Water Resources and Economics 15 (2016) 43–56

Contents lists available at ScienceDirect

Water Resources and Economics

journal homepage: www.elsevier.com/locate/wre

How cost-effective are cover crops, wetlands, and two-stage ditches for nitrogen removal in the Mississippi River Basin?

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ARTICLE INFO

Article history:
Received 30 October 2015
Received in revised form
17 June 2016
Accepted 20 June 2016

Keywords:
Water quality economics
Nutrients
Eutrophication
Conservation practices
Two-stage ditch
Wetlands
Cover crops

ABSTRACT

Excess nitrogen (N) causes numerous water quality problems, and in the upper Mississippi River Basin, much of the excess N results from landscape modifications necessary for row crop agriculture. Several conservation practices reduce N export, but cost estimates for these practices are often lacking, which can inhibit decisions by farmers and policy-makers. Many practices are eligible for cost-share funds from the United States Department of Agriculture (USDA), but these programs do not usually cover the full cost, and so farmers need to be able to approximate their share of costs. In addition, cost estimates may help the USDA to set priorities and make programmatic decisions. We address lack of cost information by estimating the direct implementation costs and USDA program costs for three agricultural conservation practices: wetlands, cover crops, and two-stage ditches, over 10 and 50 year time horizons. We then compare these costs to the N removal effectiveness of each practice, in $ kg N^{-1} removed. Wetlands were the most cost-effective practice (in $ kg N^{-1} removed) over both time horizons. Over 50 years, the two-stage ditch ranked second in cost-effectiveness and cover crops were least cost-effective, while over 10 years, cover crops were second and two-stage ditches were least cost-effective. Finally, we note that these practices need not be used in isolation, but can be implemented simultaneously to maximize N removal. Overall, our analysis suggests that careful implementation can cost-effectively mitigate N pollution.

1. Introduction

Excess nitrogen (N) is a problematic pollutant in freshwater and coastal ecosystems worldwide [1], and it can contaminate drinking water [2], decrease biodiversity [3], and cause coastal hypoxia [4]. In the Gulf of Mexico, most of the excess N originates from the upper Mississippi River Basin [5], especially in the Corn Belt where row-crop agriculture is the predominant land use [6]. In typical row-crop agriculture in the Corn Belt, N fertilizers are added annually to improve crop growth, and fields are artificially drained to keep the root zone aerated. Artificial drainage includes sub-surface tile drains and channelized streams and ditches, which often receive tile drain inputs [7]. These landscape modifications optimize crop yields, but also facilitate the export of excess N to downstream water bodies [8,9]. In an effort to maintain yields and minimize costly ecosystem impairments caused by nutrient transport, states throughout the Mississippi River Basin have developed nutrient reduction strategies [10], which focus on the widespread voluntary adoption of effective, nutrient-reducing, conservation practices (e.g., [11]).

These nutrient reduction strategies incur large

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http://dx.doi.org/10.1016/j.wre.2016.06.003
2212-4284/

economic costs, as do the consequences of N pollution [12–14]. As a result, several studies have examined how to mitigate N pollution, while minimizing costs. Macroeconomic studies have examined the large-scale effects of basin-wide management, including extensive wetland restoration and mandatory reductions in fertilizer application rates [15,16]. These analyses have provided important insights into the economic and social welfare consequences of large-scale N management. They demonstrated that mandatory fertilizer reduction was the most cost-effective option for reaching modest N-reduction goals in the Mississippi River Basin, but wetlands restoration was preferable for achieving more ambitious N-reduction goals [15,16]. Others have quantified N-reduction costs within small geographic areas, which allows for exploration of relationships between biophysical function and economic outcomes, providing estimates of the cost of various management strategies to remove a unit of N [17,18].

These management strategies involve the implementation of conservation practices, which are typically either cultural or structural in nature [19]. As described in Tyndall and Roesch [20], cultural practices include in-field practices that minimize erosion or nutrient transport, such as cover crops, nutrient management, and conservation tillage. Structural practices involve natural or artificial structures that are placed within or at field edges and often feature perennial vegetation and/or landform engineering. Structural practices are designed to capture or treat sediment and nutrient runoff, reducing delivery to downstream water bodies. In order to be used effectively, conservation practices must be: 1) bio-physically effective; 2) compatible with the landscape context and farming system; and 3) financially practicable – that is, they must be affordable and cost-effective [20].

There are often a number of pragmatic economic unknowns in the usage of conservation practices, and this financial uncertainty can be a barrier to implementation [17,21]. Specific and up-to-date cost estimates are lacking for many practices, which make it difficult for farmers to determine if a practice is affordable (in general or relative to alternative practices). Also, although the N-removal potential of most practices has been variously documented, their differing biophysical structures and costs make comparisons problematic for cost-effectiveness analysis. Cost-effectiveness analysis identifies the least cost method for meeting a specific physical outcome, for this study it is simply the reduction of 1 kg of transportable N. Finally, financial incentives, such as cost-share or rental payments, are often critical in the adoption process for many farmers [22]. Such funding is often contingent upon its disbursement being used efficiently in both economic and biophysical contexts. Unfortunately and critically, very little information exists that allows agencies such as the USDA Natural Resource Conservation Service (NRCS) or Farm Service Agency (FSA) to assess the relative effect of their financial programming on reducing the monetary burden on participating landowners. The purpose of this study is therefore three-fold: 1) to estimate the direct implementation cost for three different N-reducing conservation practices; 2) compare these costs to existing USDA cost-share programs; and 3) determine comparative measures of cost-effectiveness in terms of $ kg N^{-1} removed. For this study we examine one in-field cultural practice, cover crops, and two edge-of-field/waterway structural practices: wetland restoration and two-stage ditch construction.

Cover crops (e.g. annual and cereal ryegrass, wheat, oat, forage radish) can reduce N leaching by immobilizing N during the winter, when fields are otherwise devoid of vegetation and thus prone to N losses [23]. Cover crops are planted after the fall harvest (or inter-seeded at the end of the growing season), and grow and incorporate nutrients during the winter when fields are normally fallow [24]. In the spring, cover crops have either self-terminated over winter, or are actively terminated with an herbicide or by mechanical means (e.g. crimping, rolling, cutting), or plowed under prior to planting of the production crop. In addition to their prevention of soil N leaching, cover crops can also help to minimize soil erosion, improve soil quality, and enhance habitat [24–26].

Wetlands and two-stage ditches both capture field runoff and promote N processing and removal prior to its export downstream [27,28]. For both, N removal occurs through assimilatory uptake into biomass and via microbially-mediated denitrification into N gasses. Nevertheless, key aspects of their landscape placement and hydrology make wetlands and two-stage ditches distinct practices. Restored wetlands are designed to receive N-rich water from tile drain outlets or small ditches, and they hold and process the water before it discharges to a larger surface water outlet [27]. Wetland sediments are typically anoxic and rich in organic matter, which promotes denitrification [27,29]. Importantly, wetlands are generally most effective under base flow conditions, when water flow into wetlands is slow enough to be retained [30].

In contrast, two-stage ditches were originally developed to address the instability of stream banks in conventional, trapezoid-shaped ditches [31], which are channelized to move water downstream quickly. As such, conventional agricultural ditches and streams typically have steep banks, which often fail during high flows, depositing sediment within the channel [32]. This unsustainable morphology requires regular channel dredging to maintain drainage capacity, which minimizes biological N processing and removal. In a two-stage ditch, floodplains are constructed adjacent to the incised channel. During times of high discharge, water flows onto the floodplains, which reduces water velocity and shear stress, and the ditch remains stable, eliminating the need for periodic dredging [33]. In addition, the two-stage ditch increases water residence time and stream surface area, providing more time and space for N removal processes such as denitrification [28].

As a key complementary practice to both wetlands and two-stage ditches, grass buffer strips are uncultivated zones adjacent to the field and the wetland edge or ditch, and they are intended to serve as a transition area between row crops and the ditches or streams, and can reduce surface erosion and nutrient inputs to adjacent waterways. Grass buffer strips stand alone as a conservation practice (e.g., Vegetative Filter Strip, NRCS Practice Code

393) and are used in similar fashion to two-stage ditches in their size and location (lateral to streams). They are also often combined with both wetlands and two-stage ditches and incorporated directly into their design and cost considerations [34]. In a tile-drained landscape, the majority of N transport occurs when nutrient-laden water bypasses grass buffers via subsurface tile drainage, and is delivered directly to the stream [35]. As a result, grass buffer strips in the tile-drained Corn Belt are associated with minimal N removal [23,35–38]. Nevertheless, because grass buffers are often coupled with wetlands and two-stage ditches, we factor them into the wetland and two-stage cost assessments, but they are not included in the estimates of N reduction.

Implementation of conservation practices, including cover crops, restored wetlands, and two-stage ditches, results in a private cost to farmers/landowners while ostensibly producing a significant benefit to the watershed (society). Such practices are therefore eligible for incentive funding from the USDA where payments facilitate implementation by alleviating financial hurdles while not covering all associated costs. In most cases, profit-maximizing farmers will implement a practice when the perceived benefit(s) is/are greater than or equal to the implementation and opportunity costs [39], and therefore, they typically must receive benefits beyond direct USDA payments. Additional benefits can include direct or indirect monetary benefits, which occur when a farmer receives a financial benefit from the practice; for example, a two-stage ditch that eliminates recurring ditch clean-out costs, or rental payments on marginal lands that do not reliably turn a profit. A farmer may also benefit from the creation of wildlife habitat, improved landscape aesthetics, or other indirect benefits for which there is currently no market value, and therefore be willing to implement practices even when there is some associated net cost [40,41]. Nevertheless, monetary incentives can make conservation practices financially feasible by increasing the number of farmers who would receive a direct or indirect benefit [39].

Notably, USDA funding is finite, and thus implementation of cost-effective conservation practices may reduce excess N export without increasing costs. The cost and biophysical efficacy of each practice varies, and a comparison of their cost-effectiveness (in $ kg N^{-1} removed) can highlight practices and payment policies that are financially advantageous or burdensome from various points of view. In specific cases, the choice of practice for N mitigation may be constrained by site conditions or other environmental priorities (e.g., reducing soil erosion), but where practices are equally applicable, the most cost-effective practice may be the best choice at the field or reach scale. At broader scales, information regarding cost-effectiveness can significantly improve conservation planning [42].

In this paper, we estimate average per-unit implementation costs of cover crops, restored wetlands, and two-stage ditches for the US Corn Belt region. Next, we compare per-unit direct costs to the USDA under the Environmental Quality Incentive Program, EQIP, and the Conservation Reserve Program, CRP, so as to estimate both total USDA program expenditures and the potential remaining out-of-pocket expenses to farmers/landowners. We then assess the relative cost-effectiveness of each practice as an assessment of relative efficiency [43]. In addition, we estimate cost-effectiveness over two time horizons. In doing so, we provide comparisons of N-removal conservation practices across a range of physical and economic conditions, in order to inform policy and practice choice. Overall, we aim to provide decision-making criteria that will be useful to farmers and natural resource managers.

2. Materials and methods

We estimated direct implementation costs and costs to the USDA under current cost-share programs for cover crops, wetlands, and two-stage ditches. In addition, we calculated cost-effectiveness for each practice and estimated the benefits relative to costs. For each analysis, we focused on the U.S. Corn Belt, with its intensive agricultural land use and high nitrogen export [5], and we used two time horizons: 10 years and 50 years. The 10-year window is short, but consistent with contract lengths for CRP, which are typically 10–15 years long [44], and the 50-year period is typical of previous studies [16,45,46] and is also a more relevant timespan for a two-stage ditch [32,47,48], as well as for some wetlands.

2.1. Direct cost assessments

For cover crops, wetlands, and two-stage ditches, we determined the most typical establishment, management and opportunity costs across four Corn Belt states (IA, IL, IN, OH) using a mix of recent cost data sources, including custom rate surveys and regional transaction evidence (Supplemental Tables 1 and 2). For cover crops there are five main cost categories that farmers/ landowners must consider: (1) seed purchase, which varies regionally and year to year depending upon local seed source supplies and demand; (2) planting costs, which vary with seeding approach (e.g., aerially, broadcast seeding, interseeding); (3) termination costs, which include either herbicide application, crimping, cutting, rolling, or tillage; 4) additional costs associated with changes to cropping system management (e.g., extra time assessing cover crop progress, adjusting equipment); and (5) annual opportunity costs in the form of decreased crop yields. For this analysis, we assumed that cover crops did not negatively influence crop yields, because published accounts of negative yield impacts are inconclusive [49] and more recent studies have shown a yield benefit [50].

The overall cost for restored wetlands involves considerable and semi-irreversible structural work as well as long-term opportunity costs. Wetland establishment costs include basic design costs, earthwork, wetland and buffer seed and planting costs, and the purchase and construction of engineered water control structure(s). Management costs simply involve maintaining a grass buffer surrounding the wetland. The main cost for wetlands, however, is the long-term opportunity cost of the land converted to wetland, which is manifested in foregone land rent.

Similar to wetlands, the cost of two-stage ditches involves irreversible structural work as well as long-term opportunity costs. Establishment involves initial excavation (i.e., creating the floodplains), leveling out the side slopes, and placing spoil. Tile drain repair and/or stabilization may be required, as well. A vegetative filter strip (i.e., grass buffer) is then either established *de novo* or managed if already present. Relative to cover crops and wetlands, there is a distinct paucity of financial information available for two-stage ditches. We based our cost estimates on 16 different two-stage ditch projects completed from 2011 to 2014 in Indiana, Michigan, and Ohio. These projects were administered by The Nature Conservancy (TNC) of Indiana and The Ohio State University (Kent Wamsley, TNC and J.D. Witter, OSU, personal communication).

We capitalized those costs using a discounted cash flow procedure to calculate the present value cost (PVC) expressed in 2014 dollars. PVC is calculated as $PVC=\Sigma(P_t/(1+r)^t)$, where P_t is the cost of the practice at time t, r is the chosen discount rate, and t is time in years. The calculated total present value cost is then annualized by converting into an equal annual cost basis by applying the capital recovery factor [21]. The recommended Federal discount rate for water quality projects are published by the NRCS and represent average market yields on Treasury securities. As is appropriate for this type of analysis, it was performed using a real discount rate as opposed to nominal rate; that is, the effect of inflation is removed from the analysis [20]. The 2013 Federal "real" discount rate for water-quality oriented projects is 1.9% [51]. Only one discount rate was used because this is a cost-only assessment [52]; had revenues been involved, a range of discount rates would be necessary for sensitivity analysis.

Our cost estimates represent the average practice, but we note that there is variability within practices. Cover crop costs will vary with seeding and termination method (Supplemental Table 1) and wetland restoration costs will vary with the amount of earth movement and control structures required. Similarly, two-stage ditch costs are typically limited to soil excavation, leveling, and tile drain stabilization, but occasionally, ditches may require additional stabilization or planting. We left these costs out of our estimates, because they are atypical, but note here that most of these additional actions are also eligible for EQIP funding. These additional costs include practices that enhance conveyance capacity (NRCS Conservation Practice Standards 326,Clearing and Snagging) and channel stability (NRCS Conservation Practice Standards 322, Channel Bank Vegetation; 342, Critical Area Planting; 382, Fence; 395, Stream Habitat Improvement and Management; 472, Use Exclusion; 484, Mulching; 578, Stream Crossing; 580, Streambank and Shoreline Protection; 584, Channel Stabilization; 587, Structure for Water Control; and 647, Early Successional Habitat Development/Management).

2.2. USDA cost assessments

In addition to the direct implementation costs for each practice, we estimated the costs to the USDA under applicable conservation programs. Specifically, cover crops are funded through EQIP, which is administered by the NRCS in individual states, and can pay up to 75% cost-share for practice establishment [53]. All states in the Corn Belt include cover crops as an EQIP-eligible practice. For our USDA cover crop cost estimates, we used the average EQIP payment rate across four Corn Belt states (IN, IA, IL, and OH, Table 2).

Wetlands are eligible for funding through CRP, which is administered by the USDA FSA, and pays a minimum of 50% of direct wetlands restoration cost, a one-time sign up incentive, along with an annual rental payment to account for opportunity costs (i.e., foregone rent or profit loss from not growing crops [44]). We based our USDA wetland costs on mean CRP rental and cost-share payments from the four states for land enrolled in the program from 2010 to October 2014 (Table 2). CRP payments vary by project, depending on land values and the specific needs of the restoration. Nonetheless, the average payment is relatively consistent among states, and the average provides a good estimate for a typical wetland.

We estimated USDA two-stage ditch costs with a combination of EQIP and CRP. Floodplain excavation and other direct construction and restoration costs are covered under EQIP in Indiana and Ohio [54,55], with a 75% cost-share for implementation and no rental payments. The adjacent buffer strips are covered under CRP (CRP Practice CP21 Filter Strips), with 50% cost-share for establishment and an annual rental payment for opportunity costs. We applied EQIP payment rates from Indiana for the two-stage ditch construction costs, and applied mean CRP rental and cost-share payments from the 4-state region for buffer strip costs (Table 2). Because of their linear nature, two-stage ditch USDA payments are expressed in linear terms ($\$ \, m^{-1}$), but in order to standardize the comparison, we calculated the direct and program costs on an areal basis ($hectare^{-1}$ and $acre^{-1}$), using average agricultural ditch dimensions (floodplain width of 5.8 m (19 ft) and field-to-floodplain depth of 1.22 m (4 ft).

With this USDA data, we compare average program payments to the estimated average total direct costs of each practice so as to allocate the percent of total costs covered by USDA programs, and that for which a farmer/landowner is responsible.

2.3. Cost-effectiveness

To determine the cost per unit N removed, we divided the annualized costs by annual N removal rates for each practice, expressed in $kg\ N\ ha^{-1}\ yr^{-1}$, to get the cost of N removal in $\$\ kg\ N^{-1}$. We based our N removal rates on a literature review of the practices, as well as our own data on N removal in two-stage ditches [28,56]. We restricted our review to studies completed in the Corn Belt, to keep the N removal rates within a realistic range. For wetlands, we further restricted our review to studies of on-farm wetlands [57–62], and our cover crop review was restricted to studies that examined grass or grain cover crops [63–71], because leguminous species are used to increase soil N rather than reduce N leaching [24]. Over the 10-year and 50-year period, we assume that N removal rates remain constant, because none of the studies showed a consistent increase or decrease in N-removal rates through time.

For the two-stage ditch, we estimated annual N removal rates, based on hydrology and seasonal denitrification measurements in 6 two-stage ditches in IN, OH, and MI [28,56]. First, we identified the annual number of floodplain inundation days, based on continuously collected stream stage data from each ditch site. We then calculated the increase in N removal as the number of days inundated, multiplied by floodplain area and areal denitrification rate. We used Monte Carlo simulation (3000 runs) to determine a 95% confidence interval of annual N removal for each two-stage ditch. This simulation allows us to better estimate the range of annual N-removal rates, given the limited studies on two-stage ditches and the strong effect of hydrology and denitrification rates on N removal. In doing this simulation, we input the observed ranges of hydrologic conditions (i.e., number of floodplain inundation days) and measured denitrification rates, assuming a uniform distribution across the ranges, to estimate mean annual N removal, along with a 95% confidence interval. We completed the Monte Carlo simulation for the two-stage ditch only, because this is a newer practice with limited data (6 sites, 1–3 years of N-removal and hydrologic data).

3. Results

3.1. Cost assessment for cover crops, wetlands and two-stage ditches

Across the four states (IA, IL, IN, OH), the average annual cost of utilizing cereal winter rye (*Secale cereal*) cover crops is \$151 $ha^{-1}\,yr^{-1}$ (\$61.00 $ac^{-1}\,yr^{-1}$, Table 1). Iowa has the highest calculated costs for cover crops and Indiana the lowest. About 69% of the total cost is for establishing cover crops; the remaining 31% is for cover crop management including termination (Supplemental Table 1).

The USDA payment rate for cover crops varies with species planted, and we used the rate for cereal grains, which are generally more effective cover crops for N retention [72]. Assuming the NRCS EQIP Practice Standard 340 is used, the four-state average EQIP payment is \$121.85 $ha^{-1}\,yr^{-1}$ (\$49.33 $ac^{-1}\,yr^{-1}$, Table 2); this reduces the average cost to farmers/landowner to \$29.20 $ha^{-1}\,yr^{-1}$ (\$10.47 $ac^{-1}\,yr^{-1}$) representing an average reduction in cost to the farmer/landowner of about 80%.

Across the four states, the average total 50-year present value cost of a wetland per hectare is \$28,148 (\$11,396 ac^{-1}); annualized, the cost comes to \$ 1011 $ha^{-1}\,yr^{-1}$ (\$409 $ac^{-1}\,yr^{-1}$, Table 1). Illinois has the highest costs and Ohio the lowest, largely because of variable 2014 land rent in each state (Table 2). Over a 10-year period, the average total present value cost of a wetland hectare is \$12,987 (\$ 5258 ac^{-1}); spread out over a shorter time period the annualized cost comes to \$ 1438 $ha^{-1}\,yr^{-1}$ (\$582 $ac^{-1}\,yr^{-1}$, Table 1). For practices that experience annual opportunity costs, the percentage breakdown in cost type (e.g., establishment vs. management vs. opportunity cost) varies across time. For this analysis, wetland establishment costs ranged from 19% to 41% of total costs (50 and 10 years respectively), management costs ranged from 6% to 13% (50 and 10 years), and opportunity costs ranged from 46% to 75% of total cost (10 and 50 years, respectively).

For wetlands, CRP payments equal 50% of the restoration costs, a one-time sign-up bonus incentive of \$247 ha^{-1} (\$100 ac^{-1}), a 40% of establishment cost practice incentive, and average annual rental payments of \$585 ha^{-1} (\$237 ac^{-1}, Table 2). We also assumed the use of continuous CRP programming, meaning that the rental payments are renewed every decade. This represents a total CRP wetland program payment of \$910 $ha^{-1}\,yr^{-1}$ (\$368 $ac^{-1}\,yr^{-1}$), Applying these payments to the 50-year time frame reduces the cost to the farmer/landowner to \$101 ha^{-1} (\$41 ac^{-1}), representing an average reduction in cost to the farmer/landowner of about 90%.

For two-stage ditches, the total average 50-year present value cost of an average-sized two-stage ditch is \$83,076 ha^{-1}; annualized, this equates to \$ 2984 $ha^{-1}\,yr^{-1}$ (\$ 1208 $ac^{-1}\,yr^{-1}$, Table 1). Over a 10-year horizon, the total average present value cost is \$67,927 ha^{-1}(\$27,500 ac^{-1}) or \$ 7523 $ha^{-1}\,yr^{-1}$ (\$ 3046 $ac^{-1}\,yr^{-1}$, Table 1). For two-stage ditches the majority of the cost is upfront in its initial construction ranging from 74% to 91% of total costs (50 years and 10 years respectively). Opportunity costs range from 9% to 25% of total costs (10 years and 50 years respectively). Management costs are less than 1% of total costs, as the practice is generally self-sustaining [47].

USDA costs for the two-stage ditch were based on the Indiana EQIP payment rate of \$27.91 linear m^{-1} (\$8.72 $foot^{-1}$) for ditch construction [54], and an average annual CRP rental payment of \$524 ha^{-1} (\$212 ac^{-1}) plus 50% cost-share for the grass filter strip (Table 2). These payment rates resulted in a 50-year annualized total government program payment of \$ 2268 ha^{-1} (\$918 ac^{-1}), which reduces the cost to the farmer/landowner to \$716 $ha^{-1}\,yr^{-1}$ (\$289 $ac^{-1}\,yr^{-1}$) or a total reduction in farmer/landowner cost of about 76% (Table 1).

Some states have established special cost-share programs, e.g., the Conservation Reserve Enhancement Program (CREP), which provides additional funding to priority areas through a state-federal partnership [73]; therefore the potential cost-share amounts received by farmers may vary by location depending on what programs are in use. In this analysis we have only incorporated payments from CRP and EQIP as described above because we are estimating baseline costs of the most broadly used programs to the primary policy agent, the USDA. We also note that because CRP payments are determined in part upon local land prices, which vary across space and time, our estimated costs cannot be extrapolated to all situations; rather, they represent average costs under current policy and participation rates.

Table 1
Costs (in US$) associated with three agricultural best management practices under a series of scenarios.

	Practice	Total present value 10-yr cost, $\$\ \mathrm{ha}^{-1}$[a] (range of mean costs)	Total present value 50-yr cost, $\$\ \mathrm{ha}^{-1}$[a] (range of mean costs)	Annualized 10-yr cost, $\$\ \mathrm{ha}^{-1}\ \mathrm{yr}^{-1}$[b] (range of mean costs)	Annualized 50-yr cost, $\$\ \mathrm{ha}^{-1}\ \mathrm{yr}^{-1}$[b] (range of mean costs)	Mean N removal cost ($\$\ \mathrm{kg}^{-1}\ \mathrm{N\ yr}^{-1}$) over 10 years[c]	Mean N removal cost ($\$\mathrm{kg}^{-1}\ \mathrm{N\ yr}^{-1}$) over 50 years[d]
Total direct costs of practice	Wetland	12,987 (11,760–14,213)	28,148 (26,076–30,219)	1438 (1302–1574)	1011 (937–1085)	2.91	2.04
	Cover crop	1363 (1264–1463)	4846 (4493 – 5199)	151 (140–162)	151 (140–162)	7.95	7.95
	Two-stage ditch	67,927 (60,723–75,130)	83,076 (74,265–91,887)	7523 (6725–8321)	2984 (2668–3300)	11.63	4.61
Total costs to farmer/ landowner after incentives	Wetland with CRP[e]	3749 (3395–4102)	2812 (2605–3018)	415 (376–454)	101 (93–108)	0.84	0.20
	Cover crop with EQIP	262 (172–370)	930 (610–1316)	29 (19–41)	29 (19–41)	1.53	1.53
	Two-stage ditch with EQIP and CRP[e]	16,850 (15,063–18,637)	19,926 (17,813–22,039)	1866 (1668–2063)	16 (640 – 792)	2.88	1.11
Total cost of government programs	Wetland program payments [f]	9238 (8366–10,110)	25,336 (23,471–27,200)	1023 (926–1119)	910 (843–977)	2.07	1.84
	Cover crop program payments [f, g]	1101 (Not Applicable)	3916 (NA)	122 (NA)	122 (NA)	6.42	6.42
	Two-stage ditch program payments[f]	51,077 (45,660–56,494)	63,150 (56,5453–69,847)	5657 (5057–6257)	2268 (2027–2509)	8.74	3.51

[a] Average present value costs were calculated across four Corn Belt states (IA, IL, IN, OH) using a real discount rate of 1.9% in accordance with the USDA Natural Resource Conservation Service recommendations.

[b] Total present value costs were annualized using capital recovery factors for a 1.9% real discount rate (0.1107 for 10 years; 0.03116 for 50 years).

[c] The annual cost of N removal, based on the mean N removal rate of each practice and a 10-year time horizon. The N removal rate over the range of N removal values is shown in Fig. 1.

[d] The annual cost of N removal, based on the mean N removal rate of each practice and a 50-year time horizon.

[e] For the 50 year planning horizon, it is assumed that the cost share is a one-time occurrence and CRP rental payment are continuous. The four state average annual CRP rental payment used in the wetland calculations is $\$585\ \mathrm{ha}^{-1}$; the four state average annual CRP rental payment used in the two-stage ditch calculations is $\$524\ \mathrm{ha}^{-1}$.

[f] It is assumed that all program payments remain static over the time periods shown.

[g] Since EQIP payments for cover crops are fixed per unit area, total program costs do not vary.

Table 2
Summary of the current EQIP and CRP payments and general program parameters.

Best management practice	Natural Resource Conservation Service/Farm Service Agency Practice standard	Payments and/or payment structure per Hectare (acre)				
		Iowa	Illinois	Indiana	Ohio	Four state average[a]
Cover crops (cereal rye; *Secale cereal*)	NRCS Environmental Quality Incentive Program (EQIP) Practice Standard 340; for single species, chemical or mechanical termination	$96.06 ($38.89)	$145.26 ($58.81)	$103.74 ($42.00)	$142.32 ($57.62)	$121.85 ($49.33)
Restored wetland[b]	FSA Wetland Restoration Initiative Conservation Reserve Program (CRP) Practice Standards CP23, CP23A Wetland Restoration, Inside/outside the 100-year floodplain	Rent: $574 ($232)	Rent: $632 ($256)	Rent: $505 ($204)	Rent: $502 ($203)	Rent: $585 ($237)
		C/S: $251 ($101)	C/S: $258 ($104)	C/S: $99 ($40)	C/S: NA[c]	C/S: $250 ($101)
		Up to 50% cost share for wetland establishment. Annual rental payments for a 10–15-year period. The rental rate is based on the weighted average dry-land cash rent. One time, upfront CRP signing incentive payments range from $247 ha^{-1} to $370 ha^{-1} ($100–$150 ac^{-1}); for this study we use $247 ha^{-1}. One time practice incentive payment equal to 40 percent of the eligible costs of installing the practice.				
Two-stage ditches	NRCS EQIP Conservation Practice Standard 608 Surface drain, main or lateral; or Conservation Practice Standard 582 Open channel. In addition, NRCS EQIP CP 342 Critical Area Planting and NRCS EQIP CP 484 Mulching	–	–	Implementation: $27.91/linear m ($8.72/linear ft)	Implementation: $26.66/linear m ($8.33/linear ft)	$27.28/linear m ($8.53/linear ft)
		Program available in Indiana and Ohio EQIP only at time of publication. Both states pay 75% construction cost-share, with no annual rental payment. Both states pay per linear foot, meaning that the cost per hectare will change with ditch width.				
Vegetative filter strip[d]	FSA CRP Practice CP21 Filter Strips	Rent: $640 ($259)	Rent: $561 ($227)	Rent: $524 ($212)	Rent: $497 ($201)	Rent: $592 ($239)
		C/S: $98 ($40)	C/S: $83 ($34)	C/S: $85 ($34)	C/S: $78 ($32)	C/S: $91 ($37)
		Up to 90% cost share for filter strip establishment. Annual rental payments for a 10–15-year period.				

[a] For restored wetlands and vegetative filter strips, this is the weighted average of the four states, with costs weighted by area in the practice. For cover crops and two-stage ditches, it is the average of the payment rates in the participating states.
[b] All wetland cost-share and rental data were provided by the USDA, courtesy of Alexander Barbarika.
[c] Cost-share data are currently unavailable.
[d] NRCS EQIP Practice Code 393 would also apply but at a cost share rate only.

3.2. Relative cost-effectiveness

We calculated cost-effectiveness in terms of cost per kilogram of N removed (measured as $ kg^{-1} N). Using a compilation of data derived from previously published studies, wetlands and two-stage ditches had similar ranges in rates of N removal [28,57–61]. Specifically, wetland N removal rates ranged from 103 kg N ha^{-1} yr^{-1} [60] to 2310 kg N ha^{-1} yr^{-1} [57], with a mean of 495 kg N ha^{-1} yr^{-1} $\pm$93 SE. Two-stage ditch N removal ranged from 201 to 2598 kg N ha^{-1} yr^{-1}, with a mean of 647 kg N ha^{-1} yr^{-1} $\pm$186 SE. Nitrogen removal rates for both wetlands and two-stage ditches fell within the range reported from a meta-analysis of wetlands across the world [74], but were slightly higher than the range reported from riverine wetlands in the Mississippi River Basin [75]. Cover crops had the lowest annual N removal rates [63–71], and the entire range was lower than all values from wetlands and two-stage ditches, with a maximum value of 75 kg N ha^{-1} yr^{-1} [67] and a mean of 19 kg N ha^{-1} yr^{-1} $\pm$3 SE. In some studies, the presence of cover crops had no effect on the amount of N exiting a field [65,70], resulting in a minimum N removal rate of 0, but we omitted these studies because they seemed to be outliers. Although cover crops remove less N ha^{-1} of practice than the other conservation practices, they compare favorably because they are applied to a larger surface area which offsets the low removal rates.

The results of this three-practice analysis reveal that wetlands are the most cost-effective practice over an extended time period (i.e., 50 years) and cover crops are the least effective despite having the lowest direct costs. Under mean N removal rates, wetland N removal cost $2.04 kg^{-1} N and cover crops had an N removal cost of $7.95 kg^{-1} N (Table 1, Fig. 1). In general, cover crops were the least cost-effective primarily because of their low N removal rates, but when we applied the highest reported N removal rates [67], cover crops were comparable to the other practices (Fig. 1). Over the same time period, the cost effectiveness of two-stage ditches fell between the other two practices at $4.61 kg^{-1} N.

Duration also has an effect on cost-effectiveness however, so, the practices that have a higher upfront cost (i.e., higher establishment costs), such as wetlands and two-stage ditches, are less cost effective over a shorter time periods. Over a 10-year period, wetlands continue to be the most cost effective practice but at a slightly lower rate of $2.91 kg^{-1} N. The cost-effectiveness of cover crops at N removal remains the same at $7.95 kg^{-1} N because their use does not include opportunity costs over time. The cost-effectiveness of the two-stage ditch, on the other hand, is improved by 60% to $4.61 kg^{-1} N because most of its costs occur in the first year. The 10-year cost-effectiveness of wetlands is reduced relative to a 50 year time-horizon, but not as dramatically as two-stage ditches, primarily because their largest cost is the long-term opportunity cost.

4. Discussion

In this analysis, we compare costs, and the relative cost-effectiveness of three conservation practices that are used to reduce excess N export in agricultural landscapes. Importantly, cover crops, wetlands, and two-stage ditches are distinct in their landscape position, structural development, and time scales, and have unique considerations for implementation, as a result. In addition, they differ in their payment structure, N removal rates, and additional costs and benefits; thus decision-making is multi-faceted. Overall, all 3 conservation practices have particular characteristics that make them more or less desirable for particular situations. We distill these differences and provide cost information, with the intent to inform decision-making around practice implementation by farmers, natural resource managers, and policy-makers.

4.1. Comparison to previous studies

Our wetland cost estimates are within the range previously reported for the Corn Belt. Iovanna et al. [18] reported a mean cost-effectiveness of $3.04 kg N^{-1} to the USDA and Iowa CREP. In comparison, we estimated an average of $1.84 kg N^{-1} (over 50 years) to $2.07 (over 10 years) to USDA programs. Our slightly lower costs reflect programmatic differences; CREP includes CRP payments plus additional state-funded incentives, whereas the WRP has a similar structure as CRP. Similarly, Hansen et al. [17] reported cost-effectiveness ranging from $0.51–$0.68 kg N^{-1}for wetlands in northwest Ohio, whereas we estimated $2.04 kg N^{-1} (over 50 years) to $2.91 kg N^{-1} (over 10 years). These differences are likely regional; Hansen et al. [17] focused on a single county, whereas our study applies generally to the Corn Belt. Indeed, our results are in good agreement with a study of wetlands in the upper Mississippi River and Ohio River watersheds, which found a range of $0.07–$10 kg N^{-1} [76]. Compared to studies with a larger, macroeconomic perspective, our costs are generally lower. Doering et al. [15] reported wetlands costs ranging from $3.85–$13.12 kg N^{-1}, while Ribaudo et al. [16] reported $3.65–$4.82 kg N^{-1}. Our lower costs are a reflection of the different approaches; the macroeconomic studies examine the wide-ranging impact of large-scale wetlands restoration, whereas our study focused on the costs at the scale of a single farm. Despite the variety of approaches used to assess wetland cost-effectiveness, all are in general agreement, with costs usually less than $10 kg N^{-1} removed, and most ranging from $1.50–$5.00 kg N^{-1}.

Cover crops and two-stage ditches have not had cost-effectiveness estimates previously reported, but wetlands have been compared to other N management practices, including fertilizer reduction, on-field fertilizer management, and grass buffer strips [15–17]. The result is generally scenario-specific, with wetlands being more cost-effective in some situations, and on-field nitrogen management being more cost-effective in others. Wetlands compared favorably to grass buffer strips, with N removal in poorly-functioning buffer strips costing over 10 times as much per unit N as wetlands [17]. Similarly, wetlands were generally the most cost-effective practice in our study.

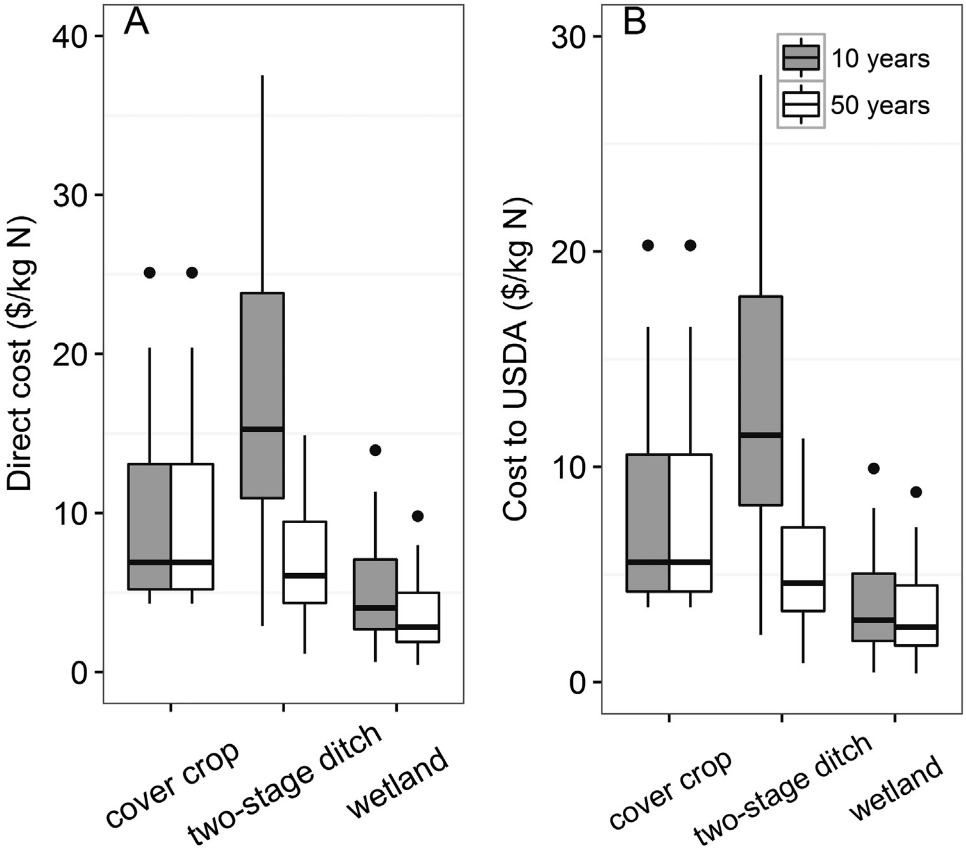

Fig. 1. (A) The direct cost per unit N removed for cover crops, wetlands, and two-stage ditches, and (B) the cost to the USDA for implementing these practices under current USDA cost-share programs. The upper and lower hinges on the boxplots represent the 75th and 25th percentiles, and the lines extend to 1.5 × the inter-quartile range. Cover crops and two-stage ditches are funded by the Environmental Quality Incentives Program (EQIP), and wetlands are funded by the Conservation Reserve Program (CRP).

4.2. *Choice of conservation practice*

Wetlands, cover crops, and two-stage ditches vary in the time-scale over which they remove N, their relative placement on the landscape, and the conditions under which they function best. Although wetlands were the most cost-effective practice, other considerations may also influence implementation decisions.

Cover crops are planted and removed once each year, so they have the lowest direct costs, they incur little to no structural changes, and they do not involve the same kind or degree of opportunity costs as the other practices. The decision on whether or not to implement cover crops can be made each year, and it does not remove any land from production, although it can influence the timing of spring planting [24]. In contrast, restored wetlands involve structural development (e.g., earth movement, use of engineered structures) and are generally considered semi-permanent or permanent, long-term investments and CRP contracts require that they be left in place for at least 10–15 years. The two-stage ditch is the longest-term, and likely permanent, investment given that it is self-maintaining practice [47]. Additionally, EQIP requires implementation for at least 15 years. Furthermore, the initial construction cost of the two-stage ditch is the highest, and so as noted prior, it is more cost-effective over longer time horizons.

These three practices also vary in their landscape position. Cover crops are a field-scale practice applied in the off-season to reduce N loss from fields, and they could be easily implemented at the watershed scale. In contrast, wetlands are located at field edges, in the riparian zone, and two-stage ditches are a modification to stream/ditch channels themselves (Fig. 2). We have presented cover crops, wetlands, and two-stage ditches as 3 independent options for N reduction from agricultural landscapes, but they are not mutually exclusive or universally applicable. For example, not all farmers own or manage land appropriate for wetland restoration or two-stage ditches, nor may they be willing to remove land from production for implementation. On the other hand, these practices may be compatible for simultaneous usage on the same farm system or watershed. In fact, differences in field placement suggest that they are complementary and could be implemented simultaneously, in a “stacked” practice configuration (Fig. 2), which would potentially maximize the strengths of each individual practice, to work either additively or synergistically. For example, cover crops reduce winter N loss from fields, wetlands are effective at reducing leached N at base flow conditions, and the two-stage

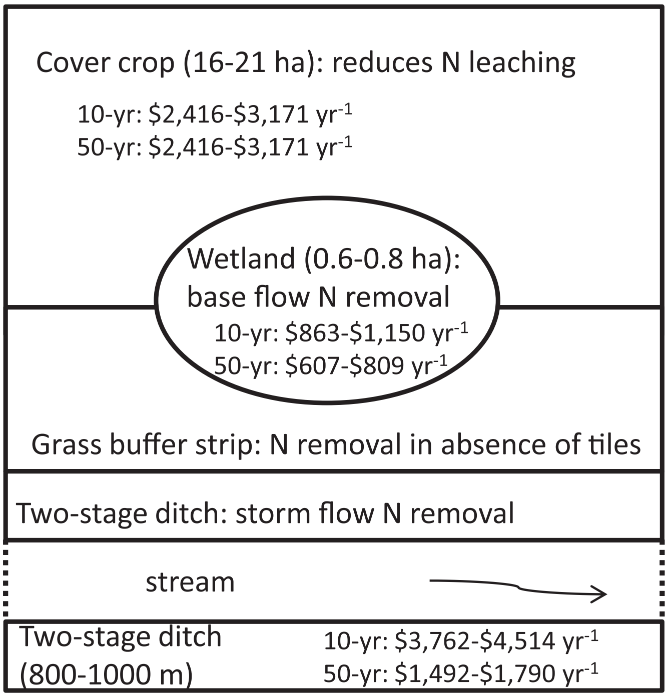

Fig. 2. Placement of 3 comparable agricultural conservation practices that are used to mitigate nitrogen pollution. The costs are the annual costs over 10 and 50 years, to achieve a 30% reduction (first number) to 40% reduction (second number) in N export, from a 100 ha field. We assume a N export rate of 1000 kg km^{-2} yr^{-1}. The area or length required to achieve a 30% and 40% reduction in N export, under mean N removal rates, is noted for each practice.

ditch reduces N export during floodplain inundation.

Unfortunately, there are insufficient N-removal data available to make any conclusions about the effectiveness or optimal placement of these practice combinations. We expect that total N removal from a particular field (kg N/ha drained) will be greater with multiple practices than with a single practice, and that their combined effect would be additive or even synergistic during times of N saturation. Saturation occurs when the biological capacity of a system is overwhelmed; when increases in N concentration do not increase the N removal rate, and it occurs regularly in agricultural ecosystems, particularly in the late winter or early spring [77]. Thus, although cover crops will reduce the N concentration in drainage water, if the N concentration remains above saturation, the wetland N removal will remain as high as in a system without cover crops. The practices could be synergistic if cover crops also slowed water flow (plant uptake can slow drainage rates), so that there was more time for N removal in the wetlands. Under non-saturating conditions, the combined N removal of multiple practices would likely not be additive or synergistic, but would still be greater than a single practice, effectively “topping off” N removal. We emphasize that there are few empirical studies quantifying this, and research on the N removal capacity of stacked conservation practices will help to refine and test these hypotheses.

All three practices varied substantially in their N removal rates, and maximum N removal occurs under different conditions for each practice. Cover crop N retention is highest as winter soil temperatures warm in spring, and also with the seasonal duration of cover crop planting [63,71]. During cold winters, cover crops grow more slowly and retain less N, or are winter-killed, rendering them less effective [23]. In contrast, wetlands and two-stage ditches are most strongly influenced by hydrology. During high flows (i.e., storms), wetlands can be overwhelmed by drainage water, resulting in a flow-through system with minimal influence on N loads [30]. Two-stage ditches however, are most effective during floodplain inundation, when water can interact with the increased surface area provided by bioreactive floodplains [28].

4.3. *Cost of meeting water quality goals*

In our initial analysis, we compared practices by standardizing their N removal and estimating cost rates, but a broader application is to examine the area and associated costs needed for each practice to improve environmental outcomes surrounding compromised water quality. For example, in order to reduce the recurrence and size of the hypoxic “dead zone” in the Gulf of Mexico, N inputs must be reduced by an estimated 30% [10] to 40% [75,78]. We assume here that the non-agricultural N sources, which account for ~10% of the total N load [79], are being simultaneously addressed, and thus the full burden of N reduction does not fall solely on agriculture. Thus, in order to meet these water quality goals, a 100 ha (1 km^2) field that typically exports 1000 kg N km^{-2} yr^{-1} (a reasonable amount for the agricultural Midwest [5]) would have to reduce its N exports by 300-400 kg N yr^{-1}. Under mean N removal rates, that reduction would require 16–21 ha of cover crop, 0.6–0.8 ha of wetland, or 0.5–0.6 ha (800–1000 m), of two-stage ditch. Over 50 years, the total present value cost would be highest with cover crops ($77,536 to $101,766 or $ 2416 to $ 3171 yr^{-1}), medium with the two-stage ditch ($41,538 to $49,846 or $ 1492 to $ 1790 yr^{-1}), and least expensive with a wetland ($16,889 to $22,518 or $607 to $809 yr^{-1}, Fig. 2). Over 10 years, in present value terms, it would cost the most to achieve this reduction with a two-stage ditch ($33,964 to $40,756 or $ 3762 to $ 4514 yr^{-1}), followed by cover crops ($21,808 to $28,623 or $ 2416 to $ 3171 yr^{-1}), and it would be least expensive with a wetland ($ 7792 to $10,390 or $863 to $ 1150 yr^{-1}, Fig. 2).

This generalized goal-based approach resulted in the same pattern of cost-effectiveness as in our initial analysis, with wetlands being the most cost-effective and either the two-stage ditch or cover crops being the most expensive, depending on the time horizon. This approach provides an approximate link between farm-scale practice implementation and regional-scale water quality goals (i.e., reduction of the hypoxic zone), and demonstrates the potential for widespread water quality benefits from systematic practice implementation [80].

4.4. Other costs and benefits

In addition to the direct implementation costs and N removal benefits, there are other relevant ecosystem-scale costs and benefits associated with practice implementation. In aggregate, these costs and benefits include the effects of land retirement on crop prices [16] and the total market and non-market values. From a benefit-cost perspective, it is important to recognize that the array of benefits (market and non-market) resulting from these practices are jointly produced and can aggregate across a landscape [81].

Wetlands not only help capture and treat excess N, but they also provide sinks for excess sediments and phosphorus [82,83]. Wetlands also directly provide landscape-scale biodiversity (e.g., migratory habitat), aid in groundwater recharge, and enhance flood protection [84]. The collective economic value of these types of wetland-mediated benefits can be quite significant depending on the scale and relative scarcity of wetlands at the landscape level [84,85]. In the context of restored wetlands in the Mississippi Alluvial Valley, Jenkins et al. [86] noted that the estimated social value of ecosystem service-related benefits surpasses the cost of wetlands restoration after just one year, and they continue to accumulate value well into the future. At watershed to landscape scales, cover crops and two-stage ditches likely enhance habitat in important ways. For example, cover crops provide winter browse and cover for wildlife [24] and two-stage ditches provide non-cultivated riparian habitat, which may be important for grassland birds, upland game, other land mammals and amphibians, although neither the magnitude of these effects nor their economic value have been estimated. In addition, two-stage ditches provide flood protection through their additional water storage capacity [33], although their effects on flood mitigation and economic contribution have not yet been estimated.

In addition to benefits that can manifest at broader spatial scales (e.g., watersheds), the conservation practices highlighted by this study have the capacity to provide many direct and indirect benefits that are privately experienced. Cover crops provide a broad array of crop production benefits, including long-term improvements in soil health, marketable biomass, carbon storage, erosion control, weed and pest suppression, *arbuscular mycorrhizal fungal colonization, and beneficial insect conservation* [87]*; benefits that may lead to improved crop yields* [24,72]. Wetlands have been well-noted for their contribution to waterfowl habitat and the economic value of enhanced private hunting and or birdwatching opportunities [86,88]. For the two-stage ditch, farmers receive a direct, monetary benefit from the elimination of recurring ditch clean-out costs, which are usually completed at least every 5–10 years. In fact, under some scenarios, installation of a two-stage ditch resulted in a net monetary gain for farmers even in the absence of USDA payments; in these cases the savings in ditch clean-outs exceeded the cost of two-stage ditch construction [46].

5. Conclusion

The vast majority of the US Corn Belt's agricultural land is privately managed by hundreds of thousands of individuals. Therefore the regional capacity to mitigate water quality impacts from row-crop agricultural systems will be dependent on collective, private, environmental management implemented at the farm scale [89]. For farmers, the decision to adopt a particular practice is fairly complex and contingent upon a number of biophysical, social, land-tenure oriented, and economic factors [90]; factors that vary in influence across individuals. Nevertheless, financial considerations are universally influential in farmer decision-making. Detailed cost information helps farmers determine the compatibility of the practice for their current operation, and cost-effectiveness information helps managers and policy-makers gauge the financial feasibility of the practice relative to farmer willingness to pay, and to alternative management options. Furthermore, it is important for financial information to be transparent in terms of inputs and outputs, assumptions, analytical methods, and time frame so as to broadly inform practice consideration and facilitate periodic analytical updates [21].

Relatedly, because government programs can be critical in facilitating the adoption process by reducing financial and informational burdens for farmers, agencies also need to provide comprehensive cost and baseline cost-effectiveness information. Conservation funding is often in flux and is expected to decline as per the 2014 US Agricultural Act [91]. Therefore, to achieve the maximum conservation benefit from limited funds, the individual practices and management strategies must be biophysically- and cost-effective. Although data for baseline cost and cost-effectiveness are scarce in the literature [42,92], these data are critically needed by agencies in order to: 1) inform benefit-cost oriented targeting to gauge priority programming; 2) better understand the impact of priority programming in reducing the financial burden to farmers; and 3) improve the efficiency and overall impact of conservation planning. Furthermore, as research and policy evolves toward more systematic, targeted conservation approaches [93], comprehensive cost information regarding specific conservation practices at the farm-scale will be required to test benefit-cost optimization and other planning goals [42].

Complex questions remain regarding the effectiveness of widespread practice implementation at watershed scales [35,94]. In particular, lag times in water quality response to practices [95] and relatively low adoption rates of practices in key watershed locations, continue to challenge regional policy initiatives [96]. Nevertheless, the financial information provided here makes progress toward a comprehensive understanding of the more pragmatic aspects of conservation practices, which is a critical component of the nutrient reduction strategies currently framing the regional response to the Mississippi River Basin's growing water quality conundrum.

Acknowledgments

We thank Alexander Barbarika of the USDA for providing recent CRP payment data, LeRoy Hansen and Skip Hyberg of USDA, two anonymous reviewers and Associate Editor Celine Nauges for valuable comments on earlier versions of this manuscript. Molly Lipscomb, Leigh Raymond, Geoff Kramer and Troy Bowman provided helpful discussions. S.S. Roley was funded by the Arthur J. Schmitt Foundation and by GLOBES, NSF-IGERT award # 0504495 to the University of Notre Dame. The GLOBES Program and the Notre Dame Environmental Change Initiative also provided an excellent forum for discussion of these ideas during SSR's graduate training.

Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.wre.2016.06.003.

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