models. Environ. Model. Softw. 40, 1–20. http://dx.doi.org/10.1016/j.envsoft.2012.09.011.

Bhattarai, R., Kalita, P.K., Patel, M.K., 2009. Nutrient transport through a vegetative filter strip with subsurface drainage. J. Environ. Manag. 90, 1868–1876. http://dx.doi.org/10.1016/j.jenvman.2008.12.010.

Bosch, N.S., Allan, J.D., Selegean, J.P., Scavia, D., 2013. Scenario-testing of agricultural best management practices in Lake Erie watersheds. J. Great Lakes Res. 39, 429–436.

Bosch, N.S., Evans, M.A., Scavia, D., Allan, J.D., 2014. Interacting effects of climate change and agricultural BMPs on nutrient runoff entering Lake Erie. J. Great Lakes Res. 40, 581–589. http://dx.doi.org/10.1016/j.jglr.2014.04.011.

Chiang, L.-C., Chaubey, I., Maringanti, C., Huang, T., 2014. Comparing the selection and placement of best management practices in improving water quality using a multiobjective optimization and targeting method. Int. J. Environ. Res. Public Health 11, 2992–3014. http://dx.doi.org/10.3390/ijerph110302992.

Chung, D., Cha, M., Guss, A.M., Westpheling, J., 2014. Direct Conversion of Plant Biomass to Ethanol by Engineered Caldicellulosiruptor Bescii 111, 8931–8936. http://dx.doi.org/10.1073/pnas.1402210111.

Dodd, R.J., Sharpley, A.N., 2016. Conservation practice effectiveness and adoption: unintended consequences and implications for sustainable phosphorus management. Nutr. Cycl. Agroecosyst. http://dx.doi.org/10.1007/s10705-015-9748-8.

Fourer, R., Gay, D.M., Hill, M., Kernighan, B.W., Laboratories, T.B., 1990. AMPL: a mathematical programming language. Manag. Sci. 36, 519–554. http://dx.doi.org/10.1007/BF01783416.

Fry, J., Xian, G., Jin, S., Dewitz, J., Homer, C., Yang, L., Barnes, C., Herold, N., Wickham, J., 2011. Completion of the 2006 National Land Cover Database for the conterminous United States. PE & RS 77 (9), 858–864.

Gesch, D., Oimoen, M., Greenlee, S., Nelson, C., Steuck, M., Tyler, D., 2002. The National Elevation Dataset. Photogramm. Eng. Remote. Sens. 68, 5–11.

Goldstein, J.H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S., Daily, G.C., 2012. Integrating ecosystem-service tradeoffs into land-use decisions. Proc. Natl. Acad. Sci. U. S. A. 109, 7565–7570. http://dx.doi.org/10.1073/pnas.1201040109.

Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, R., Ruckelshaus, M., Bateman, I.J., Duraiappah, A., Elmqvist, T., Feldman, M.W., Folke, C., Hoekstra, J., Kareiva, P.M., Keeler, B.L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., Ricketts, T.H., Rockström, J., Tallis, H., Vira, B., 2015. Natural capital and ecosystem services informing decisions: from promise to practice. Proc. Natl. Acad. Sci. 112, 201503751. http://dx.doi.org/10.1073/pnas.1503751112.

Heidelberg University, N.C. for W.Q.R, 2015. Tributary Loading Website (URL). http://www.heidelberg.edu/academiclife/distinctive/ncwqr/data, Accessed date: 3 January 2014.

Hoornbeek, J., Hansen, E., Ringquist, E., Carlson, R., 2013. Implementing water pollution policy in the United States: total maximum daily loads and collaborative watershed management. Soc. Nat. Resour. 26, 420–436. http://dx.doi.org/10.1080/08941920.2012.700761.

Houck, O.A., 2002. The Clean Water Act TMDL Program: Law, Policy, and Implementation. Environmental Law Institute, 2000 L Street, N.W. Suite 620 Washington DC 20036 USA.

International Joint Commission, 2014. A Balanced Diet for Lake Erie: Reducing Phosphorus Loadings and Harmful Algal Blooms. Report of the Lake Erie Ecosystem Priority (URL). http://www.ijc.org/files/publications/2014IJCLEEPREPORT.pdf, Accessed date: 10 May 2010.

Jarvie, H.P., Sharpley, A.N., Withers, P.J.A., Scott, J.T., Haggard, B.E., Neal, C., 2013. Phosphorus mitigation to control river eutrophication: murky waters, inconvenient truths, and "postnormal" science. J. Environ. Qual. 42, 295. http://dx.doi.org/10.2134/jeq2012.0085.

Kalcic, M.M., Frankenberger, J., Chaubey, I., 2015. Spatial optimization of six conservation practices using Swat in tile-drained agricultural watersheds. J. Am. Water Resour. Assoc. 51, 956–972. http://dx.doi.org/10.1111/1752-1688.12338.

Kalcic, M.M., Kirchhoff, C., Bosch, N., Muenich, R.L., Murray, M., Griffith Gardner, J., Scavia, D., 2016. Engaging stakeholders to define feasible and desirable agricultural conservation in western Lake Erie watersheds. Environ. Sci. Technol. 50, 8135–8145. http://dx.doi.org/10.1021/acs.est.6b01420.

Kleinman, P.J.A., Sharpley, A.N., McDowell, R.W., Flaten, D.N., Buda, A.R., Tao, L., Bergstrom, L., Zhu, Q., 2011. Managing agricultural phosphorus for water quality

protection: principles for progress. Plant Soil 349, 169–182. http://dx.doi.org/10.1007/s11104-011-0832-9.
Kling, C.L., 2011. Economic incentives to improve water quality in agricultural landscapes: some new variations on old ideas. Am. J. Agric. Econ. 93, 297–309. http://dx.doi.org/10.1093/ajae/aaq190.
Lange, A., Siebert, R., Barkmann, T., 2015. Sustainability in land management: an analysis of stakeholder perceptions in rural northern Germany. Sustainability 7, 683–704. http://dx.doi.org/10.3390/su7010683.
Lautenbach, S., Volk, M., Strauch, M., Whittaker, G., Seppelt, R., 2013. Optimization-based trade-off analysis of biodiesel crop production for managing an agricultural catchment. Environ. Model. Softw. 48, 98–112. http://dx.doi.org/10.1016/j.envsoft.2013.06.006.
Lemke, A.M., Kirkham, K.G., Lindenbaum, T.T., Herbert, M.E., Tear, T.H., Perry, W.L., Herkert, J.R., 2011. Evaluating agricultural best management practices in tile-drained subwatersheds of the Mackinaw River, Illinois. J. Environ. Qual. 40, 1215–1228. http://dx.doi.org/10.2134/jeq2010.0119.
Mani, S., Tabil, L.G., Sokhansanj, S., 2004. Grinding performance and physical properties of wheat and barley straws, corn stover and switchgrass. Biomass Bioenergy 27, 339–352. http://dx.doi.org/10.1016/j.biombioe.2004.03.007.
Maringanti, C., Chaubey, I., Popp, J., 2009. Development of a multiobjective optimization tool for the selection and placement of best management practices for nonpoint source pollution control. Water Resour. Res. 45.
Maringanti, C., Chaubey, I., Arabi, M., Engel, B., 2011. Application of a multi-objective optimization method to provide least cost alternatives for NPS pollution control. Environ. Manag. 48, 448–461. http://dx.doi.org/10.1007/s00267-011-9696-2.
McIntosh, B.S., Ascough, J.C., Twery, M., Chew, J., Elmahdi, A., Haase, D., Harou, J.J., Hepting, D., Cuddy, S., Jakeman, A.J., Chen, S., Kassahun, A., Lautenbach, S., Matthews, K., Merritt, W., Quinn, N.W.T., Rodriguez-Roda, I., Sieber, S., Stavenga, M., Sulis, A., Ticehurst, J., Volk, M., Wrobel, M., van Delden, H., El-Sawah, S., Rizzoli, A., Voinov, A., 2011. Environmental decision support systems (EDSS) development - challenges and best practices. Environ. Model. Softw. 26, 1389–1402. http://dx.doi.org/10.1016/j.envsoft.2011.09.009.
McKay, L., Bondelid, T., Dewald, T., Johnston, J., Moore, R., Rea, A., 2012. NHDPlus Version 2: User Guide.
Menne, M.J., Durre, I., Korzeniewski, B., McNeal, S., Thomas, K., Yin, X., Anthony, S., Ray, R., Vose, R.S., Gleason, B.E., Houston, T.G., 2012. Global historical climatology network - Daily (GHCN-Daily), version 3. [WWW document]. NOAA Natl. Clim. Data Cent. http://dx.doi.org/10.7289/V5D21VHZ.
Michalak, A.M., Anderson, E.J., Beletsky, D., Boland, S., Bosch, N.S., Bridgeman, T.B., Chaffin, J.D., Cho, K., Confesor, R., Daloğlu, I., DePinto, J.V., Evans, M.A., Fahnenstiel, G.L., He, L., Ho, J.C., Jenkins, L., Johengen, T.H., Kuo, K.C., LaPorte, E., Liu, X., McWilliams, M.R., Moore, M.R., Posselt, D.J., Richards, R.P., Scavia, D., Steiner, A.L., Verhamme, E., Wright, D.M., Zagorski, M.A., 2013. Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. Proc. Natl. Acad. Sci. 110, 6448–6452. http://dx.doi.org/10.1073/pnas.1216006110.
Moriasi, D., 2015. Hydrologic and Water Quality Models: Performance Measures and Evaluation Criteria. http://dx.doi.org/10.13031/trans.58.10715.
Mueller-Warrant, G.W., Griffith, S.M., Whittaker, G.W., Banowetz, G.M., Pfender, W.F., Garcia, T.S., Giannico, G.R., 2012. Impact of land use patterns and agricultural practices on water quality in the Calapooia River Basin of western Oregon. J. Soil Water Conserv. 67, 183–201. http://dx.doi.org/10.2489/jswc.67.3.183.
Nassauer, J.I., Dowdell, J.A., Wang, Z., McKahn, D., Chilcott, B., Kling, C.L., Secchi, S., 2011. Iowa farmers' responses to transformative scenarios for Corn Belt agriculture. J. Soil Water Conserv. 66, 18A–24A. http://dx.doi.org/10.2489/jswc.66.1.18A.
Nelson, E., Polasky, S., Lewis, D.J., Plantinga, A.J., Lonsdorf, E., White, D., Bael, D., Lawler, J.J., 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. Proc. Natl. Acad. Sci. U. S. A. 105, 9471–9476. http://dx.doi.org/10.1073/pnas.0706178105.
NRC (National Research Council), 2009. Nutrient Control Actions for Improving Water Quality in the Mississippi River Basin and Northern Gulf of Mexico. National Academies Press, Washington, D.C.. http://dx.doi.org/10.17226/12544.
Ohio State University(OSU)-Extension, 2015. Ohio Farm Management Enterprise Budgets [WWW Document]. Enterp. Budgets 2001–2015 (URL). http://aede.osu.edu/research/osu-farm-management/enterprise-budgets, Accessed date: 27 May 2016.
Ohio State University(OSU)-Extension, 2016. Ohio Timber Price Reports 2003–2016 ([WWW Document]. URL). http://woodlandstewards.osu.edu/ohio-timber-price-report, Accessed date: 1 April 2016.
Olmstead, J., Brummer, E.C., 2008. Benefits and barriers to perennial forage crops in Iowa corn and soybean rotations. Renewable Agric. Food Syst. 23, 97–107. http://dx.doi.org/10.1017/S1742170507001937.
Osmond, D., Meals, D., Hoag, D., Arabi, M., Luloff, A., Jennings, G., McFarland, M., Spooner, J., Sharpley, A., Line, D., 2012. Improving conservation practices programming to protect water quality in agricultural watersheds: lessons learned from the National Institute of Food and Agriculture-Conservation Effects Assessment Project. J. Soil Water Conserv. 67, 122A–127A. http://dx.doi.org/10.2489/jswc.67.5.122A.
Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., White, D., Arthur, J., Garber-Yonts, B., Haight, R., Kagan, J., Starfield, A., Tobalske, C., 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biol. Conserv. 141, 1505–1524. http://dx.doi.org/10.1016/j.biocon.2008.03.022.
Rabotyagov, S.S., Jha, M., Campbell, T.D., 2010. Nonpoint-source pollution reduction for an Iowa watershed: an application of evolutionary algorithms. Can. J. Agric. Econ. Can. d'agroeconomie 58, 411–431.
Rabotyagov, S.S., Campbell, T.D., White, M., Arnold, J.G., Atwood, J., Norfleet, M.L., Kling, C.L., Gassman, P.W., Valcu, A., Richardson, J., Turner, R.E., Rabalais, N.N., 2014a. Cost-effective targeting of conservation investments to reduce the northern Gulf of Mexico hypoxic zone. Proc. Natl. Acad. Sci. 111, 18530–18535. http://dx.doi.org/10.1073/pnas.1405837111.
Rabotyagov, S.S., Valcu, A.M., Kling, C.L., 2014b. Reversing property rights: practice-based approaches for controlling agricultural nonpoint-source water pollution when emissions aggregate nonlinearly. Am. J. Agric. Econ. 96, 397–419. http://dx.doi.org/10.1093/ajae/aat094.
Rodriguez, H.G., Popp, J., Maringanti, C., Chaubey, I., 2011. Selection and placement of best management practices used to reduce water quality degradation in Lincoln Lake watershed. Water Resour. Res. 47. http://dx.doi.org/10.1029/2009WR008549.
Ruijs, A., Kortelainen, M., Wossink, A., Schulp, C.J.E., Alkemade, R., 2015. Opportunity cost estimation of ecosystem services. Environ. Resour. Econ. http://dx.doi.org/10.1007/s10640-015-9970-5.
Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., Venter, M., Boedhihartono, A.K., Day, M., Garcia, C., van Oosten, C., Buck, L.E., 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proc. Natl. Acad. Sci. 110, 8349–8356. http://dx.doi.org/10.1073/pnas.1210595110.
Scavia, D., David Allan, J., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloğlu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T.M., Goto, D., Han, H., Höök, T.O., Knight, R., Ludsin, S.A., Mason, D., Michalak, A.M., Peter Richards, R., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., Zhou, Y., 2014. Assessing and addressing the re-eutrophication of Lake Erie: central basin hypoxia. J. Great Lakes Res. http://dx.doi.org/10.1016/j.jglr.2014.02.004.
Schilling, K.E., Chan, K.-S., Liu, H., Zhang, Y.-K., 2010. Quantifying the effect of land use land cover change on increasing discharge in the Upper Mississippi River. J. Hydrol. 387, 343–345. http://dx.doi.org/10.1016/j.jhydrol.2010.04.019.
Schmer, M.R., Vogel, K.P., Mitchell, R.B., Perrin, R.K., 2008. Net Energy of Cellulosic Ethanol From Switchgrass 2007. pp. 21–26.
Schulte, L.A., Asbjornsen, H., Liebman, M., Crow, T.R., 2006. Agroecosystem restoration through strategic integration of perennials. J. Soil Water Conserv. 61, 164A–169A.
Seppelt, R., Voinov, A., 2002. Optimization methodology for land use patterns using spatially explicit landscape models. Ecol. Model. 151, 125–142 (doi:Pii S0304-3800(01)00455-0\rDoi 10.1016/S0304-3800(01)00455-0).
Seppelt, R., Lautenbach, S., Volk, M., 2013. Identifying trade-offs between ecosystem services, land use, and biodiversity: a plea for combining scenario analysis and optimization on different spatial scales. Curr. Opin. Environ. Sustain. 5, 458–463. http://dx.doi.org/10.1016/j.cosust.2013.05.002.
Sheriff, G., 2005. Efficient waste? Why farmers over-apply nutrients and the implications for policy design. Rev. Agric. Econ. 27, 542–557. http://dx.doi.org/10.1111/j.1467-9353.2005.00263.x.
Smith, D.R., Francesconi, W., Livingston, S.J., Huang, C. Hua, 2015a. Phosphorus losses from monitored fields with conservation practices in the Lake Erie Basin, USA. Ambio 44, 319–331. http://dx.doi.org/10.1007/s13280-014-0624-6.
Smith, D.R., King, K.W., Johnson, L., Francesconi, W., Richards, P., Baker, D., Sharpley, A.N., 2015b. Surface runoff and tile drainage transport of phosphorus in the Midwestern United States. J. Environ. Qual. 44, 495. http://dx.doi.org/10.2134/jeq2014.04.0176.
Tomer, M.D., Locke, M.A., 2011. The challenge of documenting water quality benefits of conservation practices: a review of USDA-ARS's conservation effects assessment project watershed studies. Water Sci. Technol. http://dx.doi.org/10.2166/wst.2011.555.
USDA ERS, 2015. Commodity costs and returns. Costs returns corn soybean, 2001–2015 (URL). http://www.ers.usda.gov/data-products/commodity-costs-and-returns.aspx.
USDA FSA, 2017. FSA Crop Acerage Data. 2016 Acreage Data (URL). https://www.fsa.usda.gov/news-room/efoia/electronic-reading-room/frequently-requested-information/crop-acreage-data/index, Accessed date: 8 March 2017.
USDA NASS, 2012. 2012 Census Volume1, Chapter1: State Level Data. Table 8. L. 2012 2007 (URL). https://www.agcensus.usda.gov/Publications/2012/Full_Report/Volume_1,_Chapter_1_State_Level/Ohio/, Accessed date: 3 August 2017.
USDA NASS, 2016. USDA National Agricultural Statistics Service Cropland Data Layer. Publ. Crop. Data Layer (URL). https://nassgeodata.gmu.edu/CropScape/, Accessed date: 18 May 2016.
USDA NRCS, 2009. Planting and Managing Switchgrass as a Biomass Energy Crop (URL). http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1042293.pdf.
USDA NRCS, 2010. Key Findings from the CEAP-Cropland Assessment of the Effects of Conservation Practices on Cultivated Cropland in the Upper Mississippi River Basin. Resour. Invent. Assess. Div.
USDA NRCS, 2011. Tillage Study Tables and Charts-WLEB. Table 1 - West. Lake Erie Basin Watershed Tillage Syst. 3 Major Crop (URL). https://www.nrcs.usda.gov/wps/portal/nrcs/detail/oh/technical/?cid=nrcs144p2_029581, Accessed date: 8 March 2017.
USDA NRCS, 2015. Soil Survey Geographic (SSURGO) Database for [U.S.]. United States Dep. Agric (URL). http://soildatamart.nrcs.usda.gov, Accessed date: 20 November 2015.
Valcu-Lisman, A.M., Kling, C.L., Gassman, P.W., 2016. The optimality of using marginal land for bioenergy crops: tradeoffs between food, fuel, and environmental services. J. Agric. Resour. Econ. 45, 217–245. http://dx.doi.org/10.1017/age.2016.20.
Volk, M., Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S., Gassman, P.W., 2008. Integrated ecological-economic modelling of water pollution abatement management options in the Upper Ems River Basin. Ecol. Econ. 66, 66–76. http://dx.doi.org/10.1016/j.ecolecon.2008.01.016.
Volk, M., Lautenbach, S., van Delden, H., Newham, L.T.H., Seppelt, R., 2010. How can we make progress with decision support systems in landscape and River Basin

Management? Lessons learned from a comparative analysis of four different decision support systems. Environ. Manag. 46, 834–849. http://dx.doi.org/10.1007/s00267-009-9417-2.
White, M.J., Storm, D.E., Mittelstet, A., Busteed, P.R., Haggard, B.E., Rossi, C., 2014. Development and testing of an in-stream phosphorus cycling model for the soil and water assessment tool. J. Environ. Qual. 43, 215. http://dx.doi.org/10.2134/jeq2011.0348.
Woodbury, P.B., Kemanian, A.R., Jacobson, M., Langholtz, M., 2017. Improving water quality in the Chesapeake Bay using payments for ecosystem services for perennial biomass for bioenergy and biofuel production. Biomass Bioenergy. http://dx.doi.org/10.1016/j.biombioe.2017.01.024.