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Minimizing costs of reducing agricultural nitrogen loadings: choosing between on- and off-field conservation practices

Abstract

The objective of this research is to generate and cross-compare cost-effectiveness estimates of on-field (improved timing, rate, and application method) and off-field (restoring wetlands and establishing vegetative filter strips (VFSs)) approaches to reducing reactive nitrogen loadings. The analysis is based on economic and biophysical models that we've designed to evaluate public water quality conservation strategies. Results indicate that increasing the use of on-field nitrogen conservation practices, targeted to lands in corn-corn rotation (with a nitrogen-removal cost of $0.17 per pound), is the most cost-effective nitrogen conservation strategy. But the cost of further reductions in nitrogen losses by offering incentives to farmers with a corn-soybean rotation rises to $0.62 per pound. The authors found that wetland restoration is the second most cost-effective strategy. The average nitrogen removal cost of five- to ten-acre wetlands ranges from $0.23 to $0.24 per pound – when the wetlands are located to receive aquatic nitrogen flows. Nitrogen removal by VFS, in a best-case scenario, costs $0.41 per pound, but could be much higher. The analysis and results are based on conditions within a single Ohio county, two crop rotations, and a limited number of wetland options. However, the approach used here can be applied to assess different mixes of conservation practices and to determine which conservation practices are most cost-effective.

Keywords: cost-effective conservation, agricultural nitrogen loadings, targeting nitrogen control, wetlands as nutrient filters, nitrogen BMP policies, cost-minimizing nitrogen control.
JEL Classification: Q50, Q52.

Introduction

Strong public interest in reducing nitrogen loadings and hypoxia in the Gulf of Mexico, Chesapeake Bay, and elsewhere has resulted in a variety of government actions. The United States Department of Agriculture (USDA) has initiated several conservation programs designed to, at least in part, reduce nitrogen in surface waters. Although these initiatives have had positive water quality impacts, increases in the intensity of crop and animal production and non-agricultural effluence have more than offset nitrogen conservation efforts.

In recent years, Federal Agencies have worked to increase the cost-effectiveness of attaining environmental gains using science based approaches to measure the costs and benefits of their alternative programs and strategies (Euliss et al., 2011).

The objective of this research is to generate estimates of the cost-effectiveness of alternative nitrogen-conservation strategies that can be used to better target public funding for nitrogen control. We consider three nitrogen-control practices: on-field conservation, wetland restoration, and the establishment of vegetative filter strips (VFSs). We do not know the value of the environmental benefits of reduced nitrogen loadings hence we cannot maximize benefits relative to costs. But our bio-physical and economic models allow us to evaluate the cost-effectiveness of alternative approaches to reduce the quantity of nitrogen reaching surface waters. The most cost-effective strategy is the least-cost means to attain a marginal reduction in nitrogen loadings.

Background. Three earlier analyses have examined the economics of alternative nitrogen conservation strategies. These studies use partial equilibrium models to calculate the costs of policies designed to achieve up to 20-percent reductions in agricultural nitrogen loadings. The analyses generate agricultural-sector equilibrium levels of nitrogen use and costs, where costs are calculated as the sum of the changes in farm profits and consumer incomes. Given their objective, these studies do not consider spatial differences in cost-effectiveness while our study does. Given our objective of cost-effective targeting of nitrogen conservation practices, our analysis does not examine large reductions in nitrogen loadings, while the previous studies do. Although their goals and approaches differ from ours, their findings provide a perspective of past analyses of cost-effective nitrogen reduction strategies for the Mississippi watershed.

Doering et al. (1999) estimated the cost of reducing nitrogen loadings to the Gulf of Mexico. They concluded that on-field practices alone are the most cost-effective strategy – at $0.44 per pound – to attain a 20-percent (941,000 ton) reduction in nitrogen loadings. Their findings suggest that wetlands are far less cost-effective. One million acres of wetlands was estimated to remove 67,000 tons of nitrogen at an average cost of $3.03 per pound. An important contribution of their analysis is the inclusion of the macro-economic effects that large-scale changes in land use has on agricultural output and consumer and producer surplus.

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Peters (1999) found that reducing nitrogen application rates was the least-cost means of reducing excess agricultural nitrogen in the Gulf of Mexico. A 20-percent reduction was estimated to cost $1,916 million annually. Peters does not report the quantity of nitrogen reduced or per-ton nitrogen removal cost estimates. However, assuming that Doering et al.'s (1999) estimate of a 941,000-ton reduction in nitrogen is applicable here, then Peters' cost estimates suggest that per-pound nitrogen removal costs are $1.03 with on-field reduction strategy, $4.20 with a wetland restoration strategy, and $1.11 with a mixed nitrogen-reduction and wetland-restoration strategy (with 5 million acres of wetlands restored). These nitrogen removal cost estimates are more than double those of Doering et al. (1999). Peter's analysis, as with Doering et al.'s accounts for market effects by estimating conservation practices' effects on the production costs and yields of major field crops. Then, using a partial equilibrium model, they estimate the final mix of agricultural input use, output, prices, farm profits, and consumer income. While Peters (1999) and Doering et al. (1999) use the same agricultural-sector partial equilibrium model, the source of the differences in results is not apparent, which is not surprising given the complexities of the analyses, the variety of assumptions one needs to apply, and the fact that published results cannot include all of these details.

A large-scale nitrogen reduction analysis by Ribaudo et al. (2001) found on-field nitrogen conservation practices to be more cost-effective than restoring wetlands at lower levels of nitrogen reductions. At a 5-percent reduction in nitrogen loadings, on-field nitrogen removal costs average approximately $0.25 per pound. However, if nitrogen loadings are reduced by 26 percent (1.2 million metric tons), on-field nitrogen removal cost reaches $1.80 per pound and wetland restoration becomes a cost-effective strategy. While Ribaudo et al. use the same partial equilibrium model, the complexities of this analysis, as with those discussed above, makes it difficult to determine why estimates differ.

The prior research, as with ours, examines least-cost means of reducing nitrogen loadings using both on- and off-field conservation practices. The prior research evaluates the economics of nitrogen conservation program options. The analyses consider very large changes in conservation practices, so large that they must include assessments of market impacts. Our research generates estimates of cost-effective or marginal changes in conservation practices and at a much smaller scale. The findings offer insights into how public funding of nitrogen conservation can be targeted to minimize the cost of nitrogen reductions.

The results presented here are limited to a single county but the methodology presented here can be applied to other counties and watersheds and a broader range of conservation practices as models' capabilities increase. While our economic data are extensive enough to assess costs of conservation practices across much of the corn belt, our bio-physical models are limited – most limiting is the Nitrogen Loss and Environmental Assessment Package with GIS capabilities (NLEAP-GIS) (Shaffer et al. 2010), as it has thus far been calibrated to conditions in 12 counties and a limited number of conservation practices and crop rotations (Delgado et al., 2008; Delgado et al., 2010a; Delgado et al., 2010b). Our wetland bio-physical simulation model has been calibrated to conditions throughout much of the upper Mississippi watershed. These models overlap in northwestern Ohio – Williams County.

We begin by describing the economic and nitrogen models and deriving economic measures of on- and off-field nitrogen conservation strategies. We then derive model estimates and run simulations to generate measures of the costs and nitrogen load reductions of the on- and off-field conservation practices. As a means of cross-comparing the quantitative impacts of the nitrogen conservation strategies, we generate estimate of the quantity of nitrogen removed if the practices were applied to 1,000 acres. Other than our evaluation of wetlands, the acreage need not be contiguous and transactions can involve multiple landowners. The contiguous acres of the wetland watershed need not be held by a single landowner, but a single landowner sells the wetland easement. Finally, we generate the cost-effectiveness estimates and discuss their policy implications and limits.

1. The economic and nitrogen models

1.1. Modeling the cost and effects of on-field practices. The nitrogen best management practices (BMPs) examined here are those defined by NRCS (appropriate timing, method, and application rates) (USDA, NRCS, 2010).

The cost of getting another farmer to adopt nitrogen BMPs is, essentially, the incentive payment being made now. In other words, one can view an incentive payment as the price being paid now to adopt the nitrogen BMPs hence the cost of a marginal increase in farmers' use of nitrogen BMPs. Granted, an incentive payment sufficient to one farmer may not be sufficient to another[1]. Our analysis does not need to know the payment that each farmer would

[1] One of the first early analyses of the existence of differing responses to incentive payments can be found in Cooper and Keim (1996). Findings of recent research, including the more spatially refined analysis of Watzold and Drechsler (2009), are consistent with Cooper and Keim's. Reasons that the size of acceptable incentive payments vary across farmers include the effect of the practice has on expected profits and farmers' willingness to accept some costs to improve environmental quality (Bishop et al., 2010; Chouinard et al., 2008; Babcock, 1992).

require (e.g., agriculture's BMP supply function). Instead, our analysis is based on the cost of getting an additional farmer to adopt (e.g., the marginal cost). Hence, incentive payments that have been accepting represent current marginal costs.

Nearly all USDA payments made to encourage the adoption nitrogen BMPs come through the Environmental Quality Incentive Program (EQIP). Different responses to incentives is one reason why EQIP incentives have encouraged some but not all farmers to adopt nitrogen BMPs, despite its near 20-year existence. Data used in this analysis indicate that, in Ohio, as of 2005, three nitrogen BMPs were used on 25 percent of the corn-corn acreage. We know then that use of BMPs can be expanded, but marginally so at the current price (e.g., easement payment).

Changes in nitrogen losses from adopting nitrogen BMPs are calculated using NLEAP-GIS. NLEAP-GIS simulates expected nitrogen losses based on the mix of production practices, all aspects of nitrogen use and carry-over, crop rotations, and environmental conditions. To generate reliable estimates, NLEAP-GIS must be calibrated to soils and weather conditions in each study area. The effect of adopting BMPs are determined by first estimating baseline nitrogen losses then subtracting losses that are estimated to occur after conservation practices are adopted – an approach that is similar to that described in Delgado et al. (2008) and Delgado et al. (2010a). Baseline losses are estimated by running twenty-year simulations of NLEAP-GIS under the current mix of production practices and environmental conditions in our study area. Nitrogen losses after nitrogen BMPs are adopted and estimated by running 20-year simulations with the BMP practices in use. Twenty-year simulations generate estimates of probable or expected losses that embody normal variations in weather.

1.2. Modeling the cost and effects of wetlands. Payments made to restore wetlands on farmlands, as with the on-field incentives, provide a means of reducing nitrogen losses. USDA's Wetland Reserve Program (WRP) is the largest wetland restoration program in the United States. The WRP, introduced in the 1990 Farm Bill, is a voluntary program designed to restore and protect wetlands on private agricultural lands. The program offers financial incentives to enhance, protect, and restore wetlands to their original (if possible) hydrologic and ecological conditions. Through the WRP, USDA purchases 10- and 30-year and permanent easements on agricultural lands. Because most easements – approximately 87 percent – are permanent, we focus our attention on these.

The total cost of a new wetland is the sum of the cost of the land and the cost of restoring the wetland's hydrology and ecosystem. The cost of land – the minimum-sized incentive that would get a landowner to accept a wetland easement – is, primarily, the difference between the land's value with and without wetland. The WRP requires that easement payments not exceed the difference in the land's agricultural value and its value after the wetland is restored. The WRP is competitive – landowners who underbid are likelier to be accepted into the program. While it is readily apparent that the agricultural value of the land is a function of its agricultural profits, the value of the land with the wetland easement is more difficult to measure and has likely been a challenge to calculate and incorporate into WRP easement payments.

We calculate the agricultural value of land by multiplying the county-average farmland rental rate by the size of the easement.

The value of land with a wetland depends on the profit potential from the sale of marketable wetland amenities (for example, the sale of access rights to hunters and birders) and its remaining agricultural value (land can be pastured if the wetland ecosystem is not damaged). The demand for marketable amenities increases with the size of the surrounding population (Mitsch and Gosselink, 2000; Thibodeau, 1981).

The value of the land with a wetland also depends on the intrinsic value of the wetland to the owner. Several studies provide strong evidence that wetlands have positive effects on land values (Reynolds and Regalado, 2002; Earnhart, 2001; Mahan et al., 2000; Doss and Taff, 1996). As with marketable amenities, the demand for wetlands for their intrinsic value increases with size of the surrounding population.

The cost of restoring a wetland depends on the cost of restoring its hydrology and vegetative cover. Hence, restoration cost is likely to be dependent on the type of restoration and the size of the wetland.

The cost of restoring a wetland depends on the size of the wetland and the type of restoration activities required. We know the size of each easement. We do not know exactly what types of restoration activities were performed. Hence, to minimize variation in the type of restoration activities used, we confine our sample to observations in the Glaciated Inland Plains (GIP) (Figure 1) where wetlands have similar hydrology and ecology. The GIP includes major parts of Minnesota, Wisconsin, Michigan, Iowa, Illinois, Indiana and Ohio. This region is a large contributor to the nitrogen loads in the Mississippi River and the Gulf of Mexico (Turner and Rabalais, 2003; Mitsch and Day, 2006; Goolsby et al., 2001). Finally, as with any construction project, the cost of restoring wetlands may be affected by economies of scale.

Given the above and the available data, we express the total cost of a wetland as:

$WC = g(AgrValue, Acres, Acressq, Rural, Fsize)$,

where *WC* is the total (land + restoration) cost of a wetland, *AgrValue* is the land's agricultural value (the size of the easement (in acres) multiplied by its rental value ($/acre)), *Acres* is the size of the easement (in acres) and *Acressq* is the acreage squared. *Rural* is a dummy variable that equals one when the land is in a rural area. We also expect landowners with less land to need a greater incentive hence have included *Fsize* which is the county-average farm size (in acres).

The WRP contract data file includes the total cost of the wetland (*WC*), contract acres (*Acres*), and a county identifier[1].

The county identifier allows us incorporate three county-level variables. The first is the county dry-land farmland rental rate reported by the National Agricultural Statistics Service (NASS). NASS generates the rent estimates from various sources, including the recently-initiated county-level cash rents survey (USDA, NASS, 2009).

The second file has Population-Interaction Zones for Agriculture (PIZA) scores. The PIZA data classify counties as rural or three levels of urban (Documentation and PIZA data are available at: http://www.ers.usda.gov/data/populationinteractionzones/discussion.htm). For each county, *Rural* equals one if there is no urban pressure and zero otherwise. The third file has the county-average farm size (*Fsize*), from the agricultural census.

The practice of strategically placing wetlands across the landscape to remove nitrogen has been referred to as nutrient farming (Hey et al., 2005; Berry et al., 2003; Berry et al., 2005; and Tomer, 2010). To be effective, the wetland must be placed at the bottom of a wetland watershed that is large enough to supply enough water to maintain a healthy ecosystem and has nitrogen loadings that are of concern. Given the climate and agricultural practices in northwestern Ohio, a 1,000-acre watershed is very likely to support a healthy wetland ecosystem and carry a substantial quantity of nitrogen. Also, a 1,000-acre wetland watershed in this area of the country is not atypical. While other sized wetland watersheds might have been considered (and should be in future analyses), for the sake of brevity, we have not done so here. We do, however, examine the cost-effectiveness of different sized wetlands because the size of the wetland affects both total cost and the quantity of nitrogen removed.

The quantity of nitrogen a wetland removes depends on, among other things, the volume of water passing through, the concentration of nitrogen, the time water spends in the wetland, weather and climatic conditions, and the wetland size. These are interactive factors – that is, the marginal effect of one is dependent on the values of others (Crumpton et al., 2008). Of particular interest here is how restoring wetlands of various sizes affects the quantity of nitrogen removed.

Analysts' understandings of wetland functions have progressed to a point that performance forecast models are now emerging for nitrogen and phosphorous removal. For wetlands receiving significant nitrate loads, loss rates can be described by a temperature dependent first-order model (Crumpton et al., 1997; Crumpton and Goldsborough, 1998; Crumpton, 2001; Crumpton et al. 2008; Crumpton et al., in preparation; Kadlec and Knight, 1996; Kadlec and Wallace, 2008):

$J = k_{20} * C * \theta^{(T-20)}$,

where J is the area-based nitrate-N loss rate, g N m^{-2} day^{-1}; k_{20} is the area-based first order loss rate coefficient for nitrate-N at 20°C, m/day; C is the concentration of nitrate-nitrogen, g N m^{-3}; θ is the temperature coefficient for nitrate-N loss; T is the water temperature, °C.

Wetlands receiving nonpoint source loads are subjected to widely varying hydraulic and nutrient loading rates and cyclic variations in other forcing functions such as temperature. Mass balance models, such as ours, must adequately represent the variations in load and residence times for conditions at each site. Tanks in series (TIS) models have been shown to realistically represent the residence time distributions of wetlands having a wide range of morphometries and aspect ratios (Kadlec and Knight, 1996; Kadlec and Wallace, 2008). For the current study, the reaction rate expression above was incorporated into a TIS mass balance model of nitrate loading and loss to estimate the variability in performance that would be expected due to variability in temperature and loading patterns. Model coefficients were estimated as $k_{20} = 0.15$ and $\theta = 1.09$ based on mass balance measurements from experimental wetlands and wetland mesocosms (Crumpton et al., 1993; Crumpton et al., 1997; Crumpton, 2001; Crumpton et al. in preparation; Kadlec and Wallace, 2008). Inflow rates for wetlands of different sizes were estimated based on water yield for the Tiffin River at Stryker OH (USGS 04185000). Wet-

[1] The WRP data are agreements so the reported costs may differ from actual costs. We assume that the preliminary agreed-upon costs are reasonable approximations of the final costs thus expect appropriate model estimates.

land water temperature was estimated as the average of minimum and maximum air temperature for Defiance, OH constrained to > 0 ^{o}C (National Climatic Data Center, NNDC Daily Climate Data Online, Defiance, COOPID 332098). Our nitrogen-removal estimates were generated using 20-year simulations in order to capture expected variations in precipitation and rainfall intensity and their subsequent impact on nitrogen removal.

1.3. Modeling the cost and effects of vegetative filter strips. The cost of using VFS is the sum of the cost of the land and the cost of establishing the vegetative cover. The approach we used to determine these costs is the same as we used in determining the cost of on-field conservation practices. That is, we look at a conservation incentive payment as a cost to the government and a price paid to program participants. The market price represents the cost of a marginal increase in program participation. Any cost-share, borne by the program participant, is not a public cost hence should not be include in analyses of cost-effective targeting of public incentive payments. We use data from two USDA programs (EQIP and the Conservation Reserve Program (CRP)) to generate VFS cost estimates. Both programs compensate landowners for the opportunity cost of their land and a share of the cost of establishing cover.

VFSs remove nitrogen from field runoff and subsurface flows. VFS removes nitrogen from field runoff by catching and holding sediment and slowing nitrate flows which allows greater plant uptake and denitrification. Though there is an overall agreement among scientists that VFSs are effective in reducing nitrogen runoff, there is debate over their most effective design. Their effectiveness depends on the type and condition of the vegetative cover, sediment and nutrient concentrations, quantity of runoff flows, temperature, size of the strip, and other factors – which vary within fields and across the country. Rapid rates of runoff due to heavy rains will decrease VFS effectiveness. An uneven distribution of flow across a VFS leads to rapid rates of runoff along portions of the VFS and thereby decreases effectiveness. A VFS can be ineffective if all runoff flowed across one point (Mayer et al., 2005).

We used a model developed by White and Arnold (2009) to estimate nitrogen removal rates (the percent of nitrogen removed by) of VFS. While we do not have the model and data needed to directly estimate nitrogen removal rates, we can draw estimates based on their Figure 10. In this figure, White and Arnold (2009) plot expected nitrogen removal rate as a function of the field-to-VFS ratio, under four scenarios. Each scenario assumes a different distribution of the flow of field runoff across the VFS. As one would expect, removal rates are lower when the runoff passes over a small portion of the VFS.

The quantity of nitrogen removed by a VFS is the product of the estimated nitrogen removal rate and the quantity of nitrogen runoff. We use NLEAP-GIS to generate estimates of the quantity of nitrogen in surface runoff.

VFSs also remove subsurface nitrogen flows and, commonly, most nitrogen leaving agricultural fields reaches streams through subsurface flows. VFSs have been found to have very high nitrogen removal rates from subsurface flows, but only under certain conditions – primarily, when subsurface flows pass through the root zone. Additionally, factors such as high rates of water movement (especially through tile drains) and low soil temperatures reduce the effectiveness of VFSs. Unfortunately, these factors are highly variable (within and across fields) and not readily observed, making it difficult to calculate VFS effects without on-site measurements.

Recognizing these challenges, Mitsch et al. (1999) tabulated several plot studies with a focus on the quantity of nitrogen removed from subsurface flows. Their goal was to measure the effectiveness of VFSs with forested and grass cover. However, the scope of their analysis was limited by the number of plot studies. They were able to draw statistically-significant results on the effectiveness of forested VFS.

Based on the plot studies used, Mitsch et al. (1999) model parameters capture the effectiveness of 'ecologically engineered' VFSs (that is, designed for effective nitrogen removal). An important aspect of their model, as it relates to our analysis, is that it generates estimates of the quantity of nitrogen removed per acre of a VFS, not removal rates. They assume nitrogen flow rates typical of those in corn-producing areas. They do not report a field-to-VFS ratio. Hence, we cannot examine the effectiveness of VFSs' across changes in the size of the upland area or nitrogen inflows. Instead, their estimates allow us to calculate the quantity of nitrogen removed solely as a function of the size of the VFS.

2. Data and model simulations

2.1. Data and model simulations of on-field nitrogen conservation. The cost of getting another farmer to adopt is the payment being offered. EQIP incentive payments for adopting nitrogen BMPs (practice code 590) within Ohio in 2010 ranged from $5.00 to $40 per acre (http://www.oh.nrcs.usda. gov/programs/eqip2010.html#Payment_Schedule). We have chosen to apply the low EQIP incentive payment ($5.00) as the per-acre cost of adopting all BMPs. Should we be understating costs, we will be overstating cost-effectiveness. We will revisit this issue when comparing estimates of cost-effectiveness.

We calibrated NLEAP-GIS to climate and weather conditions in Williams County and for the area's two dominant soil types (sandy and clay), and for lands with and without tile drainage. We account for the 'legacy' level of soil nitrogen by running NLEAP simulations for the prior 20 years, assuming the current crop rotation. Estimates of nitrogen losses under new conservation practices are the average values generated by 20-year NLEAP simulation runs. The 20-year simulations allow us to generate expected-value estimates that account for the range in the area's possible weather conditions such as heavy rains and drought. We acknowledge that there could be a lag time while one practice is changed and results are seen (Owens et al., 2008).

Because of current resource constraints, we were only able to calibrate NLEAP for two crop rotations – corn-corn and corn-soybeans. This shortcoming may not be too significant, given that Williams County lies in a region dominated by corn and soybean production and corn is the single largest source of nitrogen loadings in US waters. A major portion of the Gulf of Mexico's hypoxia zone can be attributed to nutrient losses in the Corn Belt, home of Williams County.

To derive base-line estimates of nitrogen losses, we ran NLEAP-GIS for the current mix of nitrogen management practices. County-level data on nitrogen management practices are not available but state-level data are. So we assume that the county's current mix of production practices is equivalent to those observed at the state level. The 2005 Ohio ARMS data are the most recent and are applied here. The data indicate that, of the lands in corn-corn and corn-soybean rotations, 16.8 (83.2) percent is in a corn-corn (corn-soybean) rotation. These statistics indicate that, on 1,000 acres of farmland, one would expect to find 168 acres in a corn-corn rotation and 832 acres in a corn-soybean rotation. Also, in any one year, one would expect to find 584 acres of corn and 416 acres of soybeans.

The ARMS data also indicate that all three nutrient BMPs are used on 31 percent of all corn acreage, 25 percent of the land in corn-corn rotation, and 32 percent of the land in corn-soybean rotation (table 1) – which means that, if nutrient BMPs are to be used on all of these lands, at least one nutrient BMP will need to be adopted on 69 percent of all the corn acreage, 75 percent of the corn-corn acreage, and 68 percent of the corn-soybean acreage. No nutrient BMPs are used (hence all three will need to be adopted) on 10 percent of the total corn acreage, 51 percent of the corn-corn acreage, and 6 percent of the corn-soybean acreage. The remaining acreages had one or two nitrogen BMPs.

Looking across all corn acres, 70 percent have a BMP application rate, 62 percent have a BMP timing, and 53 percent use a nutrient BMP method (Table 2) – which means that 30, 38, and 47 percent of the land does not have a BMP rate, timing, and method, respectively. A BMP rate, timing, or method is not used on 60, 65, and 67 percent of the corn-corn acreage, respectively. Similarly, a BMP rate, timing, and method is not used on 37, 36, and 45 percent of the corn acreage. Given these baseline conditions, one or more BMPs will need to be adopted on 12-75 percent of the total – corn-corn acres and 287-69 percent of the total – corn acres (Table 1).

ARMS data do not include a measure of soil type nor do they indicate the availability of tile drainage. Therefore, we run NLEAP for each of the four possible scenarios: sandy and clay soils with and without tile drainage. The BMP application rate for land in corn-corn rotation is 132 pounds per acre, 116 on no-till lands, and 30 pounds less for lands in corn-soybean rotation, as defined in USDA, HRCS (2010).

Under baseline conditions, NLEAP-GIS simulations indicate that 14,000 to 27,000 and an average of 20,000 pounds of nitrogen are leached from the 1,000 acres of farmland each year (Table 3). Sandy soil with tile drainage systems has the highest leaching rate.

If all nitrogen BMPs are employed on the entire watershed, nitrogen leaching is estimated to fall 8,900 pounds per year. If only the nitrogen BMP application rate or the correct timing was adopted, nitrogen leaching would fall 5,150 to 5,750 pounds per year – 60 to 70 percent –which is consistent with findings of earlier research (Delgado and Bausch, 2005; Meisinger and Delgado, 2002; Delgado et al., 1996; Bock et al., 1991).

But the adoption of the BMP method increases leaching. This response reflects the fact that, while switching from broadcasting without incorporation reduces ammonia volatilization and nitrogen runoff and increases yields, it also increases the nitrogen available for leaching. Incorporation has other advantages, such as reducing soil erosion and phosphorus runoff (Claassen et al., 2001; Ribaudo et al., 2001). We've assume that, because of these advantages, from a policy perspective, incorporation would not be dropped from the set of nitrogen BMPs.

The pattern of nitrogen leaching on land with a corn-corn rotation is not inconsistent with what we see on all corn acreage (Table 3). But about 40 percent of all nitrogen leaching occurs on the 17 percent of the land that is in a corn-corn rotation (Tables 2 and 3). The high nitrogen loss on corn-

corn lands is in agreement with Meisinger and Delgado (2002). We, therefore, consider the effects of a 'targeting' approach where conservation practices are initiated only on lands in a corn-corn rotation. This will tend to give higher cost-effectiveness estimates – we will revisit this assumption.

2.2. Data and model simulations of costs and nitrogen removal by wetlands. We estimate the wetland cost model using ordinary least squares (OLS) and 3,636 WRP observations in the GIP. Variable coefficients are statistically significant with the expected signs (Table 4). The adjusted R-square indicates that the model explains 86 percent of the variation in WC (total wetland cost) and thus our cost model is likely to provide reasonable predictions of wetland costs.

The coefficient on *AgrValue* appears reasonable. The coefficient essentially, capitalizes the value of the land based on current land rent. That is, a one-dollar increase in the annual profitability of the land (all else being equal), will increase the wetland cost by $12.30. The magnitude of this coefficient is largely driven by landowners' expectations of the future value of the land (or future rental income), their implicit discount rate, and risk aversion. If there is no expectation of land value increases, then the coefficient suggests that landowners' implicit discount rate is 8.1 percent. If land values were expected to increase by 3 percent annually, the implicit discount rate would be 5.1 percent.

The coefficient on *Acres* indicates that a one-acre increase in the size of a wetland increases restoration costs by $1,070. The significant coefficient of *Acressq* supports our hypothesis of economies of scale in wetland restoration. Finally, our estimates suggest that easements cost about $350 less in rural areas. Urban proximity is thought to increase the value of land with and without wetlands. The negative sign of the coefficient of *Rural* indicates that, in net, the effect on land values without wetland dominates, but by a relatively small amount. Wetland easements on larger farms are likely to cost less – though not much. A 100-acre difference in farm size appears to change wetland costs by $74.

The estimate cost model projects the total cost of 5- to 30-acre wetlands in Williams County to range from $19,420 to $67,030 (Table 5). On a per-acre basis (WC/Acres), average costs fall – the per-acre cost of a 30-acre wetland is about 60 percent of the average cost of a 5-acre wetland. Spatial differences in costs are sizeable, just within Ohio – our model predicts that the cost of a 10-acre wetland ranges from $21,000 to over $32,000 (Figure 2). Figure 2 illustrates the relatively insignificant effect of *Rural* on total wetland cost, as there is no noticeable jump in wetland costs near urban areas. These cost estimates do not account for the possibility that there are no landowners within the county who are willing to take the current easement payment rate – or a marginally higher rate. However, every county in Ohio has program participants – an indicator that payment rates have been high enough to encourage participation.

In order to compare the cost-effectiveness of an investment in a wetland to the annual cost of EQIP incentive payments, we annualize wetland costs by assuming an infinite life and a discount rate of five percent. We believe a five-percent rate is reasonable, given that the easements are permanent and the discount rate represents the historical-average Federal funds rate. While there is no agreement on a 'correct' discount rate, a five percent rate is not uncommon. We, later, test the sensitivity of our results to the selected discount rate.

As wetland size increases, annualized incremental wetland cost stays around $95 while average cost falls from $194 to $110 (Table 5). These estimates are used to derive estimates of the cost-effectiveness of wetlands.

The wetland nitrogen-removal model was run for four different-sized wetlands at the base of a 1,000-acre wetland watershed. The expected nitrogen inflow rate is 16,000 pounds per year. Simulations were run for 20 years to capture effects of a wide variety of weather conditions, such as the effect of heavy rains and drought. Simulation results indicate that one can expect 5- to 30-acre wetlands to remove 4,000 to 10,000 pounds of nitrogen per year (Table 6). On an average, per-acre basis, the 5-acre wetland appears to be more effective at removing nitrogen.

Across the wetland sizes, we estimate incremental marginal effects by subtracting nitrogen removed by the smaller wetland from that removed by the next largest and dividing by the difference in the wetlands' acreages. For example, a 10-acre wetland removes 2,150 pounds more nitrogen than a 5-acre wetland – the five additional acres remove, on average, 430 pounds per acre (e.g., 2,150/5). Estimates of incremental effects range from 170 to 430 pounds per acre per year (Table 6). This is a rather substantial falloff in the effectiveness – the marginal effect of increasing from a 5- to a 10-acre wetland is 2.5 times the effect of increasing from a 20-to a 30-acre wetland (Table 6).

2.3. Data and model simulations of costs and nitrogen removal by VFS. The two dominating costs of VFS are the cost of taking the land out of agricultural production and the cost of establishing cover. EQIP provides incentive payments for planting cover as does the CRP. The CRP also compensates landowners for retiring their land. The CRP rental payments are a direct measure of the cost of retiring agricultural land, under permanent cover with some non-intrusive-use options available.

EQIP, among other things, establishes grassed field buffers and grassed and forested riparian buffers. The landowner commitment is 10 years. In Ohio, the 2010 EQIP payment schedule lists payments for grass filter strips at $307, $311, or $370 depending on whether seeding includes herbicides, fertilizers, or both (practice code 393, see http://www.oh.nrcs.usda.gov/programs/eqip/eqip2010.html). For hardwood forest riparian cover (practice code 391) (the payment schedule does not list forest buffer strips), EQIP pays $715/ac, $644/ac for conifers (both include the cost of seedlings), and $422/ac with direct seeding. There is an additional compensation of $42/ac if weed control is used. If trees are free, the establishment cost is $356/ac.

Annualizing across the 10-year life of the contract, at a five percent discount rate, EQIP grassed VFS costs range from $38 to $46 per acre, per year; forested strips range from $44 to $94. EQIP payments indicate no scale effects.

The CRP is a land-retirement program where farmers are compensated for the cost of land and establishing permanent cover. Most CRP contracts are 10-year contracts. Across Ohio, CRP cost-share payments have ranged from 0 to approximately $60 with a median value of $48 per acre. The CRP cover payments represent a one-time cost. Annualizing across 10 years at a five percent discount rate, CRP grass cover costs range from zero to $7.00 per acre.

We assume that the annual value of CRP land is approximately equal to the county-average farmland rental rate. County-level farmland rental rate data are obtained from NASS which built estimates on data from various sources, including the recently-initiated county-level cash rents survey. In Ohio, 2008 farmland rental rates ranged from $30 to $146 per acre. With Williams County land cost at $83 and 0 to $7 range in restoration costs, grassed VFS costs range from $83 to $90, about double the grassland cost of EQIP. One factor likely to be driving this difference is that farmers can regularly harvest grasses off EQIP acreage but not the CRP. The $83 to $90 estimate is consistent with the estimate reported by Doering et al. (1999). However, Doering et al. estimate was calculated for the entire Mississippi basin and thus does not capture regional variation in costs of VFS, as the approach used here is meant to do.

The lowest-cost estimate of VFS is $38; the highest is $94; these estimates are used to calculate cost-effectiveness. Both are based on EQIP data. The lower value is based on the cost of establishing grassed cover; the higher is based on the cost of establishing forested cover. While the $94 forest cover cost is more than double the EQIP grass cover cost, it is comparable to the CRP-based cover cost estimate.

To model VFSs' effectiveness at removing nitrogen from runoff waters, we draw from the models of White and Arnold (2009). As discussed earlier, their analyses account for the distribution of runoff flows across the VFS. In one of their scenario, where 80 percent of the runoff flows (evenly) over 10 percent of the VFS, their model predicts nitrogen removal rates of, approximately 38, 28, 22, and 20 percent when field-to-VFS ratios are 50-, 100-, 150-, and 200-to-one, respectively – a low-rate scenario (Table 7). In a second scenario, where 20 percent of the runoff flows over 10 percent of the VFS, the White and Arnold's model predicts that 83, 70, 56, and 48 percent of the nitrogen runoff is removed when field-to-VFS ratios are 50-, 100-, 150-, and 200-to-one, respectively – the high-rate scenario (Table 7).

The quantity of nitrogen runoff removed is the product of the removal rates and the quantity of nitrogen runoff. The quantity of nitrogen runoff – an output or our NLEAP simulations – ranges from 0.23 to 0.37 pounds per acre or 230 to 370 pounds per year from 1,000-acres. For simplicity, we consider the lowest and highest impact scenarios. The low-impact scenario combines the low runoff estimate with the low removal rate function – the least amount of nitrogen expected to be removed by the VFS. Conversely, the high-impact scenario combines the high runoff estimate with the high removal rate. For example, in the high-impact scenario, the high runoff level (370 pounds/year) is combined with the high removal rate function (48, 56, 70, and 83 percent) (Table 7).

The per-acre quantity of nitrogen runoff removed is derived by dividing the quantity of nitrogen removed by the size of the VFS. For example, in the high – impact scenario where the field-to-VFS ratio is 200, the VFS (associated with a 1,000-acre field) would be 5 acres; 48 percent – or 178 pounds – of the nitrogen runoff would be remove; hence the VFS removes about 36 pounds/acre (Table 7).

Our estimates of nitrogen removal from sub-surface flows are based on Mitsch et al. (1999, p. 47) who report that forested VFS remove 17.8 to 53 pounds per acre. These quantities are based on, what they assume to be, nitrogen flow rates typical of those in corn-producing areas. Hence, although NLEAP-GIS generates estimates of nitrogen subsurface flows, we cannot apply these to variations in our evaluation of the effectiveness of VFSs.

Based on Mitch et al. (1999) pounds-per-acre estimates, the total quantity of nitrogen removed by VFSs of five to 20 acres is 89 to 1,160 pounds annually which is more than twice the estimated quantity removed from runoff (Table 7). But the marginal and average removal rates applied here are equal so we see no falloff in the nitrogen removed as VFS acreage increases.

Mitsch et al. (1999) caution that their estimates are likely to overstate nitrogen removal because they do not account for such things as nitrogen returning to the water from fallen limbs and leaf litter.

3. Cost-effectiveness of nitrogen control strategies

3.1. Cost-effectiveness of on-field practices. Reducing losses of reactive nitrogen losses through nitrogen BMP incentives on land in corn production will not cost $0.33 per pound, given the assumptions applied here. If payments target lands in a corn-corn rotation, cost-effectiveness nearly doubles to $0.17 per pound (Table 8). This difference is due to the difference in BMP impacts, not costs. Moving from a strategy that targets corn-corn acreage to one that targets all corn acreage reduces nitrogen losses by over 3,300 pounds but at a cost of $0.62 per pound (Table 8). Though less cost-effective, expanding nitrogen-BMPs to the 416 acres in a corn-soybean rotation, nitrogen losses are reduced by 3,340 pounds – given the assumptions we've made.

These cost-effectiveness estimates are similar to those reported in earlier studies. For a 5-percent reduction in agricultural nitrogen losses, Ribaudo et al. (2001) estimated an average cost of $0.25 per pound, rising to $1.80 when nitrogen losses are reduced 26 percent. Doering et al. (1999) report a $0.44 per pound average cost for a 20-percent reduction in nitrogen loadings.

3.2. Cost-effectiveness of wetlands. The cost-effectiveness of wetlands is driven by changes in both the nitrogen removal rate and incremental costs. Results indicate that, at $0.23 per pound, a 10-acre wetland is the most cost-effective (Table 6). The incremental cost-effectiveness of increasing the size of the wetland from 5 to 10 acres is $0.22. Moving from 10- to 20- and 20- to 30-acres incremental cost-effectiveness falls to 0.34 and $0.54, respectively. Average cost-effectiveness drops more slowly – the cost-effectiveness of a 20-acre wetland is $0.28 per pound and, for a 30-acre wetland, is $31 per-pound.

The total quantity of nitrogen removed by a 10-acre wetland is 6,220 pounds while a 20-acre wetland removes 8,950 pounds per year, roughly 40 percent more. A 30-acre wetland, given our scenario, is estimated remove over 10,600, and additional 20-percent increase, a greater quantity that would be removed by implementing nitrogen BMPs on 1,000 acres, given our assumptions.

Our estimates of wetlands' cost-effectiveness is lower than reported by Ribaudo et al. (2001) – $1.80 per pound – and Doering et al. (1999) – $3.03 per pound. This difference would result if wetlands were less cost effective outside of our study area. Also, the previous studies evaluated large increases in wetland acreage so that their results may reflect a fall in the effectiveness of wetlands, a rise in the cost of wetlands, or both.

3.3. Cost-effectiveness of VFSs. To derive upper- and lower-bound estimate, we combined the least-cost estimate with the high nitrogen removal scenario and, conversely, the high cost estimates with the low nitrogen removal scenario. This approach suggests that cost-effectiveness ranges from $0.41 to $4.25 per pound (Table 9). Within both scenarios, the 5-acre VFS is most cost-effective. However, a 5-acre VFS in the best-case scenario is more than seven times as cost-effective a 5-acre VFS in the worst-case scenario.

Results also indicate that smaller VFSs in our analysis are more cost-effective (Table 9). Average cost-effectiveness falls by 25 percent (from $0.41 to $0.52 per pound in the best-case scenario) when the size of the VFS increases four-fold (the VFS increases from five to 20 acres). Incremental increases in cost-effectiveness are greatest around mid-sized (~10 acre) VFSs. The non-linearity of the cost-effectiveness estimates is driven solely by VFS effectiveness – incremental cost is constant.

The quantity of nitrogen removed, given the VFS protects 1,000 acres, ranges from 135 to 1,470 pounds, depending on the size and effectiveness of the VFS, a lesser quantity than on-field and wetland-restoration strategies (Table 7).

4. Discussion of results

Based on our results, we make the following caution conclusions.

First, the most cost-effective means of reducing nitrogen loadings is to target land in a corn-corn rotation – $0.17 per pound of nitrogen reduced (Table 8). Given the crop rotations in our analysis and 1,000 acres of cropland, nitrogen losses would fall by approximately 5,560 pounds per year.

Second, the cost-effectiveness of a wetland, at $0.23 per pound, is the second most cost-effective strategy (Table 6). The quantity of nitrogen that would be removed by a 10-acre wetland at the bottom of a 1,000 acre wetland watershed is an estimated 6,220 pounds.

Third, expanding to a 20-acre wetland is next most cost-effective strategy. The additional 10 acres would remove an additional 2,730 pounds at a cost of approximately $0.34 per pound. In total, a 20-acre wetland will remove 8,950 pounds of nitrogen at an average cost of $0.28 per pound.

Fourth, the cost-effectiveness of a VFS, in a best-case-scenario, is $0.41 per pound. This estimate is based on a 1:200 ratio between the VFS and the upland acreage so that, given a total of 1,000 upland acres and 5 acres of VFS, nitrogen losses would be reduced by 468 pounds per year. Thirty acres of VFS would remove 1,470 pounds at a cost of $0.52 per pound. This scenario is a best-case scenario – in the worst case scenario, costs 1,000-percent higher.

The fifth most cost-effective strategy is to move from a 20- to a 30-acre wetland. Nitrogen losses are reduced by 1,700 pounds at a cost of $0.54 per pound. The total quantity of nitrogen removed by the wetland would be 10,650 pounds at an average cost of $0.31 per pound.

Finally, expanding nitrogen BMPs to lands in corn-soybean rotation will reduce nitrogen losses by approximately 3,340 pounds at a cost of $0.62 per pound. The gross, overall reduction in nitrogen losses through nitrogen BMP adoption on 1,000 acres is 8,900 pounds at a cost of $0.33 per pound. This is about the same quantity of nitrogen that could be removed by a 20-acre wetland, but at a cost of $0.28 per pound. Hence, when choosing among nitrogen conservation strategies, one must also consider the nitrogen-reduction goal. For example, if the environmental goal is to decrease the quantity of nitrogen reaching waterways by 5,000 pounds per year, then targeting land in corn-corn rotation is most cost-effective. Should the goal be to reduce nitrogen loadings by 8,000 pounds, then wetland restoration may be most cost-effective. Of course, an additional, equally important consideration is the total number of acres that a policy would affect; the 1,000 acres we have used serves as an example. Additionally, one could narrow the scope of onsite targeting by, for example, targeting corn-corn acreage on sandy soils. Efficiency can be increased 10 to 20 percent. Gains from alternative strategies can be calculated from the estimates in our tables.

The strength of these conclusions is moderated by assumptions we've made and the present limits to our models' capabilities. Because of spatial limitations to our models, we are unable to examine cost-effectiveness across large areas, relevant to Federal policy.

Given the assumptions we've made, we may be overstating the cost-effectiveness of on-site practices. One reason is that we've assumed that producers would not violate BMP practice agreements. Some may, given that they are unlikely to be caught. Violations would be reduced with monitoring, but then the monitoring cost would lower cost-effectiveness. Another reason is that we've assumed that at least some producers would adopt at the lowest nitrogen BMP incentive payment ($5.00 per acre), which may not happen. EQIP already offers payments as high as $40.00, though it is not clear why. We have no idea as to what extent these assumptions might be violated. So, the best we can say is that we have probably overstated estimates of the cost-effectiveness of nitrogen BMPs.

We are limited to an evaluation of two crop rotations. But these are very important rotations. One reason is that the corn-corn rotation is common and has the highest levels of nitrogen losses. The second reason is that the corn-soybean rotation is the most common rotation in Ohio. Also, our analysis did not account for lag times. Lag time decrease the cost-effectiveness of practices because of the time value of money. All three conservation strategies are likely to have time lags. If lag times are equal, then the relative cost-effectiveness of conservation strategies will not change. Analyses of practice time lags are beyond the scope of this analysis.

Our evaluation of the cost-effectiveness of wetlands may also be high. Wetlands would be less cost-effective if not restored at a site that is well-suited for nitrogen entrapment. Also, if USDA would need to increase the size of its payments to get additional landowners to participate, wetland restoration would not be as cost-effective. The results are based on the assumption that there are sites where wetlands might be restored – there may not be any. However, because results are area- or county-wide approximations, the potential site can lie in a wide area. For example, the wetlands model results are likely to be applicable to lands within a 20-mile radius or more, which embodies more than 1,200 square-miles. Furthermore, much of the area was once covered by wetlands. For example, two wetland complexes, the Great Black Swamp and the Great Kankakee Marsh, each more than 1,000,000 acres once lay in northern Indiana, Illinois, and Ohio but were systematically drained for agriculture (Mitsch and Gosselink, 2000). Note too that a wetland-restoration strategy can involve any sized wetland watershed – our 1,000-acre watershed serves as an example. Our wetlands model can be used to examine the cost-effectiveness of different-sized wetland watersheds.

In evaluating the cost-effectiveness of wetlands, we assumed a five percent discount rate, which we feel is reasonable. A look at the function used to estimate cost-effectiveness shows that the effect of the discount rate is a direct multiple to the cost-effectiveness estimate. A lower (higher) will raise (lower) wetlands' cost-effectiveness. For example, if one was to assume a four percent discount rate, the cost-effectiveness of a 10-acre wetland would move from \$0.23 per pound to \$0.18 per pound.

Our analysis of VFS may have overestimated cost-effectiveness. We have assumed that landowners would accept the current payment rate. What's more, we focused our discussion on the best-case scenario and (for lack of better data) have applied higher, forested land nitrogen removal rate and a cost estimate based on grassland restoration. The best-case scenario serves as an upper-bound example – we cannot argue that best-case scenario exists.

Our analysis did not evaluate the cost-effectiveness of mixed strategies, such as the use of wetlands and on-field conservation practices. Mixed strategies may be cost-effective, but calculating cost-effectiveness is more complicated because the cost-effectiveness of one strategy is dependent on the use of others. For example, full adoption of on-field conservation practices will have a smaller water quality impact if there is a wetland already removing nitrogen. Similarly, the quantity of nitrogen removed by a wetland is reduced if on-field conservation practices have been expanded and reduced on-field nitrogen losses.

Another relevant factor, but beyond the scope of this analysis, is the extent to which conservation practices decrease variations in nitrogen loadings. High nitrogen loadings for a short period of time can have greater adverse environmental impacts than a steady flow of nitrogen at lower loadings, irrespective of the quantity of nitrogen. In a simulation analysis, Bystrom et al. (2000) found that, when one includes the uncertainty in damages associated with variations in nitrogen loadings, the cost-effectiveness of wetlands increases (given their ability to slow runoff) relative to on-field practices.

Summary and direction for future research

The objective of this analysis is to provide a framework for determining how public funds could be allocated, spatially and by type of conservation practice, in order to reduce reactive nitrogen loadings at least cost. To do so, our analysis generates local-level, not site-specific, estimates of the costs and the effects of various nitrogen conservation practices. This framework, when applied to localities across a wide geographic area, can provide spatial approximations of where one conservation practice is more cost-effective than another and where each practice is likely to be most cost-effective.

To meet this objective, we have developed, estimated, and integrated biophysical and economic models of on- and off-field nitrogen conservation practices for a county in the State of Ohio. Our cost estimates are based on current USDA conservation incentive payments, which are direct costs hence well-suited, measures of public costs. Our biophysical models account for a range of conservation options and are calibrated to local environmental conditions.

The cost-effectiveness estimates for the on-field conservation practices combine USDA conservation incentive payments rates with our estimates of the reductions in nitrogen losses generated by NLEAP-GIS. NLEAP-GIS is a field-level simulation model that generates expected nitrogen losses based on the mix of production practices, all aspects of nitrogen use and carry-over, crop rotations, and a range of environmental conditions.

The cost-effectiveness estimates for wetlands are based on models explicitly designed to estimate costs and wetlands' nitrogen removal. Our nitrogen removal model has been demonstrated to provide reasonable results, as long as the wetland is located in an area where it captures large-scale runoff (Crumpton et al.). Our wetland-cost model was estimated using USDA incentive payment rates for for restoring wetlands on privae agricultural lands and other data. The model captures 86 percent of the variation in wetland costs (our dependant variable).

At present, the extent to which we have calibrated NLEAP-GIS confines our analysis to a single county. Therefore, the results presented here cannot be used to guide spatial decisions on nitrogen conservation funding. But the approach presented here can be applied to broader regions as model development continues.

Results suggest that, within our study area, on-field nitrogen BMPs, targeted to fields in a corn-corn rotation are most cost-effective, having a nitrogen-reduction cost of \$0.17 per pound, followed by a 10- to 20-acre wetland (located at the bottom of a 1,000-acre wetland watershed) with cost-effectiveness ranging from \$0.23 to \$0.26 per pound. These results, as with earlier research (Ribaudo et al., 2001; Doering et al., 1999; Peters, 1999), indicate that on-field conservation practices are most cost-effective, at least when targeting lands in corn-corn rotation. However, our results also suggest that wetland restoration might be more cost-effective than reported in past research.

Future research could expand the rigor and spatial range of this analysis. For example, NLEAP-GIS, or a similar model, might include a broader range of crop rotations and field practices and be calibrated to a wider area. Also, cost assessments of on-field practices might be improved as program data that capture spatial variability in farmers' responses to marginal increases in nitrogen BMP incentives become available. Our wetlands nitrogen-removal model is calibrated to conditions throughout much of the upper Mississippi watershed and to a variety of wetland sizes. But expanding the model to areas such as the Chesapeake Bay watershed would allow for more-extensive policy applications. The wetland cost model is being expanded to a wider area. Extensive analysis of the effectiveness of VFS is needed along with geospatial data that allows one to determine the pattern, rate, and depth of underground water movement.

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Appendix

Table 1. The distribution of all combinations of nutrient BMPs in use by crop rotation in the study area

Crop rotation	All BMPs	BMP rate and timing	BMP rate and method	BMP timing and method	BMP rate	BMP timing	BMP method	No BMPs used
	Percent							
Corn-corn	25	4	4	2	7	4	2	51
Corn-soybean	32	12	10	13	19	8	0	6
All corn	31	11	10	12	18	8	<0	10

Note: Rows may not sum to 100 due to rounding.

Table 2. Changes in nitrogen leaching with the adoption of nutrient BMPs and conservation tillage on 584 acres in corn production

Soil type with and without tile drainage systems	Current practices	Adopt BMP application	Adopt BMP timing	Adopt BMP method	Adopt all BMPs
	Total leached	Change in leaching			
	1,000 lbs				
Sandy	22.66	-6.53	-5.88	6.13	-9.95
Sandy with tiles	26.90	-6.89	-6.65	6.15	-11.04
Clay	14.17	-4.07	-3.98	3.27	-6.69
Clay with tiles	16.22	-5.53	-4.08	3.96	-7.92
Average	19.99	-5.75	-5.15	4.88	-8.90

Table 3. Changes in nitrogen leaching with the adoption of nutrient conservation practices on 90 acres in a corn-corn rotation

Soil type with and without tile drainage systems	Current practices	Adopt BMP application	Adopt BMP timing	Adopt BMP method	Adopt all BMPs
	Total leached	Change in leaching			
	1,000 lbs				
Sandy	8.81	-4.38	-3.70	4.80	-5.98
Sandy with tiles	9.39	-4.41	-3.89	4.70	-6.25
Clay	5.83	-3.19	-2.60	2.75	-4.61
Clay with tiles	7.13	-4.23	-2.62	3.23	-5.40
Average	7.79	-4.05	-3.21	3.87	-5.56

Table 4. Parameter estimates of the land cost model

Variable	Estimate	Standard error	*t*-value
Intercept	-4350	2780	-3.17
AgrValue	12.3	0.348	34.6
Acres	1,070	29.0	34.5
Acressq	$-5.01*10^{-2}$	$4.37*10^{-3}$	16.4
Rural	-353	14.9	-8.14
Fsize	-0.740	0.0830	-12.4

Table 5. Estimated costs of restoring wetlands in the study area

Wetland size	Wetland cost			Annualized wetland cost	
	Total	Incremental	Average	Incremental	Average
Acres	$	$/acre		$/acre/year	
5	19,420	na	3,884	na	194
10	28,950	1,906	2,895	95	145
20	48,000	1,905	2,400	95	120
30	67,030	1,903	2,204	95	110

Note: na – not available.

Table 6. The nitrogen removal and cost-effectiveness of a restored wetland receiving 16,000 pounds of nitrogen annually

Wetland size	Nitrogen removed	Incremental quantity removed	Average removed*	Incremental cost-effectiveness	Average cost-effectiveness
Acres	lbs	lbs/ac		$/lb	
5	4,070	na	816	na	0.24
10	6,220	430	622	0.22	0.23
20	8,950	273	447	0.34	0.28
30	10,650	170	355	0.54	0.31

Note: na – Not available.

Table 7. Nitrogen removed by vegetative filter strips

Size of VFS	Runoff removal rates		Total quantity removed		Incremental quantity removed		Average quantity removed	
	Low rate[1]	High rate[2]	Low rate and low runoff[3]	High rate and high runoff[4]	Low rate and low runoff	High rate and high runoff	Low rate and low runoff	High rate and high runoff
Acres	percent		lbs		lbs/ac		lbs/ac/yr	
5	20	48	46	178	na[5]	na	9.2	35.6
6.7	22	56	51	207	3.3	8.8	8.4	30.9
10	28	70	64	264	3.9	11.4	6.4	26.4
20	38	83	87	313	2.3	4.90	4.3	15.3
Subsurface nitrogen flow								
Size of VFS	Subsurface removal rates		Total removed		Incremental quantity removed		Average quantity removed	
			Low rate[6]	High rate[7]	Low rate	High rate	Low rate	High rate
Acres	percent		lbs		lbs/ac		lbs/ac	
5	na	na	89	290	17.8	58	17.8	58
6.7	na	na	119	389	17.8	58	17.8	58
10	na	na	178	580	17.8	58	17.8	58
20	na	na	356	1160	17.8	58	17.8	58
Nitrogen runoff plus subsurface flow							Surface runoff plus subsurface nitrogen flow	
Size of VFS	Runoff removal rates		Total removed		Incremental quantity removed		Average quantity removed	
			Low scenario	High scenario	Low scenario	High scenario	Low scenario	High scenario
Acres	percent		lbs		lbs/ac		lbs/ac	
5	na	na	135	468	na	na	27.0	93.6
6.7	na	na	170	596	21.1	66.8	26.2	88.9
10	na	na	242	844	21.7	69.4	24.2	84.4
20	na	na	443	1473	20.1	62.9	22.1	73.3

Notes: [1]110% of the VFS receives 80 percent of the runoff; [2]10% of the VFS receives 80 percent of the field runoff; [3] low runoff = 230 lbs/yr; [4]high runoff = 370 lbs/yr; [5]not appropriate; [6]removal rate = 17.8 lbs/yr; [7]removal rate = 58 lbs/yr.

Table 8. Reductions in nitrogen losses by and cost-effectiveness of on-field conservation practices

Rotations	Total acres	Land converted to BMPs	Total nitrogen reduced	Incremental quantity removed	Average quantity removed
	Acres		lbs	lbs/ac	$/lbs
Corn-corn	168	126	5,560	33	0.17
Corn-soybean	416	287	3,340	8.0	0.62
All corn	584	413	8,900	15	0.33

Table 9. Cost-effectiveness of VFS

Size of VFS	Incremental cost/incremental quantity		Average cost/average quantity	
	Low scenario	High scenario	Low scenario	High scenario
Acres	$/lb			
5	na	na	3.48	0.41
6.7	4.45	0.57	3.59	0.43
10	4.33	0.55	3.88	0.45
20	4.68	0.60	4.25	0.52

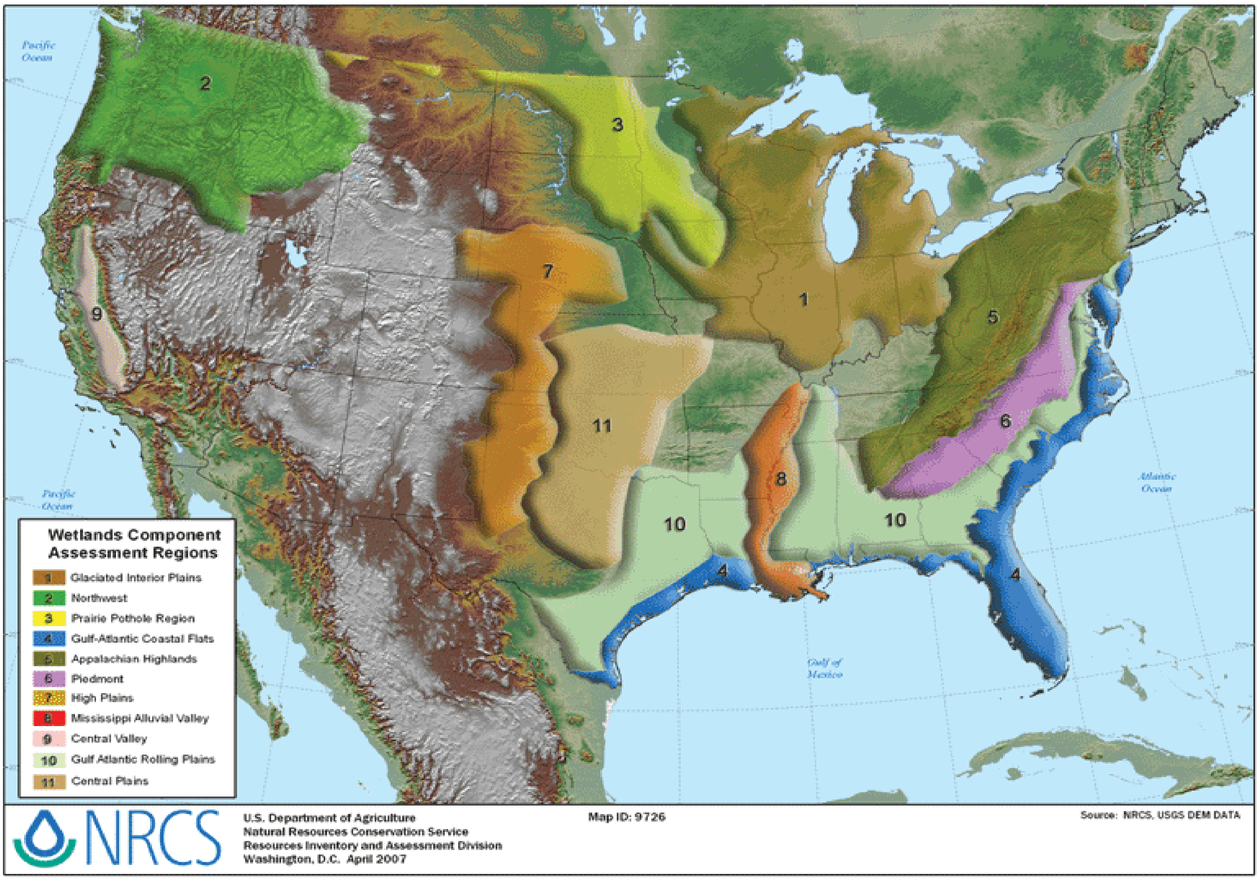

Fig. 1. The Glaciated Interior Plains and other major wetland regions

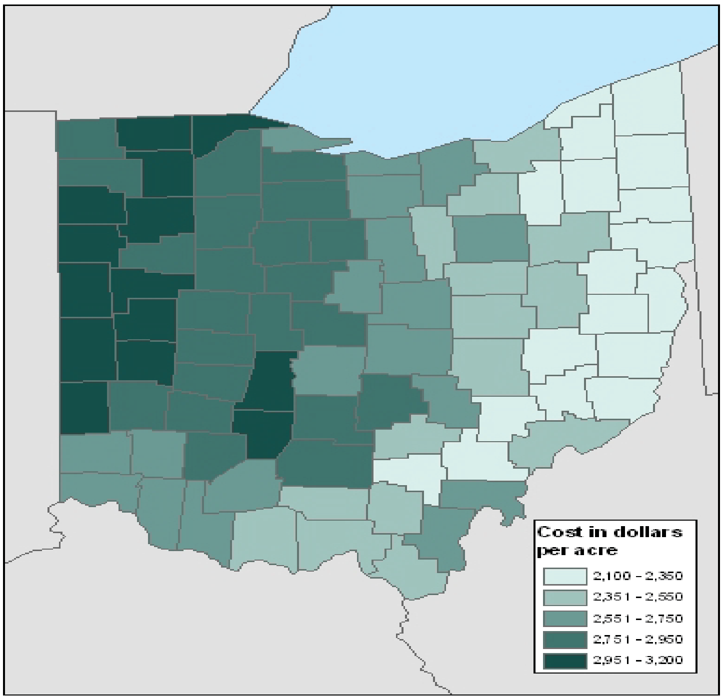

Fig. 2. Variation in wetland easement costs across Ohio