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# Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage

Laura E. Christianson
*Iowa State University*

John C. Tyndall
*Iowa State University*, jtyndall@iastate.edu

Matthew J. Helmers
*Iowa State University*, mhelmers@iastate.edu

**Abstract**
Much work has been invested in the development of practices and technologies that reduce nitrate losses from agricultural drainage in the US Midwest. While each individual practice can be valuable, the effectiveness will be site specific and the acceptability of each approach will differ between producers. To enhance decision making in terms of water quality practices, this work created average cost effectiveness parameters for seven nitrate management strategies (controlled drainage, wetlands, denitrification bioreactors, nitrogen management rate and timing, cover crops, and crop rotation). For each practice, available published cost information was used to develop a farm-level financial model that assessed establishment and maintenance costs as well as examined financial effects of potential yield impacts. Then, each practice's cost values were combined with literature review of N reduction (% N load reduction), which allowed comparison of these seven practices in terms of cost effectiveness (dollars per kg N removed). At −\$14 and −\$1.60 kg $N^{-1}$ $yr^{-1}$, springtime nitrogen application and nitrogen application rate reduction were the most cost effective practices. The in-field vegetative practices of cover crop and crop rotation were the least cost effective (means: \$55 and \$43 kg $N^{-1}$ $yr^{-1}$, respectively). With means of less than \$3 kg $N^{-1}$ $yr^{-1}$, controlled drainage, wetlands, and bioreactors were fairly comparable with each other. While no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety, this analysis provides measures of average cost effectiveness across these seven strategies that allows direct comparison.

**Keywords**
Natural Resources Ecology and Management, Nitrate, Drainage, Water quality, Cost effectiveness

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Laura Christianson [a,b,*], John Tyndall [c], Matthew Helmers [d]

[a] Department of Agricultural and Biosystems Engineering, Iowa State University, Ames, IA 50011, USA
[b] The Conservation Fund Freshwater Institute, Shepherdstown, WV 25443, USA
[c] Department of Natural Resources Ecology and Management, 238 Science II, Iowa State University, Ames, IA 50011, USA
[d] Department of Agricultural and Biosystems Engineering, 219B Davidson Hall, Iowa State University, Ames, IA 50011, USA

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Much work has been invested in the development of practices and technologies that reduce nitrate losses from agricultural drainage in the US Midwest. While each individual practice can be valuable, the effectiveness will be site specific and the acceptability of each approach will differ between producers. To enhance decision making in terms of water quality practices, this work created average cost effectiveness parameters for seven nitrate management strategies (controlled drainage, wetlands, denitrification bioreactors, nitrogen management rate and timing, cover crops, and crop rotation). For each practice, available published cost information was used to develop a farm-level financial model that assessed establishment and maintenance costs as well as examined financial effects of potential yield impacts. Then, each practice's cost values were combined with literature review of N reduction (% N load reduction), which allowed comparison of these seven practices in terms of cost effectiveness (dollars per kg N removed). At −$14 and −$1.60 kg N$^{-1}$ yr$^{-1}$, springtime nitrogen application and nitrogen application rate reduction were the most cost effective practices. The in-field vegetative practices of cover crop and crop rotation were the least cost effective (means: $55 and $43 kg N$^{-1}$ yr$^{-1}$, respectively). With means of less than $3 kg N$^{-1}$ yr$^{-1}$, controlled drainage, wetlands, and bioreactors were fairly comparable with each other. While no individual technology or management approach will be capable of addressing

* Corresponding author at: The Conservation Fund Freshwater Institute, Shepherdstown, WV 25443, USA.
Tel.: +1 304 870 2241.
*E-mail address:* l.christianson@freshwaterinstitute.org (L. Christianson).

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## 1. Introduction

Artificial subsurface drainage systems in the Midwestern "Corn Belt" region have allowed for increased productivity over the past century [1], but nitrate ($NO_3^-$) losses in drainage have caused significant multi-scale environmental concerns [2,3]. Much work has been done developing and advancing practices to reduce $NO_3^-$ losses in subsurface agricultural drainage. Dinnes et al. [1] provide a comprehensive review of $NO_3^-$ reducing technologies for the Midwest including in-field "preventative" N strategies (e.g., N management, cover crops, diversified rotations) and "remedial" strategies for N removal from drainage (e.g., controlled drainage, bioreactors, wetlands). While each strategy and individual practice can be valuable, the $NO_3^-$ removal effectiveness will be site specific and the acceptability of each individual approach will differ between producers. Nevertheless, no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety [1,4]; as such, a suite of approaches used across these landscapes will be required [5].

On an individual farmer basis, adoption of environmental management practices designed to mitigate or prevent issues such as $NO_3^-$ losses through drainage to surface waters are motivationally different from production innovations largely because short-term economic advantages of adopting a mitigation technology are rare [6,7]. Farm level action involving use of technology is in large part influenced by owner and operator beliefs and attitudes (i.e., regarding environmental and financial risk) in combination with personal environmental goals and knowledge about technology [8]. Perceptions of a technology in turn are shaped by external factors such as cost, overall complexity and effectiveness of the available technology, and available technical/financial support [9,10]. As such, crop producers require comprehensive information about water quality technologies with regard to the context for use, operational parameters, performance efficacy, and the full range of financial parameters (e.g., upfront and long-term costs). Of particular and universal concern for farmers is the financial feasibility of a particular technology in the context of their production system, as well as comparative advantage across technology-based management options. Moreover, comprehensive financial information is needed to calibrate agricultural conservation cost-share programming and targeting and to better guide federal and state technical service provision at county levels [4].

To enhance land-use decision making, this work investigates and makes transparent the financial parameters of seven $NO_3^-$ management strategies; three are remedial N strategies: controlled drainage, wetlands, denitrification bioreactors and four are preventative N strategies: N rate reduction, spring N application, cover crops, and crop rotation. It bears to note early-on; however, that the Midwest is a heterogeneous region where not every abatement strategy will be equally appropriate (i.e., costly or effective) in any given situation. Suitability, in addition to $NO_3^-$ reduction effectiveness, can vary by soil type, topography, landscape position, and microclimate (e.g., rainfall patterns, winter severity) for each of the seven distinct practices investigated here. For example, winter cover crops may be more difficult to establish in northern Minnesota vs. southern Indiana, and controlled drainage will be most cost effective on flatter topographies. The assumed baseline cropping system for this work was a corn/soybean rotation, reflective of the Midwestern agricultural landscape [11], and because tillage generally has a relatively small impact upon tile drainage $NO_3^-$ export [12], it was not included as a variable here.

Controlled drainage (also known as drainage water management) is a strategy that addresses agricultural $NO_3^-$ loading through the use of a series of structures installed in drainage pipes or drainage ditches that allow control of the water table depth [13,14]. Though this practice can be used to achieve agronomic and/or environmental objectives [14], a major limitation is that controlled drainage becomes more expensive on slopes greater than 0.5–1% [1,15]. The second practice under consideration here, denitrification or woodchip bioreactors, uses control structures to regulate drainage water flowing through an excavation (typically $>30$ m long, $>1$ m wide) filled with a carbon source allowing enhanced denitrification of the $NO_3^-$ in the drainage water [16,17]. These systems have been tested for treating drainage from "field-sized" areas of approximately 20 ha and usually require very little to no land to be removed from production by fitting in grassed edge-of-field areas [17]. The third of the remedial strategies, constructed wetlands, is a long-term $NO_3^-$ reduction strategy intended for watershed-scale treatment [18,19]. A key consideration for N removal in wetlands is the wetland to treatment area ratio with increased N removal possible at increased wetland: watershed area ratios [18,20–22].

Regarding in-field, preventative practices, N fertilizer management, here in terms of rate and timing, is one of the farm operator-controlled factors to reduce N losses in agricultural drainage [1,12,23,24]. Water quality benefits of reduced application rates will be a function of the original and the modified rate [25,26]. Lawlor et al. [27] proposed that a corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.) rotations can be described with:

$$\text{N Concentration in Drainage} = 5.72 + 1.33e^{(0.0104 \times N\ Rate)} \tag{1}$$

where N concentration is in mg N $L^{-1}$ and rate is in kg N applied $ha^{-1}$ [27]. Spring N application in the U.S. Midwest more closely synchronizes the application with plant uptake [28,29], an outcome that is preferable from both water quality and agronomic perspectives [24]. Nevertheless, fall N applications are a way to manage risk associated with uncertain spring weather and spring-time field activities [30].

The "preventative" strategy of winter cover crops such as rye, oat, winter wheat, brassica, or winter-hardy legumes, utilizes plant uptake as the major water quality improvement mechanism [31,32]. Benefits of cover crops (as well as several of the other practices) extend beyond drainage water $NO_3^-$ reduction (e.g., erosion control, pest control, enhancement of soil productivity) [29,33] but were not included here as this analysis focuses solely on $NO_3^-$ reduction; see Table 1 for abbreviated comments and Christianson et al. [34] for a broad discussion of ecosystem services associated with the use of any of these seven practices. The main limitations of winter cover crops are that they need to grow well under non-ideal conditions [1,32], some need to be killed before planting the main crop, and a corn yield reduction following certain covers is possible [31,32]. The final practice, crop rotations that include perennials, similarly provides water quality benefits via N and water uptake [1,35] and additional benefits to the soil [36]. Although the main limitations for this sort of rotation include access to markets, crop storage, and additional machinery requirements, Dinnes [29] reported diversifying cropping systems in Iowa has the most potential to reduce $NO_3^-$ loadings compared to any other best management practice.

The objectives of this exploratory financial assessment are two-fold: (1) characterize and quantify the financial (cost) parameters of the seven $NO_3^-$ reduction strategies; and (2) explore and compare the average cost efficiency of each strategy (dollars per kg N removed) using published measures of N reduction effectiveness. The primary motivation of this work is that while cost assessment of this type is fairly straight-forward, cost comparison analysis across various agricultural best management practices is invariably challenged [37] by (1) limited availability of published cost information, (2) variable methodology in published financial assessments, (3) limited methodological transparency in published cost assessments, (4) variable discount rates, (5) inconsistent analysis horizons due to variable life spans or management horizons, and (6) many costs are often site specific and therefore can exhibit significant ranges. This analysis is therefore an attempt to make transparent the structure and timing of cost parameters associated with using any of these $NO_3^-$ management strategies, and to develop comparable measures of average cost effectiveness across these seven $NO_3^-$ management strategies. Nevertheless, we recognize an inherent limitation of this work arises from the site-specific nature of the practices being compared; their application at different sites and under different conditions will necessarily confound a comparison of their effectiveness in reducing N loads and hence their calculated cost efficiencies.

**Table 1**
Description of the scenarios, uncertainty ranges for the Total Present Value Costs, and the additional benefits and costs that were not quantified for seven nitrogen reduction practices for agricultural drainage; see Christianson et al. [34] for more specific discussion of ecosystem services of these practices.

| Practice | Practicable lifespan (yr) | Specific scenario | Uncertainty of ranges for TPVC | Unquantified costs and benefits |
|---|---|---|---|---|
| Controlled drainage | 40 | 1 structure per 4 ha–8 ha | Low uncertainty | Potential yield impacts<br>Potential increase in soil erosion, soil compaction, or surface runoff |
| Bioreactor | 40 | 20.2 ha field treated with a 0.1 ha bioreactor | Low uncertainty | None |
| Wetland | 50 | 405 ha treated by a 4 ha wetland plus buffer | Moderate uncertainty due to predominance of land cost and the variability of this factor | Additional ecosystem services including pollination, wood fuel, ornamental resources, natural hazard regulation, and recreation |
| N rate reduction | 1 | 168 kg N $ha^{-1}$–140 kg N $ha^{-1}$ | Large uncertainty due to yield impact variability | Probabilistic variability of yields |
| N spring application | 1 | Apply N in spring instead of fall | Large uncertainty due to unquantified risk and yield impact variability | Cost of infrastructure potentially required for fertilizer storage, handling, etc.<br>Probabilistic variability of yields<br>Potential loss of yield by a delayed planting date |
| Cover crop | 4 | Rye drilled | Large uncertainty as this practice is primarily implemented for reasons other than N reduction and due to yield impact variability | Additional ecosystem services including pollination and erosion and pest regulation;<br>Potential future yield enhancement due to cover crop-induced soil quality and organic matter enhancement |
| Rotation | 10 | 3 years alfalfa, 2 years corn | Very large uncertainty due to rotation complexity and the variability of alfalfa-induced yield increase | Additional ecosystem services including pollination and erosion and pest regulation;<br>Potential future yield enhancement due to perennial-induced soil quality and organic matter enhancement |

## 2. Materials and methods

There is limited availability of published cost information regarding drainage $NO_3^-$ reduction strategies, and the variable methodology and limited transparency for the studies that have been done in this area make comparison between published analyses difficult. The timing of costs particularly complicates comparisons of water quality practices. For example, controlled drainage, bioreactors, and wetlands all have large initial capital outlays and intermittent management costs, while N management, cover crops, and crop rotations largely involve variable annual costs. Cost assessments have been carefully constructed for all seven practices with itemized cost parameters and unit cost data for each strategy collected from various secondary sources (e.g., published literature, published custom rate surveys, and when necessary personal communication with knowledgeable individuals). Total present value costs (TPVCs) were assessed with a discounted cost model that aggregates total fixed and variable costs.

$$\mathrm{TPVC}_{\mathrm{practice}} = C_{\mathrm{est,practice}} \text{ in year } 1 + C_{\mathrm{main}} \text{ occuring over n years} \tag{2}$$

where $\mathrm{TPVC}_{\mathrm{Practice}}$ is the total present value of the cost of a practice, $C_{\mathrm{est,Practice}}$ is the full establishment cost, and $C_{\mathrm{main}}$ involves all annual and/or periodic maintenance costs of the practice applicable for and discounted over n years. The specific variations of this general model for each individual technology are presented in Supplemental material.

To develop a range of costs for each practice, minimum and maximum values for each individual cost category were summed to develop a minimum and maximum TPVC, respectively (Tables 2–7). If only a single value (i.e., mean) was available for a cost, this value was used in both the minimum and maximum TPVC calculation for that practice. As is appropriate for this type of cost comparison assessment (e.g., [38–41]), the minimum and maximum TPVCs for each practice were then used to develop a range of equal annual costs (EACs) for the strategies (Table 9). The EAC approach involves determining the equal annual payment (in present value terms) that would be made at the end of each year to fully cover costs over a planning horizon, and allows for the direct comparison of total present value costs from practices that have different practicable life spans [42]. More pragmatically, the EAC format allows farm-level decision makers to consider environmental best management practice costs essentially on a similar basis that they consider typical farm-level production costs [43].

Following Burdick et al. [44] and Tyndall and Grala [45], conversion to EACs was done using a capital recovery factor (CRF):

$$\mathrm{EAC} = \mathrm{TPVC} \times \mathrm{CRF} \tag{3}$$

where TPVC is the total present value of the cost of the practice and the CRF is calculated using:

$$\mathrm{CRF} = \frac{i(1+i)^n}{(1+i)^n - 1} \tag{4}$$

where $i$ is the annual real discount rate and $n$ is the number of years in the evaluation (i.e., planning horizon). The analysis was carried out using a 4% real discount rate, and the $n$ was set to each practice's individual practicable lifespan (Table 1). A 4% discount rate represented the average real interest rate on Iowa farmland loans during 2008–2010 and was very similar to the 2011 rate for federal water projects (4.125%) [46].

Calculated EACs were combined with published measures of $NO_3^-$ removal efficacy (% load reduction; Table 8) to develop an average efficiency parameter of dollars per kg N removed. This literature review-based approach (as opposed to a more site-specific modeling approach, which was outside the scope of this financial parameterization work) allowed capture of some inherent variability as the literature contains observations across sites and conditions. Dividing the EAC of each strategy by the amount of $NO_3^-$-N removed is a standard way to present total costs per unit e.g., [44,47]. To do so, a Midwestern-representative load of 31.4 kg N $ha^{-1}$ was developed from an average of Jaynes et al. [48] tile and drain N loads and Lawlor et al. [49] drainage N loads at their 168 kg N $ha^{-1}$ application rate. Then, the minimum and maximum EAC for each practice were each applied to that practices' range for $N$ load reduction (mean, median, 25th, and 75th percentiles from Table 8 and Fig. 1

**Table 2**
Itemized costs and Total Present Value Costs for controlled drainage in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 40 years.

| Item | Cost timing (yr) | Minimum cost ($ ha$^{-1}$) | Mean cost ($ ha$^{-1}$) | Maximum cost ($ ha$^{-1}$) | Notes and assumptions | Reference |
|---|---|---|---|---|---|---|
| Structure cost | 1 | $61.78 | | $247.11 | New drainage system: 1 structure per 8 ha at $500–$2000 per ea. | [15] |
| | 1 | $123.55 | | $494.21 | Existing drainage system: 1 structure per 4 ha at $500–$2000 per ea. | |
| Transport structures | – | | | | Assumed included above | Assumption |
| Design cost | 1 | | $80.63 | | For new drainage systems but also included as design cost of existing | [100] |
| Contractor fees | 1 | $4.32 | $9.47 | $15.44 | Structure installation: Back hoeing at $35.00 h$^{-1}$, $76.65 h$^{-1}$, $125.00 h$^{-1}$ for 8 h to treat 65 ha | [81] |
| **Total cost of establishment** | | **$146.73** | | **$343.18** | **New (TPVC)** | |
| | | **$208.51** | | **$590.29** | **Existing (TPVC)** | |
| Time to raise/lower | $1-n$ | $0.99 | | $4.94 | Four hours × two to four times a year; labor at $8–$20 h$^{-1}$, 65 ha treatment area | [81] |
| Stop log/gate replacement | 8, 16, 24, … | $17.67 | | $35.34 | Summation of single sum TPV every eight years for 5 gates per structure at original cost of $14.17–$15.32 per ea. for 15 cm structures, 1 structure per 4 (Existing) or 8.1 (New) ha | [101] |
| **Total cost of establishment, maintenance, and replacement** | | **$183.96** | | **$723.44** | **TPVC** | |

**Table 3**
Itemized costs and Total Present Value Costs for a denitrification bioreactor in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 40 years.

| Item | Cost timing (yr) | Minimum cost ($ ha$^{-1}$) | Mean cost ($ ha$^{-1}$) | Maximum cost ($ ha$^{-1}$) | Notes | Reference |
|---|---|---|---|---|---|---|
| Both control structures | 1 | $49.42 | | $197.68 | Two control structures at $500–$2000 ea.; 20.2 ha treatment area | [101] |
| Structure transport | – | | | | Assumed included above | Assumption |
| Woodchip cost | 1 | | $116.14 | | Two semi loads at $975 chips + $200 transport ea.; 20.2 ha treatment area | [102] |
| Woodchip transport to farm | – | | | | Included above | |
| Design cost | 1 | $0.00 | | $31.63 | Assumed: $40 h$^{-1}$ for 2 days of work or NRCS service provider; 20.2 ha treatment | Assumption |
| Contractor fees | 1 | $27.68 | $60.61 | $98.84 | Back hoeing at $35.00 h$^{-1}$, $76.65 h$^{-1}$, $125.00 h$^{-1}$ for 16 h to treat 20.2 ha | [81]; Assumptions |
| Seeding bioreactor surface | 1 | $0.05 | $0.11 | $0.15 | Seeding grass, broadcast with tractor; for 20.2 ha treatment and 0.10 ha bioreactor at $9.88, $22.61, and $29.65 h$^{-1}$ | [81] |
| Seed cost | 1 | | $1.11 | | Seed costs from dealer: $222.27 ha$^{-1}$ for CRP Mix (CP23) Diversified mix; bioreactor surface 0.005 of treatment area | [82] |
| Misc. materials | 1 | | $8.80 | | 6″ tile $890 per 305 m(1000 ft); Assume 61 m needed for control structure connections for 20.20 ha treatment area | [101] |
| **Total cost of establishment** | | **$203.19** | | **$454.35** | **TPVC** | |
| Time to raise/lower | $1-n$ | $1.19 | | $2.97 | Three hours per yr with farm labor wages at $8–$20 h$^{-1}$, 20.2 ha treatment area | [81]; Assumption |
| Mowing/maintenance | $1-n$ | $0.12 | | $0.62 | Spot mowing bioreactor at $24.71–$123.55 ha$^{-1}$ for 20.2 ha treatment | [83] |
| Replacement year 20 | 20 | $65.66 | | $98.18 | Single sum TPVC at 20 years: woodchips, contractor, seeding | Assumption |
| Gate replacement | 8, 16, 24,… | | $14.14 | | Summation of single sum TPV every eight years for 5 gates per structure ($14.17–$15.32 per ea. for 15 cm structure) 2 structures per 20.2 ha | [101] |
| **Total cost of establishment, maintenance, and replacement** | | **$308.91** | | **$637.59** | **TPVC** | |

**Table 4**
Itemized costs and Total Present Value Costs for a wetland in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 50 years.

| Item | Cost timing (yr) | Minimum cost ($\$ ha^{-1}$) | Mean cost ($\$ ha^{-1}$) | Maximum cost ($\$ ha^{-1}$) | Notes | Reference |
|---|---|---|---|---|---|---|
| Design cost | 1 | | $71.17 | | Assumed: $40 $h^{-1}$ for 90 days of work (8 h $d^{-1}$) for 405 ha site | Assumption |
| Contractor fees | 1 | $28.17 | $34.43 | $41.51 | Building ponds at 8 h $d^{-1}$ for 15 days with Custom Rate Survey $ $h^{-1}$ for 405 ha wetland , not including seeding time | [81] |
| Seeding buffer | 1 | $0.35 | $0.79 | $1.04 | Tractor broadcasting at $9.88, $22.61, or $29.65 $ha^{-1}$ for 14 ha wetland buffer for 405 ha treatment | [81] |
| Seed cost | 1 | $7.43 | | $95.38 | Seed costs from dealer: $212.39 $ha^{-1}$ for CRP wetland program mix to $162.09 $kg^{-1}$ for "wetland seed mix" at needed 16.8 kg $ha^{-1}$ | [82,84] |
| Weir plate | 1 | | $14.83 | | $30 per sq ft. for 40 ft width × 5 ft sheet pile plate, for 405 ha site | Assumption |
| Control structure | 1 | $3.26 | | $7.25 | One large control structure ($1320–$2935 per ea.), for 405 ha site | [101] |
| Land acquisition | 1 | $529.08 | | $679.31 | $11,757–$15,095 $ha^{-1}$ for 4 ha wetland plus 14 ha buffer treating 405 ha; 2010 state-wide Iowa average for high and medium grade lands | [85] |
| **Total cost of establishment** | | **$654.28** | | **$910.48** | **TPVC** | |
| Time to manage | $1-n$ | $0.09 | | $0.43 | Spot mowing 10% of buffer area at $24.71–$123.55 $ha^{-1}$ | [83] |
| Control structure and weir replacement | 40 | $4.55 | | $5.75 | Single sum TPVC at year 40 includes costs of a new structure and weir and 16 hrs of earth work | Assumption |
| **Total cost of establishment, maintenance, and replacement** | | **$660.69** | | **$925.52** | **TPVC** | |

**Table 5**
Itemized costs and Total Present Value Costs for N management for corn in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 1 year.

| Item | Cost timing (yr) | Minimum cost ($ ha$^{-1}$) | Mean cost ($ ha$^{-1}$) | Maximum cost ($ ha$^{-1}$) | Notes | Reference |
|---|---|---|---|---|---|---|
| Fertilizer application | $1-n$ | $14.83 | $24.09 | $42.01 | Anhydrous-injecting, w/tool bar | [81] |
| Diesel for equipment | – | | | | Included above | |
| Fertilizer cost | $1-n$ | | $156.40 | | North Central US mean 2008–2010 anhydrous ammonia price paid: $762.80 metric ton$^{-1}$; 168 kg N ha$^{-1}$; AA:82-0-0 (82%) | [56] |
| Total cost of establishment for baseline application | | $171.23 | | $198.41 | Using Fertilizer cost: $156.40 ha$^{-1}$ considering application of 168 kg N ha$^{-1}$ in Fall | [56] |
| **Total cost of establishment at a lower rate (from 168 kg N ha$^{-1}$ to 140 kg N ha$^{-1}$)** | | **$145.16** | | **$172.34** | Using Fertilizer cost: $130.33 ha$^{-1}$ for application of 140 kg N ha$^{-1}$ rather than $156.40 ha$^{-1}$ for 168 kg N ha$^{-1}$ | [56] |
| **Total cost of establishment of Spring application** | | **$178.42** | | **$205.60** | Spring price of $798 metric ton$^{-1}$ at 168 kg N ha$^{-1}$ application rate ($163.59 ha$^{-1}$) | [56,58] |
| Annual baseline revenue | $1-n$ | | $1850.12 | | Iowa mean 2008–2010 yield of 10.84 metric ton ha$^{-1}$ and 2008–2010 mean corn price received of $0.17 kg$^{-1}$; at 99% yield for 168 kg N ha$^{-1}$ | [55,56] |
| Annual revenue from changed yields due to N management (Lower rate) | $1-n$ | | $1831.44 | | Iowa mean 2008–2010 yield of 10.84 metric ton ha$^{-1}$ and 2008–2010 mean corn price received of $0.17 kg$^{-1}$; at 98% yield for 140 kg N ha$^{-1}$ | [55,56] |
| Annual revenue from changed yields due to N management (Spring application) | $1-n$ | | $1947.30 | | Iowa mean 2008–2010 yield of 10.84 metric ton ha$^{-1}$ and 2008–2010 mean corn price received of $0.17 kg$^{-1}$; with 4.2% yield boost for spring application | [56] |
| Total cost of establishment and revenue impacts for baseline application | | −$1614.32 | | −$1588.19 | TPVC (negative represents a revenue) | |
| **Total cost of establishment and revenue impacts at a lower application rate** | | **−$1621.42** | | **−$1595.28** | **TPVC (negative represents a revenue)** | |
| **Total cost of establishment and revenue impacts for Spring application** | | **−$1700.85** | | **−$1674.71** | **TPVC (negative represents a revenue)** | |
| **N Rate Marginal Cost** | | **−$7.09** | | **−$7.09** | **Marginal TPVC** | |
| **Spring N Marginal Cost** | | **−$86.52** | | **−$86.52** | **Marginal TPVC** | |

**Table 6**
Itemized costs and Total Present Value Costs for a cover crop in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 4 years.

| Item | Cost timing (yr) | Minimum cost ($ ha$^{-1}$) | Mean cost ($ ha$^{-1}$) | Maximum cost ($ ha$^{-1}$) | Notes | Reference |
|---|---|---|---|---|---|---|
| Seed costs | $1-n$ | $14.83 | | $29.65 | Planted at 63 kg ha$^{-1}$; cereal rye | [32,103] |
| Planting Drill | $1-n$ | $18.53 | $32.12 | $49.42 | Custom cost to have small grains drilled | [81] |
| Diesel for equipment | – | | | | Included above | |
| Spraying | $1-n$ | $11.12 | $15.07 | $21.99 | Ground, broadcast, tractor | [81] |
| Herbicide cost | $1-n$ | | $14.09 | | Herbicides, Glyphosate, 480 kg m$^{-3}$, Price paid, US Total, 2010: $6023 m$^{-3}$ ; 0.0023 m$^3$ ha$^{-1}$ | [32,56] |
| **Total cost of establishment** | | **$58.56** | | **$115.15** | **TPVC** | |
| Annual baseline revenue (no cover crop) | $1-n$ | | $1868.81 | | Iowa mean 2008–2010 yield of 10.84 metric ton ha$^{-1}$ and 2008–2010 mean corn price received of $0.17 kg$^{-1}$; at 100% yield | [56] |
| Annual revenue from changed yields due to cover crop | $1-n$ | | $1752.95 | | Iowa mean 2008–2010 yield of 10.84 metric ton ha$^{-1}$ and 2008–2010 mean corn price received of $0.17 kg$^{-1}$; at 6.2% yield reduction for corn following rye | [56] |
| Difference in annual revenue from baseline | | | $115.87 | | Considered a cost of cover crop with corn grown in every other year | |
| **Total cost of establishment and revenue impacts** | | **$594.98** | | **$800.39** | **TPVC** | |

**Table 7**
Itemized costs and Total Present Value Costs for a diversified crop rotation in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 10 years.

| Item | Cost timing (yr) | Minimum cost ($ ha$^{-1}$) | Mean cost ($ ha$^{-1}$) | Maximum cost ($ ha$^{-1}$) | Notes | Reference |
|---|---|---|---|---|---|---|
| Seed costs | Year 3 of every 5 | $101.19 | | $140.48 | Legume, alfalfa, public and common seed or proprietary seed, price paid, National, 2010: $273–$379 cwt$^{-1}$; planted 16.8 kg ha$^{-1}$ | [56] |
| Planting drill | Year 3 of every 5 | $18.53 | $32.12 | $49.42 | Custom cost to have small grains drilled | [81] |
| Diesel for equipment | – | | | | Included above | Assumption |
| Soil preparation | Year 3 of every 5 | | $34.10 | | Disking, harrow: Default values from ISU Ag Decision Maker | [72] (alfalfa) |
| Herbicide | Year 3 of every 5 | | $37.81 | | Default values from ISU Ag Decision Maker (machinery and chemical) | [72] (alfalfa) |
| Labor | 3–5 of every 5 | | $81.54 | | Pre-harvest labor: 7.4 h ha$^{-1}$ at $11.00 h$^{-1}$ | [72] (alfalfa) |
| Fertilizer | 3–5 of every 5 | $307.15 | | $481.36 | Default values from ISU Ag Decision Maker for establishment year (min) and production year (max); machinery and chemical | [72] (alfalfa) |
| Harvesting – mowing | 3–5 of every 5 | $19.77 | $30.64 | $37.07 | Mowing/conditioning | [81] |
| Harvesting – baling | 3–5 of every 5 | $74.13 | $123.55 | $172.97 | Haying baling - small square: $0.30–$0.70 bale$^{-1}$; 12.4 ton ha$^{-1}$ at 45.4 kg bale$^{-1}$ | [81]; Assumption |
| **Total cost of alfalfa establishment** | Year 3 of every 5 | **$674.23** | | **$860.55** | | |
| Total cost of alfalfa maintenance | Year 4 and 5 | $656.81 | | $772.95 | Labor, fertilizer and harvesting costs from above | |
| Corn in year 1 | YEAR 1 of 5 | | $1183.64 | | Cost of corn establishment (corn following soybean to be more accurate for years 6, etc.); land rent removed, 10.84 metric ton ha$^{-1}$ yield | [72] (corn following soybean) |
| Corn in year 2 | Year 2 of 5 | | $1312.13 | | Cost of corn establishment (corn following corn); land rent removed, 10.84 metric ton ha$^{-1}$ | [72] (corn following corn); [49] |
| **Total costs for five year diversified rotation** | | **$4214.00** | | **$4588.79** | **TPVC: Corn in years 1 and 2 with alfalfa establishment in year 3 and alfalfa maintenance in years 4–5** | |
| Alfalfa revenue | 4–5 of every 5 | | $1511.46 | | Alfalfa average yield 12.4 ton ha$^{-1}$ (assuming 3 cuttings); Iowa mean 2008–2010 alfalfa hay price received: $134.85 metric ton$^{-1}$ | [56,72] |
| Corn revenue | | | $1868.81 | | | [56] |

| | | | | | | |
|---|---|---|---|---|---|---|
| | 1–2 of every 5 | | | | Iowa mean 2008–20109 corn yield: 10.84 metric ton $ha^{-1}$ and 2008–2010 mean corn price received of \$0.17 $kg^{-1}$ | |
| **Total revenue for five year diversified rotation** | | | **\$6850.51** | | **TPV: Corn revenue in year 1 plus 4.5% yield boost, corn revenue in year 2, alfalfa revenue divided by 3 (only 1 cutting) in alfalfa establishment year, and alfalfa revenue in year 4–5** | [73] |
| **Total costs and revenue for diversified crop rotation for 10 yr horizon** | | **−\$10,456.91** | | **−\$8970.43** | **TPVC (negative represents a revenue)** | |
| Cost of corn and soybean five year rotation | | | \$4469.53 | | TPVC: Five year cost of corn soybean rotation; starting with corn (ISU Decision Maker, corn following soy, yield 10.8 metric ton $ha^{-1}$); soybean cost: \$637.53 $ha^{-1}$ ISU Ag Decision Maker for herbicide tolerant soybeans following corn, yield 3.33 metric $ton^{-1}$; land rent removed | [72] |
| Revenue of corn and soybean five year rotation | | | \$7564.77 | | TPV: Five year revenue of corn soybean rotation, starting with corn; corn revenue described above; soybean revenue: Iowa mean 2008–2010 yield of 3.33 metric ton $ha^{-1}$ and mean price \$0.38 $kg^{-1}$ yields \$1281.05 $ha^{-1}$ | [56] |
| **Total costs and revenue for corn and soybean rotation for 10 yr horizon** | | | **−\$12,276.31** | | **TPVC (negative represents a revenue)** | |
| **Marginal cost** | | **\$1819.40** | | **\$3305.87** | **Marginal TPVC** | |

**Table 8**
Review of nitrogen load reduction effectiveness for seven drainage water quality practices in the U.S. Midwest.

| Practices and references | N load reduction | | | Notes |
|---|---|---|---|---|
| | Minimum (%) | Mean (%) | Maximum (%) | |
| Controlled drainage | | | | |
| [86] | 30 | | 40 | Overview of this N management practice |
| [15] | 15 | | 75 | Controlled drainage factsheet |
| [87] | 48 | 75 | 100 | Load reduction for mean loads from six months of free drainage vs. controlled water tables at 0.25 m and 0.5 m above the drain; Ontario, Canada |
| [14] | | 30 | | Overview of this N management practice |
| [13] | 10 | | 20 | An original paper on drainage control |
| [88] | | 43 | | Controlled drainage/sub-irrigation system, Canada |
| [29] | 0 | | 50 | N technology comparison |
| [89] | 31 | 44 | 51 | Simulation of Midwestern region with Root Zone Water Quality Model-Decision Support System for Agrotechnology Transfer (RZWQM –DSSAT) |
| [90] | | 26 | | Mean of DRAINMOD-NII simulated N losses for drain spacing 18 m–36 m for conventional vs. controlled drainage; Waseca, Minnesota |
| Bioreactor | | | | |
| [76] | 11 | | 13 | Bioreactor in Iowa |
| [76] | 47 | | 57 | Bioreactor in Iowa |
| [76] | 27 | | 33 | Bioreactor in Iowa |
| [91] | 40 | 55 | 65 | Denitrification trenches surrounding tile drain, Iowa |
| [92] | 23 | 33 | 50 | Bioreactor in Illinois |
| [93] | | 47 | | Bioreactor in Illinois, slug of $NO_3^-$ injected |
| [94] | 18 | | 47 | Bioreactor in Minnesota |
| [94] | 35 | | 36 | Bioreactor in Minnesota |
| Wetland | | | | |
| [21] | 25 | | 78 | Review table |
| [18] | 33 | 40 | 55 | Annual N load reduction for three wetlands, three years of data; Champaign County, Illinois |
| [95] | | 33 | | Wetland in Illinois |
| [20] | 9 | | 15 | Mean N load reduction for two years from wetland with area treatment ratio of 1046:1; Iowa |
| [20] | 34 | | 44 | Mean N load reduction for two years from wetland with area treatment ratio of 349:1; Iowa |
| [20] | 55 | | 74 | Mean N load reduction for two years from wetland with area treatment ratio of 116:1; Iowa |
| [29] | 20 | | 40 | N technology comparison |
| [54] | 40 | | 90 | Summary of CREP wetlands in Iowa |
| Spring N application | | | | |
| [96] | −67 | 6.4 | 44 | Load difference between fall and spring (corn phase) |
| [97] | 0 | 27 | 41 | Load difference between fall and spring (corn phase) |
| [23] | 24 | | 30 | 6-yr period at Waseca, Minnesota |

| | | | | |
|---|---|---|---|---|
| [29] | −10 | | 30 | N technology comparison |
| [59] | 14 | 35 | 52 | Simulation with Environmental Policy Integrated Climate (EPIC) for central Illinois; Fall vs. spring application at five rates ranging from 112 kg N $ha^{-1}$ to 224 kg N $ha^{-1}$ for Drummer soil |
| [49] | −62 | −23 | 7.4 | N load difference between spring and fall applied at 168 and 252 kg N $ha^{-1}$; Iowa |
| N rate reduction | | | | |
| [23] | 21 | | 28 | 6-yr period at Waseca, Minnesota; 134 kg $ha^{-1}$ vs. 202 kg $ha^{-1}$ application |
| [29] | 20 | | 70 | N technology comparison |
| [98] | 17 | | 40 | Central Iowa; loadings of 48 kg N $ha^{-1}$, 35 kg N $ha^{-1}$, and 29 kg N $ha^{-1}$ for high, medium and low N application rates, respectively |
| Cover crops | | | | |
| [66] | | 13 | | Southwestern Minnesota, three year study |
| [100] | | 40 | | Based on review |
| [71] | −13.5 | −3.3 | 7.6 | Four year loads and mean for corn treatment vs. corn with rye cover; Gilmore City, Iowa |
| [31] | | 61 | | Four year average; Boone County, Iowa |
| [29] | 10 | | 70 | N technology comparison |
| Crop rotation | | | | |
| [36] | 14 | | 77 | Review |
| [99] | 11 | | 14 | Six year average losses from corn/soybean or soybean/corn vs. rotation with three years alfalfa followed by corn, soybean, oats; Nashua, Iowa |
| [35] | 18 | 48 | 80 | Conversion from alfalfa pasture; three year study, compared with corn and soybean and continuous corn rotations; Lamberton, Minnesota |
| [29] | −50 | | 95 | N technology comparison |

**Table 9**
Nitrogen load reduction effectiveness and Equal Annual Costs in terms of treatment area or nitrogen removal for seven drainage water quality practices in the U.S. Midwest (without government payments).

| | EAC (area-based) | | Load reduction from Fig. 2 | | | | EAC (N-based) | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Minimum ($ ha$^{-1}$ yr$^{-1}$) | Maximum ($ ha$^{-1}$ yr$^{-1}$) | 25th (%) | 75th (%) | Mean (%) | Median (%) | Mean (Standard Deviation, $ kg N removed$^{-1}$ yr$^{-1}$) | Median ($ kg N removed$^{-1}$ yr$^{-1}$) | Minimum [a] ($ kg N removed$^{-1}$ yr$^{-1}$) | Maximum [a] ($ kg N removed$^{-1}$ yr$^{-1}$) |
| Controlled Drainage | $9.30 | $37.00 | 26.0 | 50.0 | 40.5 | 40.0 | $2.00 ($1.40) | $1.70 | $0.60 | $4.50 |
| Bioreactors | $16.00 | $32.00 | 27.0 | 47.0 | 37.5 | 36.0 | $2.10 ($0.90) | $2.00 | $1.10 | $3.80 |
| Wetland | $31.00 | $43.00 | 30.9 | 55.0 | 42.8 | 40.0 | $2.90 ($0.80) | $2.80 | $1.80 | $4.40 |
| N rate reduction | −$7.40 | −$7.40 | — | — | 14.5 | — | −$1.60 ($0.00) | −$1.60 | −$1.60 | −$1.60 |
| Spring N application [b] | −$90.00 | −$90.00 | −2.5 | 31.3 | 9.3 | 19.0 | −$14.00 ($12.00) | −$12.00 | −$31.00 | −$0.07 |
| Cover crop | $164.00 | $221.00 | 4.9 | 45.3 | 23.1 | 11.5 | $55.00 ($48.00) | $38.00 | $12.00 | $144.00 |
| Crop rotation | $224.00 | $408.00 | 14.0 | 77.0 | 34.1 | 18.0 | $43.00 ($29.00) | $39.00 | $9.30 | $93.00 |

[a] Minimum and maximum calculated using the minimum EAC and the 75th percentile load reduction and the maximum EAC and the 25th percentile load reduction, respectively.
[b] Due to confounding effects of negative EAC and negative 25th percentile load reduction (indicating a contribution to the N load), the maximum value for Spring N application was calculated using the marginal increase to the baseline load based on the 25th percentile and the minimum value was calculated from the mean load reduction.

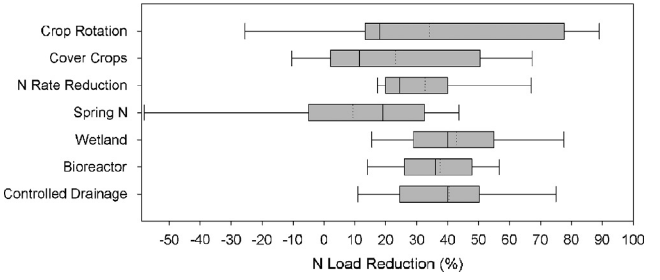


**Fig. 1.** Comparison of nitrogen load reductions obtained from literature for seven water quality improvement strategies in the U.S. Midwest; the box boundaries represent the 25th and 75th percentiles, the solid line represents the median, the dotted line represents the mean, and the whiskers show the 10th and 90th percentiles.

are shown in Table 9).

$$\frac{\text{EAC \$}}{\text{kg N yr}} = \frac{\text{minimum or maximum EAC \$}}{ha\ yr} \div \left(\frac{\text{31.4 kg N lost baseline}}{\text{ha}} \times \text{Load removal percentage mean, median, 25th, or 75th}\right) \tag{5}$$

In the case of modified N application rate, rather than use load reduction values from literature, a correlation from Lawlor et al. [27] was used (Eq. (1)). For this practice, literature values proved to be too variable as they were not for the specific rates used in this comparison. After drainage $NO_3^-$–N concentrations were developed via Eq. (1) for the two application rates, a constant drainage volume was assumed to develop a percent N load reduction. While, Eq. (1) was specifically applicable to the database and site from which it was developed (northwestern Iowa), and does not account for other factors that affect N leaching losses (e.g., soil mineralizable N, the time of N application relative to crop N uptake, soil moisture content, weather conditions), it provided a straight forward approach to estimate approximate concentrations based upon N fertilizer application rates.

Finally, because cost-share has been shown to be an important incentive for operators to make environmental mitigation decisions, the impact of existing government cost-share and incentive programming was assessed. In Iowa, USDA environmental quality incentive program (EQIP) payments were available for each of the practices evaluated here except for modification of fertilizer rate [50] (Table 10). EQIP cost rates used were standard rate, not the higher rates available for historically underserved groups. Incentives for controlled drainage, bioreactors, wetlands, and N management were treated as one time, present value payments (year 1), while the others occurred in years $1-n$ with time limits set by EQIP payment schedules. Though EQIP funding is available for wetlands, cost share payments from the Iowa Department of Agriculture and Land Stewardship's Conservation Reserve Enhancement Program (IDALS CREP) are more appropriate because the wetland in this analysis was sized based upon Iowa CREP guidelines. For a CREP 30 year easement agreement, compensation included 15 annual rental payments of 150% the soil rental rate, cost-share for 100% of the wetland installation (90% federal, 10% state), and a one-time incentive payment ($247 ha$^{-1}$) [19,51]. The soil rental rate was assumed to be the average cash rental rate for 2008–2010 for the state of Iowa ($447 ha$^{-1}$) [52].

### *2.1. Controlled drainage*

The major cost of controlled drainage is the capital expense of the structures and their installation. Because of this expense, land slope limitations are an important factor as more structures are needed at steeper sites. Another important consideration is the cost difference between implementing controlled drainage in existing vs. newly designed drainage systems [14].

For this evaluation of controlled drainage, the costs to retrofit an existing drainage system and the cost to implement a new drainage system designed for controlled drainage are considered. To reflect

**Table 10**
Environmental Quality Incentives Program (EQIP) payment schedule rates for Iowa for seven nitrogen reduction practices [50] and calculated total present value ($TPVC_{Govt}$) of this government cost-share for this evaluation.

| | EQIP practice name | Practice code | Payment schedule cost | Payment unit | Minimum life (yr) | Year of payment | Payment (\$ ha treated $^{-1}$) | $TPVC_{Govt}$ (\$ ha$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| Controlled drainage [a] | Drainage water management | 554 | \$364.08 | Per number of water control zones | 1 | 1 | \$44.98 | \$44.98 |
| Bioreactors [b] | Denitrifying bioreactor | 747 | \$3999.50 | Per bioreactor | 10 | 1 | \$197.66 | \$197.66 |
| Wetland [c] | Wetland creation | 658 | \$680.00 | Per acre | 15 | 1 | \$16.80 | \$16.80 |
| N rate reduction [d] | — | — | — | — | — | — | \$0.00 | \$0.00 |
| Spring N application [d, f] | Nutrient management | 590 | \$11.00 | Per acre | 1 | 1 | \$27.18 | \$27.18 |
| Cover crop [e, f] | Cover crop (and green manure) | 340 | \$53.26 | Per acre | 1 | 1–3 | \$131.61 | \$379.83 |
| Crop rotation [f] | Conservation crop Rotation | 328 | \$52.00 | Per acre | 1 | 1–3 | \$128.50 | \$370.85 |

[a] Used scenario of 65 ha, requiring eight zones.
[b] EQIP specifies treatment of drainage from 12.1 ha which was less than the treatment area assumed here of 20.2 ha; EQIP cost-share was not used in replacement years for bioreactors or controlled drainage.
[c] Based on CREP 30 yr contract incentives rather than EQIP cost share shown here (see Section 2).
[d] Based on a mid-range payment rate requiring only two additional enhancement practices.
[e] Based on “cover crop winter hardy” rate for a winter cover of rye.
[f] EQIP funding for N management, cover crop and crop rotation practices has three year payment time limits; payments for cover crop and crop rotation were assumed to happen in the first three years of the analysis period and because N management had a planning horizon of $n=1$, only 1 year of EQIP was included.

the marginal cost of water quality improvement and not just the cost of new drainage systems, contractor tiling and materials expenses for new systems are not included. Full cost components are described in Table 2. Regarding more long-term costs, the cost of maintenance for this practice includes landowner time to manipulate the control structures; this would vary based on the number of structures, distance between them, and management intensity a landowner chose. The control structure stop logs/gates need to be replaced every eight years. Because the structures themselves would need to be replaced in year 40, this determined the practicable lifespan of this practice ($n=40$).

### 2.2. Bioreactors

As with controlled drainage, bioreactor establishment costs include design, contractor and structure fees. However, unlike controlled drainage, bioreactor treatment area differs from the surface area of the technology. Here, the \$ ha$^{-1}$ values referred to the treatment area not the bioreactor surface footprint. On an itemized basis, a maximum value for engineering fees of \$40 h$^{-1}$ for 16 h of work is assumed, though if the bioreactor is designed by a technical service provider, these fees may not apply. Although no land is typically removed from production for bioreactors, seeding the surface is important to prevent erosion of the soil cap. Bioreactors are typically less than 0.5% of the drainage treatment area, so this area ratio is used for the seeding and mowing costs. Bioreactor full cost components are described in Table 3.

Farmer time for adjusting the control structures is minimal compared to the controlled drainage practice due to fewer structures here. In addition to annual maintenance, the bioreactor material is replaced once in year 20 (involving costs associated with new woodchips, seeding and contractor fees) before the structures' lifespan is exhausted in year 40 (bioreactor practicable lifetime, $n=40$). Similar to controlled drainage, the stop logs/gates are replaced every eight years.

### 2.3. Wetlands

Wetlands are unique in that their capital expense can be very high, but they are capable of treating drainage from far larger areas than the other strategies considered here. Design and construction are important components of wetland establishment but the largest single expense is the land acquisition cost. Longer-term economic considerations sometimes include the opportunity cost of lost crop income (e.g., Prato et al. [53] and Crumpton et al. [22]), as well as maintenance and mowing expense and potential income streams.

For the purposes of this comparison, a 405 ha treatment area is assumed with a wetland occupying 1% of this area (4 ha) consistent with the conservation reserve enhancement program (CREP) guidelines for Iowa which specify a wetland size of 0.5–2% of the treatment area (not including associated wetland buffer) [19,54]. Accordingly, in addition to the wetland basin, a grass buffer is required. The wetland buffer has a 3.5:1 area ratio with the wetland (i.e., 3.5% of the treatment area in buffer, 14 ha) (Iowa Department of Ag. and Land Stewardship, personal communication, 2011). Because land acquisition costs are the largest portion of CREP wetland expense, this is included here; however, land for the other practices (e.g., edge-of-field area for the bioreactor or fields for the in-field practices) is assumed to be owned. Alternatively, forgone annual land rent would be another way to account for land costs. The cost per area for this practice reflected the area treated, not the area of the wetland and associated buffer. Wetland cost components are shown in Table 4.

Structural components include a water control structure and a weir plate, which are used to control wetland flow. The annual maintenance cost involves mowing 10% of the buffer area. Replacement costs of the control structure and sheet pile weir in year 40 are included within the 50 year wetland planning horizon ($n=50$). Also, over the life of a wetland, sediment removal and earthwork maintenance would be required, though those costs are not incorporated in this analysis because their timing would be difficult to estimate and may occur at greater than the 50 year planning horizon.

### 2.4. N rate reduction (168–140 kg N $ha^{-1}$)

The establishment costs for both N management practices (rate reduction and timing) are similar and include custom rates for application machinery usage and fertilizer costs as described in Table 5. Because an N management practice is an annual occurrence, there are no long-term maintenance costs but, rather, establishment cost and revenue impacts occur every year (practicable lifespan, $n=1$). For these N management strategies, a baseline scenario of fall applied 168 kg N $ha^{-1}$ is developed for comparison. The marginal difference in TPVC between the baseline and the rate/timing alternative is used in the analysis rather than the absolute value of the rate/timing TPVC themselves. Using these marginal costs of the lower rate practice and of the spring timing practice allows evaluation of their cost solely due to water quality improvement.

Financial analysis of lowering the N application rate consists of less fertilizer expense in addition to the cost of potential yield loss depending upon the initial and final application rates [25]. This analysis is complicated by the variability of the impacts of initial and revised fertilizer rates. In practice, challenges to N fertilizer rate reduction include the fact that the optimum rate is indeterminable at application time (though soil testing can help) and is highly variable year to year. Sawyer and Randall [25] provide a detailed explanation of these variable negative and positive returns based on initial and final fertilizer rates.

In analyzing the costs of reduced fertilizer rate here, “establishment” cost consists of less fertilizer purchased (i.e., a cost savings) as well as the effect of potentially reduced yield. The Iowa State University N-Rate Calculator [55] is used to estimate the yield impact from changing the fertilizer rate. Using a three-yr average (2008–2010) anhydrous ammonia price of \$763 metric $ton^{-1}$ [56] and a three-yr average (2008–2010) Iowa corn price of \$0.17 $kg^{-1}$ [56], the calculated percent of maximum yield is 99% at an N application rate of 168 kg N $ha^{-1}$ and is approximately 98% at 140 kg N $ha^{-1}$ (corn following soybean rotation). However, it is worth noting that shifting to this lower rate permanently may not be sustainable over long periods if soil N pools become depleted [57].

### 2.5. Spring N application

The cost of shifting application from the fall to the spring is affected by differences in both fall/spring fertilizer price and yield. Because current fall vs. spring fertilizer prices are no longer published by USDA, the average historical difference in the fall and spring fertilizer prices, on a percentage basis, is used to calculate the average increase in expense for spring anhydrous application. Between 1960 and 1994, the average prices for September/October were \$184 metric $ton^{-1}$ and for April/May were \$193 metric $ton^{-1}$ [58], thus an increase of 4.6% over the average 2008–2010 anhydrous price of \$763 metric $ton^{-1}$ is used for spring (spring: \$798 metric $ton^{-1}$).

Multiple authors have reported lower drainage $NO_3^-$ loadings with corresponding higher corn yields for spring vs. fall N applications [23,59,60]. Spring N fertilizer applications may increase yield by 8–14% compared to fall applications [23,60], though this may not always be the case. For example, there was no corn yield difference between fall and spring applications at two different application rates during a study in Iowa [49]. Despite this variability, an overall 4.2% corn yield boost is included for the practice of spring application (site year average from Refs. [49,61–64]).

### 2.6. Cover crops (cereal rye)

For the purposes of this evaluation, cereal rye (*Secale cereale* L.) is studied as a cover crop because this crop has good potential to improve water quality in cool Midwestern climates [31] and is popular in this region [65]. First year costs of a cover crop (Table 6) (assuming a no-till system in this analysis) include planting as well as herbicide application because cereal rye overwinters [32]. Cover cropping is an annual practice, thus there are no long-term maintenance costs but rather annual establishment costs. A yield reduction for corn following rye is also an important part of the analysis. A 6.2% corn yield reduction is assumed compared to a baseline where no cover crop was used (site year average from Refs. [31,66–71]). This corn revenue reduction is assumed to occur every other year during the planning horizon (i.e., a corn/soybean rotation; cover crop practice period, $t=2$; cover crop planning horizon, $n=4$).

### 2.7. Crop rotation (multiple years of alfalfa)

The number of possible rotation combinations is quite large, and to simplify this work, a multi-year incorporation of alfalfa (*Medicago sativa* L.) into a corn rotation is investigated. Only one year of alfalfa in a rotation may not be as beneficial as several years considering high seed cost and potential low alfalfa yield in the establishment year [36]. Therefore, this diversified crop rotation consists of two years of corn (years 1–2) followed by three years of alfalfa (years 3–5). The major costs for such a crop rotation are the seed, planting, and harvesting. The cost components of this rotation are shown in Table 7, with the rotation practice period ($t$) equal to five years and the planning horizon, $n$, equal to 10 years.

Within the rotation, enterprise budget information published by Iowa State University is used to specifically estimate the costs of corn following soybean (for years 1, 6, etc.; most applicable for corn following alfalfa) and for corn following corn (in years 2, 7, etc.) [72]. Default Iowa State University Ag Decision Maker [72] values were used after removing land rent costs (i.e., assumed land owned) and substitution of average Iowa 2010 corn yield from USDA NASS [56].

A multiple year alfalfa rotation may provide monetary benefit via reduced fertilizer requirements, reduced tillage and other field trips, and revenue from the alfalfa harvest. Here only direct revenue streams are considered with alfalfa revenue in years 3–5 and corn revenue in years 1–2. The establishment year of alfalfa is assumed to only have one cutting rather than the three as in the maintenance years (i.e., establishment years had one third of the yield experienced in maintenance years). Corn following alfalfa may have an increased yield of 19–84% compared to corn after corn according to a review by Olmstead and Brummer [36], but Liebman et al. [73] showed more moderate corn yield increases averaging 4.5% which was used here for the first year of corn.

Additionally, the TPVCs for this crop rotation scenario are compared against TPVCs for traditional corn/soybean rotations. Similarly to the N management practices, this allowed evaluation of the cost of this water quality practice (i.e., marginal cost of the practice). The corn/soybean baseline scenario is evaluated using the same five year framework as the extended rotation with cost values taken from ISU Ag Decision maker for corn following soybeans and herbicide tolerant soybeans following corn with default values except for removal of land rent costs and use of average yields (2008–2010, USDA NASS data) (Table 7) [72].

## 3. Results and discussion

### 3.1. Equal annualized costs

The TPVCs from the seven practices ranged from a cost savings of approximately \$90 $ha^{-1}$ for spring applied N fertilizer to a cost of \$3306 $ha^{-1}$ for a diversified crop rotation (Tables 2–7), and the resulting EACs ranged from −\$90 $ha^{-1}$ $yr^{-1}$ (Spring N, representing cost savings) to \$408 $ha^{-1}$ $yr^{-1}$ (crop rotation) (Table 9). The highest EACs were associated with the two in-field vegetated practices, cover crops and crop rotations, and the lowest were associated with the N management strategies. However, the high EACs developed for the cover and diversified cropping practices were associated with large uncertainties (Tables 1 and 9).

With regard to spring N applications, Randall and Sawyer [24] also noted long-term economic gains of \$46–\$126 $ha^{-1}$ $yr^{-1}$ (seven and fifteen year averages). However, a complete shift from fall fertilization could be expensive for individual producers in terms of both additional infrastructure required for spring applications (storage, equipment, labor, handling, application, etc.) and in the potential loss of yield by a delayed planting date [74]. Additionally, when lower N rates are applied, the risk of a yield loss is increased compared to higher application rates if it is a year where corn is more responsive to N inputs (depending upon the soil mineralizable N). In these years, the probability of obtaining a certain yield percentage declines when lower rates are applied; this probabilistic variability was not reflected here. Any such potential increased risk for either of these N management practices is an important factor in terms of producer decision-making.

Along with the relatively high EACs for the rye cover crop (\$164–\$221 $ha^{-1}$ $yr^{-1}$; Table 9), several comments should be noted. First, costs to kill the cover are contingent upon producer actions. For example, in a no-till system as assumed here, an early burn-down application of herbicide may be done regardless if a cover crop was present; likewise, in a tilled system, a producer may do a second tillage pass in the spring regardless of a cover crop. Second, rye cover crop implementation costs can be \$10–\$15 $ha^{-1}$ lower if a landowner chooses not to use a custom operator [75]. Next, potential negative yield impacts will likely be reduced or minimized through several years of experience with cover crop management. This increased experience also likely means a more effective cover, though returns to farm management can improve under highly skilled managers regardless of the production practice. Finally, some of the N taken up by a cover crop will be returned to future crops. It is difficult to place an economic value on this, but it is worth noting the multiple benefits to the soil provided by cover crops [33]. Because cover crops are typically done for reasons other than drainage water quality improvement, it has been suggested that only a portion of the cost should be attributed to N. However, because this work was solely focused on N reduction cost effectiveness, see Table 1 or Christianson et al. [34] for discussion of the ecosystem services provided by these practices.

The EAC for the diversified rotation was \$224–\$408 $ha^{-1}$ $yr^{-1}$ (Table 9). The values developed here were contrary to values from Olmstead and Brummer [36] who showed a diversified rotation was more profitable than a conventional rotation. One major caveat worth noting is the potential for large scale market effects if this rotation were done by a large numbers of producers in a limited area; if this practice became widespread, the alfalfa price could markedly decline.

The two field-scale constructed practices, controlled drainage and bioreactors, had similar EAC ranges at \$9.30–\$37 $ha^{-1}$ $yr^{-1}$ (spanning both existing and new drainage systems) and \$16–\$32 $ha^{-1}$ $yr^{-1}$, respectively. For reference of installation costs, bioreactor TPVC estimates (Table 3) were within the range of five bioreactor installations in Iowa (total costs of \$4400–\$11,800 to treat drainage from 12 ha to over 40 ha [76]), and overall TPVCs estimated for constructed wetlands (\$661–\$926 $ha^{-1}$, Table 4) compared well with cost assessments from IDALS CREP wetlands constructed in Iowa. CREP wetlands average approximately \$880 $ha^{-1}$ including land acquisition (\$513 $ha^{-1}$), establishment and maintenance costs (\$297 $ha^{-1}$), and engineering costs (\$69 $ha^{-1}$). As of 2011, 72 wetlands had been installed under the CREP wetland program in Iowa with an average treatment area of 505 ha (Iowa Department of Ag. and Land Stewardship, personal communication, 2011).

### 3.2. Comparative average cost effectiveness of nitrogen mitigation

In addition to variation between practices in TPVCs and EACs, the practices also varied widely in terms of N removal effectiveness (Fig. 1). For example, modification of fertilizer timing had comparatively low N removal, ranging notably into the potential for negative water quality impacts, while the constructed practices tended to have relatively better water quality performance. Bioreactors had the smallest range of N load reduction between the 25th and 75th percentiles with mean and median values above 35% load reduction. The other two constructed practices, controlled drainage and wetlands, had similarly high load reduction potential (means and medians $\geq$40%). Note, because the 25th percentile for spring N application was a negative value ($-2.5\%$), indicating a contribution to the N load, the resulting marginal increase to the baseline load was used to calculate the \$ kg $N^{-1}$ $yr^{-1}$ for this value.

When these N removal performances were combined with the cost data, spring N application timing was the most cost effective option for removing N from drainage (mean \$14 kg $N^{-1}$ $yr^{-1}$ cost savings or revenue) and cover crop the least (mean \$55 kg $N^{-1}$ $yr^{-1}$) (Table 9, Fig. 2). Both N management practices yielded negative average cost efficiencies indicating a savings or increased profitability. However, it's important to note nutrient management practices alone may not be sufficient to meet all N water quality goals in the Midwestern Region. In addition to the highest mean values, the cover crop and the diversified rotation had the largest standard deviations (pre-government payment), which highlighted the variability of these two in-field vegetative practices both in terms of costs and N removal potential. The more constructed practices of controlled drainage, bioreactors and wetlands had fairly comparable average cost efficiencies with mean values between \$2 and \$3 kg $N^{-1}$ $yr^{-1}$ (Table 9, Fig. 2).

To put these average cost efficiencies in context of other reported values is difficult in light of the variable methodology and limited transparency of other assessments. Nevertheless several practices were in the range of literature, while others were distinctly different. For example, the cost efficiency of controlled drainage in this analysis was \$2.00 ± \$1.40 kg $N^{-1}$ $yr^{-1}$, which was similar to reports which are often in the range of \$2–\$4 kg $N^{-1}$ [77,78]. Moreover, the average cost efficiency of wetlands is often reported at approximately \$3–\$4 kg $N^{-1}$ [51,54,77,79]; the value reported in our study was \$2.90 ± \$0.80 kg $N^{-1}$ $yr^{-1}$. Only one report was available for bioreactors; in a multi-year cost analysis of a theoretical denitrification system, Schipper et al. [80] calculated costs of \$2.39–\$15.17 kg $N^{-1}$. This range was higher than what was estimated for a bioreactor in our study (\$2.10 ± \$0.90 kg $N^{-1}$ $yr^{-1}$). Finally, cover crops have been reported to be less expensive per kg N removed than calculated in this analysis (mean \$55 ± \$48 kg $N^{-1}$ $yr^{-1}$). Values from cover crop literature have ranged from \$1.26 kg $N^{-1}$ to \$11.06 kg $N^{-1}$ [32,75,77], though these previous reports may not have included corn yield impacts.

Inclusion of EQIP or CREP payments generally increased the average cost effectiveness of the practices from a farmer's perspective (Table 9 vs. Table 11) with the largest percentage change

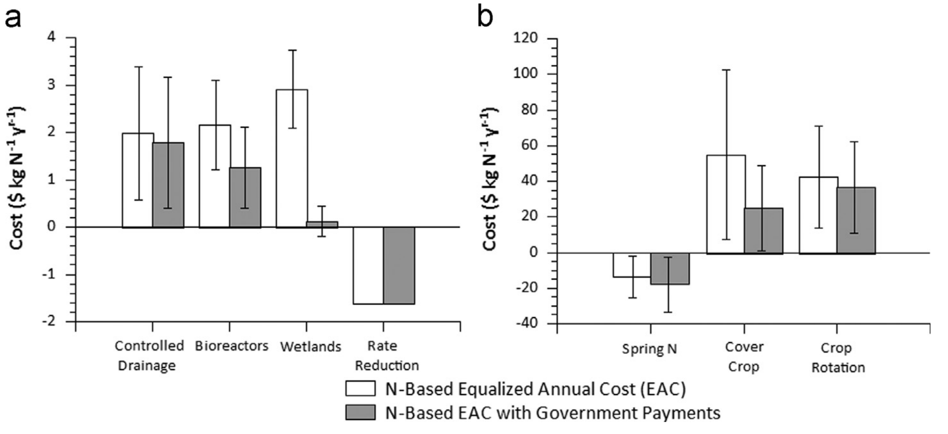


**Fig. 2.** Equal Annual Costs (\$ kg $N^{-1}$ $yr^{-1}$) on a nitrogen removal basis for seven agricultural practices in the U.S. Midwest with and without government payments at real discount rate of 4% and analysis horizons of practicable lifespans by practice; note y-axis scales differ for figure parts (a) and (b).

**Table 11**
Nitrogen removal-based Equal Annual Costs for seven drainage water quality practices in the U.S. Midwest including government payments and additional revenue at real discount rate of 4 % and analysis horizons of practicable lifespan by practice.

| | Equal annual costs | |
|---|---|---|
| | Mean (standard deviation, \$ kg $N^{-1}$ $yr^{-1}$) | Median (\$ kg $N^{-1}$ $yr^{-1}$) |
| Controlled drainage | \$1.80 (\$1.40) | \$1.50 |
| Bioreactors | \$1.30 (\$0.86) | \$1.10 |
| Wetland | \$0.12 (\$0.32) | \$0.09 |
| N rate reduction | −\$1.60 (\$0.00) | −\$1.60 |
| Spring N application | −\$18.00 (\$16.00) | −\$16.00 |
| Cover crop | \$25.00 (\$24.00) | \$16.00 |
| Crop rotation | \$36.00 (\$26.00) | \$33.00 |

occurring for the wetland practice. Without government payments, the practices in order of average cost effectiveness were (based on mean value): Spring N application, N application rate reduction, controlled drainage, bioreactors, wetlands, crop rotation and cover crops. When government payments were included, wetlands and bioreactors became the third and fourth most cost effective practices, respectively, and diversified crop rotations became the least cost effective (from the farmer's perspective) (Fig. 2).

## 4. Conclusions

Each drainage N reduction strategy provides landowners an additional distinct option for drainage water quality improvement and different strategies or combinations of such will be applicable in different locations. In this work, the N management practices were the most cost effective as both lowering the application rate (from 168 to 140 kg N $ha^{-1}$) and moving applications to spring resulted in negative costs. Of course, the scenarios here were limited in scope, and there is a wide range of N management and application possibilities that could yield different results. Importantly, a complete ban of fall fertilization could have large-scale economic effects, which were not investigated in this farm-level analysis. The least cost effective practices were the in-field vegetative practices of cover crop and crop rotation though these average cost efficiencies had wide standard deviations. Moreover, benefits like soil productivity, erosion protection, and management or reduction of multiple contaminants were not quantified. The three constructed practices were comparable in terms of pre-cost share \$ kg $N^{-1}$ $yr^{-1}$ although wetlands were very cost effective when CREP incentives were included. A final important note is that while this study focused on water quality $NO_3^-$ mitigation, several of these practices provide significant additional ecosystem services not quantified here.

In an applied sense, these average cost efficencies need to be considered in context of the multiple agricultural and environmental objectives that will differ for each farm and for each farmer. Though the N management practices had the most attractive cost efficiencies, sole focus on N management either on farm or in policies will likely be insufficient to meet water quality goals in entirety. And while improved N management may be “low hanging fruit” for farmers aiming to improve water quality, there are important large scale impacts (e.g., infrastructure requirements for a complete fall fertilizer ban) that were not investgated in this farm level study. At the other end of the cost efficiency spectrum, the in-field vegetative practices were the least attractive in this analysis. However, with this work defined narrowly by reduction of N in drainage water, several potential additional agronomic and environmental benefits of these practices were excluded. Reduction of erosion and improved soil qualities potentially provided by these practices may be important considerations for farm decision makers. These strategies should certainly not be overlooked as Dinnes [29] reported that diversifying cropping systems in Iowa has the most potential to reduce $NO_3^-$ loadings compared with any other best management practice.

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## Appendix A. Supplementary information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.wre.2013.09.001.

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