JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION

Vol. 50, No. 4 AMERICAN WATER RESOURCES ASSOCIATION August 2014

CONSTRUCTED WETLAND TREATMENT OF NITRATES: REMOVAL EFFECTIVENESS AND COST EFFICIENCY[1]

Alan R. Collins and Neil Gillies[2]

ABSTRACT: A constructed wetland (CW) was strategically placed to treat nitrates in groundwater as part of a watershed-based farmer engagement process. Using stream water quality data collected before and after installation, this CW was found to reduce stream concentrations of nitrogen from nitrate (NO_3-N) during the growing season by about 0.14 mg/l at mean streamflow, a 17% reduction. Based upon realistic ecological and economic assumptions, about 80 kg of NO_3-N were removed annually by the CW at a cost of around US$30/kg. This per unit cost is at the low range of small wastewater treatment plant costs for nitrates, but higher than the costs of reduced fertilizer application.

(KEY TERMS: best management practices [BMPs]; water quality economics; groundwater remediation; econometrics; watersheds.)

Collins, Alan R. and Neil Gillies, 2013. Constructed Wetland Treatment of Nitrates: Removal Effectiveness and Cost Efficiency. *Journal of the American Water Resources Association* (JAWRA) 50(4): 898-908. DOI: 10.1111/jawr.12145

INTRODUCTION

Excess nutrients remain an important water quality problem in many of this nation's waterways. Nitrogen (N) in the form of nitrates (NO_3) is one nutrient that creates conditions for enhanced eutrophication. Water bodies like the Chesapeake Bay are particularly susceptible to the problems created by eutrophication due to its large land to water area ratio. Thus, a bay-wide total maximum daily load requirement was imposed across six states and the District of Columbia in December 2010 to limit nutrient loads (USEPA, 2010).

Agricultural nonpoint pollution is a primary contributing factor to excess nutrients in many watersheds. For example, Moore *et al.* (2011) estimate that the agriculture sector contributes 48% of nitrogen to the Potomac River in West Virginia. Groundwater transport is estimated to be the pathway for about one-half of stream level nitrogen in the Chesapeake Bay watershed (Phillips and Lindsey, 2003). Solving agricultural nonpoint pollution traditionally has involved nutrient and sediment loss prevention through implementation of best management practices (BMPs). However, offering cost-share incentives to encourage pollution prevention has proved to be a relatively costly method to improve water quality due to a lack of targeting cost-share eligible BMPs within watersheds (Ribaudo *et al.*, 1999).

The objective of this research is to examine the removal effectiveness and cost efficiency of using a

[1]Paper No. JAWRA-12-0229-P of the *Journal of the American Water Resources Association* (JAWRA). Received October 25, 2012; accepted September 16, 2013. **Discussions are open until six months from print publication**.

[2]Respectively, Professor (Collins), Agricultural and Resource Economics Program, P.O. Box 6108, West Virginia University, Morgantown, West Virginia 26506; and Former Executive Director (Gillies), Cacapon Institute, High View, West Virginia 26808 (E-Mail/Collins: alan.collins@mail.wvu.edu).

constructed wetland (CW) treatment system to reduce nitrogen from nitrates (NO_3-N) discharges into a surface stream. CW systems have a number of pathways to remove nutrients and have been used to treat wastewater and groundwater nutrients in numerous applications (Kadlec and Knight, 1996; Vymazal, 2007). The CW described in this article was a strategically targeted BMP whose placement was the end result of a farmer-involved process to identify sources of nitrogen pollution within a watershed. This research will assess the ability of the CW to statistically reduce stream concentrations of NO_3-N and estimate a cost per kilogram (kg) for NO_3-N removed based on projected base flow of the stream and operational life of the CW. Per unit costs will be compared with those in the research literature on nitrogen prevention and interception approaches to assess the cost efficiency of this CW.

STUDY AREA DESCRIPTION AND FARMER ENGAGEMENT PROCESS

The CW examined in this research was placed on the Cullers Run watershed. Cullers Run is a tributary of the Lost River located in the eastern panhandle region of West Virginia (Figure 1). Cullers Run watershed occupies 2,978 hectares (7,360 acres) in Hardy County, West Virginia's largest poultry production county. Sixteen percent of the watershed is devoted to agriculture, mostly pasture or hay land. Row crops comprise 3.63% of the agricultural land and are located primarily in the floodplain (Cacapon Institute, 2002). The rest of the watershed is forest. There are 12 poultry houses raising approximately 500 animal units of primarily broilers in the watershed. Most agricultural fertilizer use in the watershed is provided by poultry litter.

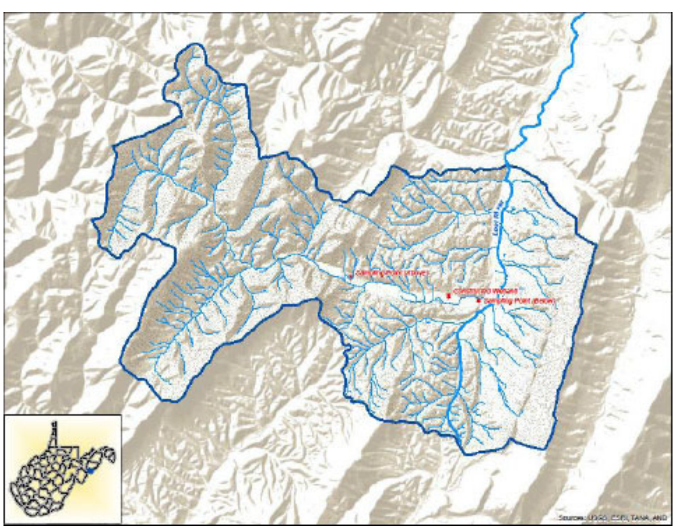

FIGURE 1. Location of Cullers Run Watershed, Sampling Points, and Constructed Wetland.

The process used to engage farmers on this watershed was a field experiment conducted from 2007 to 2009. Performance-based payments were provided to a group of farmers in the watershed based on NO_3-N concentrations in Cullers Run. Monthly payments were made to farmers as a group and were based on a formula which included the quantity and the quality of water flowing from the stream. Water quality was measured as a ratio of NO_3-N concentrations in Cullers Run relative to a control watershed.

As a result of these payments, the farmer group had two main responses: (1) they requested additional water quality information throughout the watershed, and (2) implemented cost sharing of BMPs. For the first response, information consisted of three watershed-wide samplings conducted in 2007 and 2008 to estimate nitrate load contributions per subwatershed. This was followed by a series of focused source tracking samplings in the lower section of Cullers Run where subwatershed contributions were high to detect specific areas within this subwatershed contributing elevated levels of NO_3-N. The second response included funding of cover crops plantings for two years on one corn field and fencing livestock away from the stream on one pasture. One farmer, on his own, installed a covered winter feedlot within a small subwatershed which showed a higher than average nitrate loading.

In August 2008, 12 water quality sampling points along the stream pinpointed a concentrated groundwater flow path for NO_3-N contributions in the lower part of the watershed, where most cropland is located. Between two sample points, the concentration of NO_3-N in Cullers Run increased from 0.84 to 2.15 mg/l. Operating under monetary incentives created by performance-based payments, participating farmers showed a willingness to address this NO_3-N discharge by placing a CW on hay land adjacent to the concentrated groundwater flow-path discharge point. Maille *et al.* (2009) provide a more complete description of the process and incentives provided in this field experiment.

Wetland Treatment System

After discovery of this concentrated flow path for NO_3-N, identification of possible source(s) of nitrates was undertaken with groundwater monitoring wells on the agricultural land adjacent to the NO_3-N discharge. These wells were located along what was apparently (from aerial photographs) an old stream channel that intersected the adjacent agricultural land. Their purpose was to confirm the existence of nitrate rich, shallow groundwater consistent with the groundwater seeps flowing into the stream. This land had been used for many years to produce hay and did not receive commercial fertilizer. Groundwater quality sampling was conducted in February and March 2009. While somewhat equivocal, NO_3-N concentration results from the well closest to the stream and measured groundwater inflow to the stream supported the existence of a subsurface flow path rather than field runoff. A CW system was selected as a strategically "targeted" BMP to be considered by the farmer group because of its shorter time frame to be operational compared with other alternatives (e.g., shrub plantings in the field) plus this BMP would take minimal agricultural land out of production (less than 1/10 of a hectare). See Figure 1 for its location on the watershed.

The Canaan Valley Institute (CVI) was hired in July 2009 to design a wetland treatment system. The CVI engineer came up with an initial design which was modified based on discussions with and addressing concerns of the landowner. The final design called for a 0.08-ha (0.2-acre) wetland consisting of a lined, horizontal trench to be constructed parallel to the stream. This wetland was designed to push up subsurface flows and funnel them through an anaerobic, carbon-rich environment to denitrify the groundwater, thus a horizontal subsurface flow CW. The farmer group agreed to pursue a treatment wetland system as a solution to subsurface nitrate flows in August 2009. After more than two months of negotiations, the landowner (who was not a participating farmer in the project) agreed to allow construction of the wetland in October 2009. Engaged farmer participants were instrumental in convincing the landowner to participate by both meeting with him and sending him letters.

Construction and material bids were put out in October 2009. Two bids were received and the farmer group selected the lowest bid. Construction of the treatment system occurred in November 2009. The organic material installed in the wetland consisted of surplus hay provided by one of the farmers in the group. With hay as a carbon source, the anaerobic denitrification process was assumed to begin shortly after completion of the CW. A slide show of the construction is available at the project web site: http://www.cacaponinstitute.org/wvunri.htm.

METHODS

To determine the cost efficiency of the CW as a nitrate treatment system, we needed to answer two basic questions: (1) what annual reductions of NO_3-N in Cullers Run can be attributed to the treatment system? and (2) what is the annualized cost of wetland treatment system? Both of these questions involve stochastic elements — efficacy of system, rainfall/streamflow dynamics, and length of time that the system will continue to treat NO_3-N. The circumstances of answering the first question were that the CW was installed as a final action of a grant project. Thus, funding limitations required relatively inexpensive instream monitoring of NO_3-N rather than a more comprehensive CW inflow *vs*. outflow monitoring of groundwater. To assess the cost efficiency of the second question, there needs to be a basis for making cost comparisons with other nitrate reduction alternatives. A literature review was conducted to determine per unit cost estimates of nitrogen removal.

Post-installation, there were indications of NO_3-N removal efficacy. Groundwater seeps were sampled above and below the system 17 times between February 2010 and September 2011. Above the system, nitrate levels averaged 6.09 mg/l whereas below the system the average level was 4.31 mg/l. This 30% reduction was attributed to the treatment system. In addition, a limited amount of water quality data (28 observations) existed for NO_3-N concentrations in Cullers Run sampled within 30 m above and below the wetland treatment system pre- and post-installation. These data showed that on average, the treatment system reduced the increase in stream NO_3-N concentrations from 62 (pre-installation) to 46% (post-installation).

Much more data existed for water quality samples taken 2.5 km upstream (above) and 0.6 km downstream (below) that were taken as a part of the field experiment. In total, 99 water quality samples were used as observations between April 2006 and September 2011. From these data, a regression model was formulated to explain streamflow concentrations of NO_3-N (CONC) and the impact of installation of the wetland treatment system. The basic model utilized was as follows:

$$\mathrm{CONC} = f(\mathrm{EXPER}, \mathrm{FLOW}, \mathrm{PREVMON}, \mathrm{SEASON}) \quad (1)$$

CONC was expressed as a function of three physical factors — streamflow (FLOW), previous month's rainfall total (PREVMON) acquired from a nearby National Oceanic and Atmospheric Administration weather station, and season when a sample was

TABLE 1. Description of Variables Used in the Wetland Treatment Analysis.

Variable	Description	Descriptive Statistics		
		Mean (range)	Std. Error	*N*
ABOVE CONC	mg/l of NO_3-N, sample taken at Cullers Run at 2nd bridge, upstream from the treatment system	0.64 (0.05-1.98)	0.043	99
BELOW CONC	mg/l of NO_3-N, sample taken at Cullers Run at Route 259, downstream from the treatment system	1.67 (0.26-5.28)	0.083	99
EXPER	1 = payment for environmental services experiment was on-going for sample, 0 = sample prior to experiment	0.46 (0-1)	0.050	99
FLOW	m^3/s flow in Cullers Run[1]	0.243 (0.0006-1.69)	0.037	99
PREVMON	Total rainfall in cm for the previous 30 days from the sample observation (Lost River weather station)	8.388 (0.483-23.317)	0.436	99
SEASON	1 = sample taken during growing season (April 15-October 15), 0 = sample taken during nongrowing season (October 16-April 14)	0.585 (0-1)	0.049	99
TREAT	1 = sample taken after wetland treatment system was installed, 0 = sample taken prior to installation	0.222 (0-1)	0.041	99

[1]Flow as measured directly (from February 2007 through October 2010). Other times flow was projected from a statistical relationship between Waites Run (with a U.S. Geological Survey gauge) and Cullers Run (flow set at 0.01 m^3/s when this relationship gave a negative value).

taken, growing *vs*. nongrowing season (SEASON). The EXPER variable was included to encompass samples taken during the field experiment (2007-2009) as a treatment variable. This variable was coded as either zero for CONC samples taken outside the treatment period or one for CONC samples within the treatment period. Table 1 contains an explanation and descriptive statistics of all the variables included in the regression model.

Separate regression models were estimated for samples taken above the treatment system *vs*. samples taken below the treatment system. Three different functional forms were analyzed for the relationship between CONC and FLOW in Equation (1): linear, log/log, and nonlinear models. As the exact impact of the CW on CONC was not known, Equation (1) was evaluated for a structural change in the regression parameters for those samples taken prior to installation *vs*. post-installation of the CW treatment system. A Chow test (Gujarati, 2003) was performed comparing the 77 sample observations taken before November 2009 with the 22 observations taken after that date. Dummy variables multiplied times the intercept and/or independent variables were added to the model if the Chow test showed a statistically significant *F*-statistic.

After testing for structural change, the different function forms were evaluated with *J*-tests to determine the best fit model for the data (Greene, 2011). Maddala's (1992) four-step procedure was used where the predicted dependent variable from one model was included in a second model regression equation. A statistically significant coefficient for this predicted dependent variable indicated the second model did not fully explain dependent variable variation.

To compute the NO_3-N reduction in Cullers Run from the wetland treatment system, a "differences in differences" approach was used to assess reductions in NO_3-N concentrations. The equation for this approach is shown below:

$$\begin{aligned}\text{Reduced CONC} \\ &= (\text{BELOW CONC} - \text{ABOVE CONC})_{\text{prior}} \\ &- (\text{BELOW CONC} - \text{ABOVE CONC})_{\text{post}}\end{aligned} \tag{2}$$

This approach compared the differences in CONC below and above the CW prior to *vs*. post-installation. Thus, structural changes in the regression equations were accounted for in the ABOVE CONC regression model that may have occurred separate from the installation of the CW.

The reduced CONC from Equation (2) was used along with base-flow computations to compute an average annual reduction in NO_3-N quantities attributed to the CW system. Four different information sources were considered for computations of base flow: (1) a computed base-flow regression equation applied to Cullers Run from Tiruneh (2007); (2) a computed base-flow regression equation applied to Cullers Run from Stuckey (2006); (3) use of the PART computerized base-flow estimation (Rutledge, 1998) on a comparable watershed with a U.S. Geological Survey (USGS) gauge (Waites Run) and projected onto Cullers Run with a regression of monthly flows between April 2007 and December 2009; and (4) use of actual data collected at Cullers Run — average of monthly low flow readings (July 2008 to December 2009) separated into growing and nongrowing seasons.

TABLE 2. Constructed Wetland Costs, 2009 Dollars.

Cost Component	Expenses
CW design by CVI	US$5,129
CVI oversight of installation	US$7,743
Nonbid materials (fabric and plants)	US$3,450
Landowner compensation	US$2,000
Installation bid	US$11,140
Total	US$29,462

Note: CW, constructed wetland; CVI, Canaan Valley Institute.

The cost estimate for installation of the wetland treatment system was derived from actual spending during the field experiment. Table 2 shows a breakdown of the US$29,462 in monetary expenses. This cost estimate reflects design and oversight management by CVI, but not the costs of researchers' and farmer involvement in the decision-making process to site the wetland. In addition, farmer contribution of organic matter for the CW (leftover hay) was not included. Annual operation and maintenance (O&M) costs were assumed to be zero based on our experience to date of no O&M costs having incurred since the CW was installed plus no future plans for O&M expenses.

This installation cost was considered on the low side of wetland costs. Kadlec and Knight (1996) reported a median cost for subsurface flow wetlands to be US$358,000/ha in 1993 dollars. For a 0.08 ha wetland in 2009 dollars, this median cost computes to US$43,000. Installation costs were amortized over 5-, 10-, 15-, 20-, 25-, and 30-year operating lives with discount rates ranging from 1 to 4%. Annualized costs ranged from over US$6,600 to under US$1,200 and were reduced by the use of a lower discount rate and/or a longer operating life (Figure 2). Cost comparisons with other nitrogen removal technologies included a literature review of published and on-line sources for per unit nitrogen removal costs with an emphasis on estimates from the mid-Atlantic region. Per unit costs from the literature were indexed to 2009 dollars using the prices paid index for production items from the National Agricultural Statistics Service.

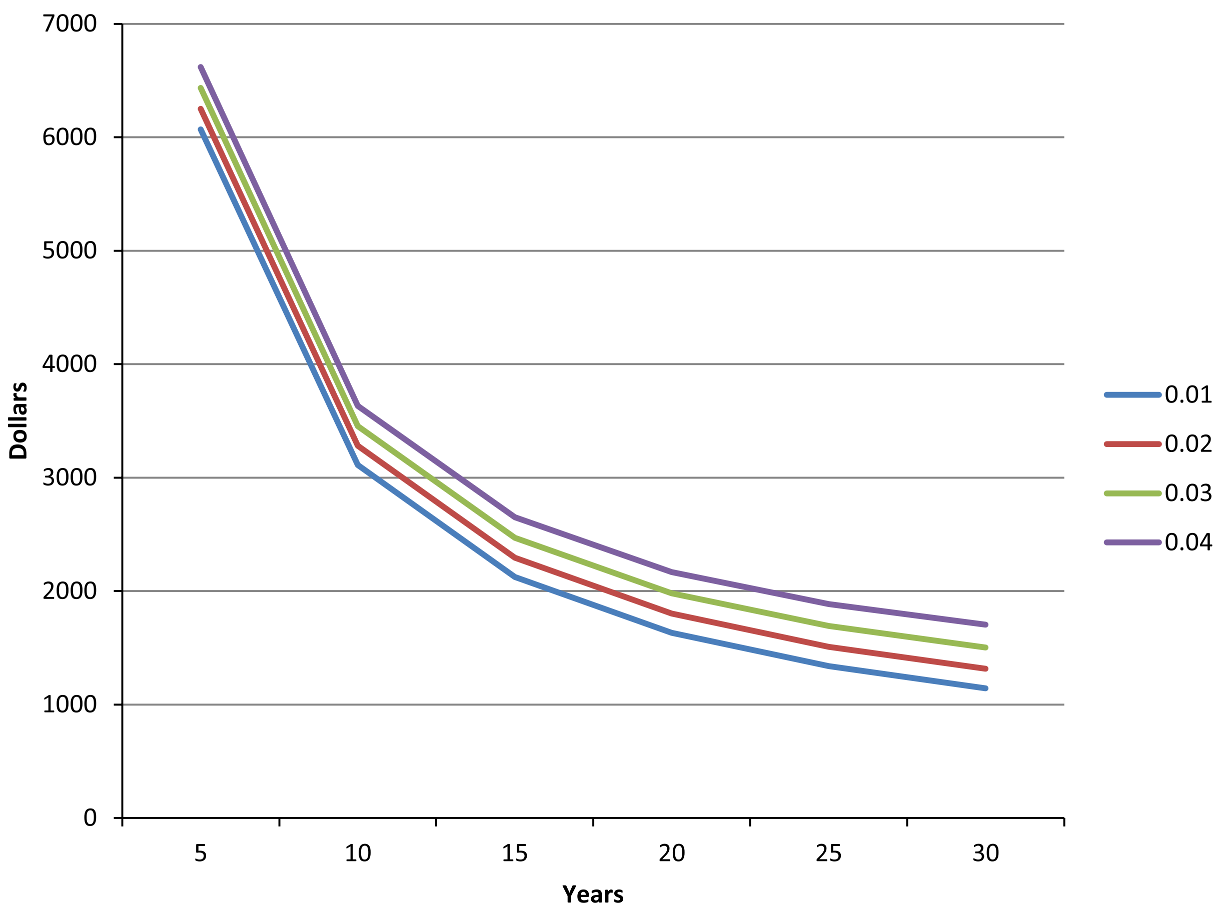

FIGURE 2. Annualized Costs for the Constructed Wetland over Four Discount Rates and Six Operating Lives.

TABLE 3. Structural Change and *J*-Test Results for ABOVE CONC and BELOW CONC Models.

Functional Form	Chow Test	*J*-Test *vs.* Linear	*J*-Test *vs.* Nonlinear	*J*-Test *vs.* Log-Log
	ABOVE CONC Model			
	***F*-Statistic**	***t*-Ratio of Predicted Value in Regression Model**		
Linear	$F_{5,89}$ = 1.015	-	0.058	0.206
Nonlinear	$F_{6,87}$ = 1.356	3.146*	-	1.339
Log-Log	$F_{5,89}$ = 4.050*	3.016*	1.731	-
	BELOW CONC Model			
Linear	$F_{5,89}$ = 2.494**	-	3.473*	−0.030
Nonlinear	$F_{6,87}$ = 1.992	1.659	-	0.097
Log-Log	$F_{5,89}$ = 4.644*	2.722*	4.130*	-

Note: Statistical significance at *1% and **5% levels, respectively.

RESULTS

F-statistics from the Chow tests showed evidence of structural change in model parameters after November 2009 for the linear model of BELOW CONC and both log/log models (Table 3). For the BELOW CONC model, *J*-tests revealed that a log/log functional form was clearly superior to linear and nonlinear functional forms. The predicted values for the log-log model had statistically significant coefficients within the linear and nonlinear models, but insertion of predicted values for linear and nonlinear into the log-log model were statistically insignificant (Table 3). The results were somewhat less clear for the ABOVE CONC model. Here, the log-log model was superior to the linear model, but only marginally better than the nonlinear model (at a 10% statistical significance rate).

The results of the log-log functional form regression analyses with statistically significant shift variables are presented in Table 4. For the ABOVE CONC model, the impact of FLOW was increased after November 2009. In the BELOW CONC model, growing season had a greater reduction in NO_3-N concentrations after November 2009. Each model was statistically significant in explaining NO_3-N concentration.

The FLOW variable impacted nitrate concentrations differently between models. For the ABOVE CONC model, greater flow increased concentration. The impact of FLOW on CONC increased in the above model after November 2009, perhaps due to the conclusion of the field experiment. Conversely, the BELOW CONC model had FLOW with a negative impact on NO_3-N concentration. Thus, increased flow raised stream concentrations of nitrates in the largely forest and pasture portion of the watershed above the treatment system — as was observed in the control watershed. Once the crop-intensive lower portion of the watershed was added, the FLOW variable showed the opposite effect on concentration.

TABLE 4. Regression Results for ABOVE CONC and BELOW CONC Models.

Variable	ABOVE CONC[1]	BELOW CONC[1]
CONSTANT	0.355 (0.192)[2]	0.086 (0.061)
EXPER	−0.298** (0.122)	−0.156* (0.039)
FLOW[1]	0.512* (0.080)	−0.103* (0.025)
PREVMON	−0.017 (0.017)	0.015* (0.005)
SEASON	−0.121 (0.144)	−0.100** (0.047)
Shift variable — FLOW	0.366* (0.106)	
Shift variable — SEASON		−0.308* (0.066)
Number obs.	99	99
$F_{5,93}$	18.56*	9.55*
Adj. R^2	0.473	0.304

Note: Statistical significance at *1% and **5% levels, respectively.
[1]Expressed in natural logarithms.
[2]Standard error.

The EXPER variable (during 2007-2009 field experiment) had statistically significant coefficients in both the above model ($p < 0.016$) and the below model ($p < 0.001$). This reduction reflected primarily short-term activities undertaken by farmers to reduce NO_3-N discharges. These actions included the planting of a cover crop, reduced poultry litter applications, and installation of temporary fencing to keep livestock out of the stream. One long-term BMP (improved manure storage and winter feeding) was put in place upstream of the ABOVE CONC sampling point on the watershed.

The SEASON shift variable had a statistically significant coefficient for the below model ($p < 0.0005$), but not for the above system model. This shift variable was interpreted as the impact of the treatment system — reducing NO_3-N concentrations in Cullers Run primarily during the growing season. While debated in the literature, there is no strong evidence that subsurface CW for groundwater operate differ-

TABLE 5. Cullers Run Base Flow and NO_3-N Reduction Estimates from the Constructed Wetland.

Source	Range of Base-Flow Estimates for Cullers Run	NO_3-N Reduction Estimates (kg/yr)
Tiruneh (2007)[1]	0.024-0.082 m^3/s	140-486
Stuckey (2006)[2]	0.095-0.190 m^3/s	563-1,121
Rutledge (1998) and software available at http://water.usgs.gov/ogw/part/	560,000 m^3/month	1,277
Cullers Run: average of monthly low flow readings (July 2008-December 2009)	0.014 m^3/s (growing) 0.055 m^3/s (nongrowing)	Mean: 80 Range: 41-114[3]

[1]Annual base-flow recurrence intervals between 2 and 20 years.
[2]90% confidence interval.
[3]95% confidence interval SEASON shift variable in BELOW CONC regression model.

ently based upon season (Kadlec and Knight, 1996; Vymazal, 2011). Our interpretation for this seasonal impact from the CW was that nitrate concentrations within groundwater flows have a larger influence on stream nitrate concentrations during the growing season when streams like Cullers Run are more frequently at base flow and thus are more impacted by groundwater contributions compared to the nongrowing season. The end results are that nitrate concentration reductions resulting from the CW were revealed in surface stream water quality samples during the growing season.

Equation (2) was used to compute the estimated mean reduction in NO_3-N concentration in Cullers Run due to the treatment system. Flow estimates were based on Cullers Run average low flows in growing season as base-flow discharge levels (Table 5). The estimated mean impact of the treatment system at low flows, accounting for the shift in flow impact in the ABOVE CONC model, was to reduce NO_3-N concentrations in the stream during the growing season by about 0.37 mg/l, with a 95% confidence interval of 0.19-0.54 mg/l. At average flow during the growing season (0.215 m^3/s), the mean reduction was 0.14 mg/l, a 17% reduction in stream NO_3-N concentration. This percentage reduction was slightly lower than the percentages computed from seep and stream sampling data taken around the CW, as would be expected because the regression model accounted for streamflow impacts.

This 17% reduction from installation of just one CW treatment system is substantial given that Inamdar *et al.* (2001) report reductions of 29 and 41% in total nitrogen (TN) stream concentrations were possible when all farms within a watershed implemented agricultural land BMPs (no tillage, filter strips, nutrient management, etc.). This same study reported an increase in NO_3-N concentrations between pre- and post-BMP concentrations. Thus, CW treatment systems can be effective in controlling NO_3-N lost to leaching and groundwater flows, which have been shown not to be reduced by land management BMPs (Inamdar *et al.*, 2001; Flores-López *et al.*, 2010; Gassman *et al.*, 2010).

Using the "difference in differences" approach, NO_3-N load reduction estimates attributed to the CW treatment system were computed for each of the four projected base-flow computations (Table 5). Even though all load reduction estimates were computed for only the growing season period (April 15-October 15), they varied widely. The Cullers Run low flow estimates were selected as the most representative because they accounted for seasonal flow fluctuations between growing and nongrowing seasons whereas other flow computations were annual averages.

A representative cost per kilogram (kg) of NO_3-N was computed using the following assumptions: a 15-year CW life with no maintenance costs, a 3% discount rate, and a mean reduction of 80 kg of NO_3-N annually from the low monthly flows base-flow computation. These assumptions were regarded as conservative. Kadlec and Knight (1996) note that properly designed subsurface wetlands should last for decades in terms of pore space and Vymazal (2011) observed that horizontal, subsurface systems maintained their nutrient removal efficiency over a 10-year period. Jamison and Jamison (2011) recommend a 3% discount rate to adequately consider the future; and a reduction of 80 kg/yr was the lowest computed mean value of NO_3-N reductions among the four base-flow estimates. The 95% confidence interval of 80 kg/yr (40-114 kg/yr) includes the denitrification estimate of 49 kg/yr provided by CVI, which was based upon CW performance rates obtained from the literature.

Using these assumptions, a representative unit cost was estimated at US$30.85/kg (US$13.99/pound) in 2009 dollars. Based on the 95% confidence interval for the SEASON shift variable coefficient from the regression equation, this unit cost estimate has a range US$21.65-US$60.19/kg. See the Appendix for a complete listing of unit costs over different CW operating lives, discount rates, and at the 95% confidence interval range of NO_3-N load reductions. Per unit costs for the CW treatment system obtained as low as US$10/kg only at 30 years, a 1% discount rate, and projecting a 114 kg/yr load reduction.

How does a unit cost of US$30.85/kg compare with other treatment costs for nitrogen? From the first two years of our field experiment (2007-2009), farmer payments and cost share expenses were totaled and divided by an estimated annual NO_3-N kg reduction from the EXPER coefficient (240 kg/yr.). This resulted in an estimate of over US$21.4/kg. This estimate was lower than the CW treatment system unit cost, but did not include the costs of a winter feeding and manure storage facility installed and paid for by one of the participating farmers without cost share from the field experiment.

To compare the CW treatment system with point source treatment costs, not much information was found in the published literature on average or marginal costs to reduce nitrogen discharges from wastewater treatment plants (WWTPs). The cost information that is available was either based on other discharges (Fraas and Munley, 1984; McConnell and Schwarz, 1992) or not well enough reported so that nitrogen removal costs can be computed (Horan *et al.*, 2002; Hanson and McConnell, 2008). There are indications in the literature that nitrogen removal costs from WWTPs are under US$15/kg (Piehler and Smyth, 2011; Pollack *et al.*, 2013). What is clear is that WWTP average and marginal treatment costs vary substantially by plant size and effluent standards (McConnell and Schwarz, 1992). Specific to West Virginia and most relevant to this site are small WWTPs. Khatri-Chhetri (2012) computed the additional costs for small WWTPs to reduce TN discharges to meet an 8.0 mg/l standard along the Greenbrier River. Among seven plants, these costs were computed to have a weighted average (in 2011 dollars) of US$118/kg (US$53/pound). For individual WWTPs, per unit costs to remove TN ranged from US$29 to US$2,780/kg. Thus, the CW treatment system per unit costs was at the low end of this cost range.

Examples of per unit cost estimates found in the research literature have been dominated by those formulated from economic models rather than calculation from actual discharge treatment. For nitrogen reductions from nonpoint agricultural BMPs, one example provided by Stephenson *et al.* (2010) in Virginia found that the least expensive nonpoint option was fertilizer reduction at a range from US$10 to US$68/kg of nitrogen. These computations were made with projected costs and reported nitrogen reduction rates from the Virginia Department of Environmental Quality after a minimum of five BMPs are installed as a baseline. Petrolia and Gowda (2006) computed the costs of reducing nitrogen discharges from tile drained agricultural land using a watershed scale model in southern Minnesota. For abatement levels between 10 and 20%, the costs to reduce fertilizer use ranged between US$3.17 and US$12.86/kg of nitrogen (in 2009 dollars). Finally, Ribaudo *et al.* (2001) computed a US$1.79/kg economic welfare loss (expressed in 2009 dollars) from fertilizer reductions for the entire Mississippi River basin to achieve a 15% abatement of nitrogen. Wetland costs were projected at between US$5.60 and US$6.50/kg of nitrogen (2009 dollars) for a comparable abatement level.

The above per unit cost estimates for reduced nitrogen show that fertilizer reduction is generally much less costly than abatement by wetland treatment. However, when oversight management and some type of yield loss payment are included to induce farmer participation and compliance for lower fertilizer applications, Green *et al.* (2011) show that per unit costs ranged from US$8 to US$27/kg of nitrogen for a field edge reduction in nitrogen on Pennsylvania corn fields. These per unit costs are sensitive to corn price increases. They increased US$0.50-US$2/kg of nitrogen for every dollar per bushel increase in corn prices.

CONCLUSIONS

A CW was strategically placed to intercept an underground flow of nitrogen discharging into Cullers Run in Hardy County, West Virginia. Its placement was the result of a farmer engagement process to reduce nitrogen losses from the watershed. This CW has resulted in groundwater being moved up into this wetland and plant growth on the CW has been acceptable with additional shrubs on the stream side having been established. The surface of the wetland, however, is very rarely wet. Overall, as a result of this CW, we found: a statistically significant reduction in NO_3-N concentrations in Cullers Run of 0.14 mg/l during the growing season (a 17% reduction); a conservative estimate of reduced NO_3-N by 80 kg/yr; and, conservatively, the unit cost to remove NO_3-N should be around US$30/kg. In terms of estimating costs, our approach had the advantages of using actual, statistically generated instream NO_3-N reduction estimates along with on-the-ground installation costs to derive per unit costs.

One environmental disadvantage of CW treatment of nitrates is that the final products of denitrification within a CW include nitrogen gas (N_2), nitrous oxide

(N_2O), or nitric oxide (NO) (Kadlec and Knight, 1996). While only a small percentage of nitrogen is converted to N_2O, this gas is a powerful greenhouse gas and ozone layer destroyer (Schlesinger, 2008). Thus, the use of CW treatment systems to remove nitrogen from groundwater involves a slight tradeoff between less nitrogen in surface waters and atmospheric deterioration.

Overall, the lowest cost approach to nitrogen reductions in aquatic environments should be reduced fertilizer applications in agriculture. However, convincing many farmers to reduce their fertilizer use would involve a substantial level of social engagement and/or persuasion plus yield reduction compensation which may negate its cost advantage (see Green *et al.*, 2011). In our study, a group decision-making process among farmers in a watershed resulted in an interception approach to nitrogen that was achieved by installation of a single CW treatment system. While social engagement and persuasion was certainly an important part of installing a CW treatment system, we make the observation that typically fewer farmers must be convinced to make management changes with an interception approach to nitrogen. An interception approach was appropriate for this study area given how nitrogen is being transported within this watershed and its acceptability to farmers as a method to problem-solve reducing nitrogen losses.

Utilization of CW treatment systems for an interception approach requires an understanding of the hydrogeology of watersheds. Research has shown that roughly half of nitrogen in streams within the Chesapeake Bay watershed comes from groundwater transport (Bachman *et al.*, 1998; Sprague *et al.*, 2000). Location of these groundwater source contributions is critical to employing an interception approach. Methods to efficiently locate the intersection of ground and surface water are an active area of research at West Virginia University where the use of aerial drones with infrared sensors to detect stream water temperature differences is currently being investigated.

In summary, we can identify some advantages of a CW treatment system for reducing stream level nitrates: it treats a nitrogen source that alteration of surface land management does not reduce; it consists of only fixed costs which negates future rising per unit costs (as compared with fertilizer reduction BMP costs which rise with crop price increases), and its installation depends upon fewer, but more targeted farmer cooperation efforts. However, we cannot conclude that the use of an interception approach for nitrates with a CW treatment system offers a low cost compared with nonpoint BMPs. This system can only be cost competitive with other BMPs if it lasts for decades and abates load reductions at the high end of its statistically estimated range.

Lastly, confounding factors to this cost estimate that are not addressed in this research include: lag times between treatment of groundwater and surface water reduction in NO_3-N; other possible factors that may explain declining nitrogen levels in the stream such as improved air quality or changing land management practices; and how long will the CW remove nitrogen? There is uncertainty about nitrogen removal rates once the CW exhausts its initial carbon input and reaches an equilibrium point where it depends solely upon carbon generated from wetland plant growth. Further research on the stochastic elements of treatment system life, efficacy level, and rainfall/streamflow connection could be developed to create Monte Carlo simulations that generate mean and confidence intervals for per unit cost estimates. In addition, an examination of system efficacy over time to see if changes occur would be a useful research endeavor.

APPENDIX

TABLE A1. Mean Costs per kg of NO_3-N Removed with a 95% Confidence Interval Using Projected Removal Rates from Average Cullers Run Monthly, Low Flow Readings.

Operating Life (years)	Discount Rates 1%	2%	3%	4%
	US$/kg			
5	75.88 (53.25-148.06)	78.13 (54.83-152.45)	80.41 (56.43-156.91)	82.72 (58.05-161.41)
10	38.88 (27.29-75.87)	41.00 (28.77-80.00)	43.17 (30.30-84.24)	45.40 (31.86-88.60)
15	26.56 (18.64-51.83)	28.66 (20.11-55.92)	30.85 (21.65-60.19)	33.12 (23.24-64.63)
20	20.41 (14.32-39.82)	22.52 (15.81-43.95)	24.75 (17.37-48.30)	27.10 (19.02-52.87)
25	16.72 (11.73-32.63)	18.86 (13.24-36.81)	21.15 (14.84-41.27)	23.57 (16.54-46.00)
30	14.27 (10.01-27.84)	16.44 (11.54-32.08)	18.79 (13.19-36.66)	21.30 (14.95-41.56)

ACKNOWLEDGMENTS

This project was funded under the USDA/CSREES National Research Initiative project number 2006-35102-17261. Peter Maille assisted in the development and implementation of this project. The farmers and community of Cullers Run watershed are to be congratulated on their interest and passion in protecting their watershed.

LITERATURE CITED

Bachman, L.J., B.D. Lindsey, J.W. Brakebill, and D.S. Powars, 1998. Groundwater Discharge and Base-Flow Nitrate Loads of Nontidal Streams, and Their Relation to a Hydrogeomorphic Classification of the Chesapeake Bay Watershed, Middle Atlantic Coast. U.S. Geological Survey Water-Resources Investigations Report 98-4059, 71 pp. http://md.water.usgs.gov/publications/wrir-98-4059/.

Cacapon Institute, 2002. Final Report to the U.S. Fish and Wildlife Service on Water Quality Studies in the Cacapon River's Lost and North River Watersheds in West Virginia. High View, West Virginia.

Flores-López, F., Z.M. Easton, and T.S. Steenhuis, 2010. A Multivariate Analysis of Covariance to Determine the Effects of Near-Stream Best Management Practices on Nitrogen and Phosphorus Concentrations on a Dairy Farm in the New York Conservation Effects Assessment Project Watershed. Journal of Soil and Water Conservation 65(6):438-449, doi: 10.2489/jswc.65.6.438.

Fraas, A. and V. Munley, 1984. Municipal Waste Treatment Costs. Journal of Environmental Economics and Management 11:28-38.

Gassman, P.W., J.A. Tisl, E.A. Palas, C.L. Fields, T.M. Isenhart, K.E. Schilling, C.F. Wolter, L.S. Seigley, and M.J. Helmers, 2010. Conservation Practice Establishment in Two Northeast Iowa Watersheds: Strategies, Water Quality Implications, and Lessons Learned. Journal of Soil and Water Conservation 65(6):381-392, doi: 10.2489/jswc.65.6.381.

Green, T., J. Cubie, R. Ressl, M. Anderson, J. Baird, B. Brandt, J. McCarthy, M. Kieser, and J. Gilbert, 2011. Nitrogen Management Performance Guarantee System for Corn Producers in Pennsylvania: Feasibility to Improve Water Quality in Chesapeake Bay. Report submitted to the American Farmland Trust, 56 pp. http://www.farmland.org/documents/FeasibilityStudyFINAL03-01-12.pdf, *accessed* September 2012.

Greene, W.H., 2011. Econometric Analysis, 7th Edition. Prentice-Hall Inc., Boston, Massachusetts, ISBN-13: 978-0131395381.

Gujarati, D.N., 2003. Basic Econometrics, 4th Edition. McGraw-Hill Higher Education, New York City, New York, ISBN: 0-07-233542-4.

Hanson, J.C. and K.E. McConnell, 2008. Simulated Trading for Maryland's Nitrogen Loading in the Chesapeake Bay. Agricultural and Resource Economics Review 37(2):211-226.

Horan, R.D., D.G. Abler, J.S. Shortle, and J. Carmichael, 2002. Cost-Effective Point-Nonpoint Trading: An Application to the Susquehanna River Basin. Journal of the American Water Resources Association 38(2):467-477.

Inamdar, S.P., K.M. Brannan, P.W. McClellan, and S. Mostaghimi, 2001. BMP Impacts on Sediment and Nutrient Yields from an Agricultural Watershed in the Coastal Plain Region. Transactions of the ASAE 44(5):1191-1200.

Jamison, D.T. and J. Jamison, 2011. Characterizing the Amount and Speed of Discounting Procedures. Journal of Benefit-Cost Analysis 2(2), doi: 10.2202/2152-2812.1031.

Kadlec, R.H. and R.L. Knight, 1996. Treatment Wetlands. CRC Press, Boca Raton, Florida, ISBN: 0-87371-930-1.

Khatri-Chhetri, A., 2012. A Feasibility Assessment of Water Quality Trading in the Greenbrier River Watershed, West Virginia. Ph.D. Dissertation, West Virginia University, Morgantown, West Virginia.

Maddala, G.S., 1992. Introduction of Econometrics. Macmillan Publishing Co., New York City, New York, ISBN-13: 978-0023745454.

Maille, P., A. Collins, and N. Gillies, 2009. Performance-Based Payments for Water Quality: Experiences from a Field Experiment. Journal of Soil and Water Conservation 64(3):85-87.

McConnell, V.D. and G.E. Schwarz, 1992. The Supply and Demand for Pollution Control: Evidence from Wastewater Treatment. Journal of Environmental Economics and Management 23:54-77.

Moore, R.B., C.M. Johnston, R.A. Smith, and B. Milstead, 2011. Source and Delivery of Nutrients to Receiving Waters in the Northeastern and Mid-Atlantic Regions of the United States. Journal of the American Water Resources Association 47(5):965-990, doi: 10.1111/j.1752-1688.2011.00582.x.

Petrolia, D.R. and P.H. Gowda, 2006. An Analysis of the Role of Tile-Drained Farmland under Alternative Nitrogen Abatement Policies. Journal of Agricultural and Resource Economics 31(3):580-594.

Phillips, S.W. and B.D. Lindsey, 2003. The Influence of Groundwater on Nitrogen Delivery to the Chesapeake Bay. U.S. Geological Survey Fact Sheet FS-091-03, 4 pp. http://md.water.usgs.gov/publications/fs-091-03/html/, *accessed* August 2012.

Piehler, M.F. and A.R. Smyth, 2011. Habitat-Specific Distinctions in Estuarine Denitrification Affect Both Ecosystem Function and Services. Ecosphere 2(1):art12.

Pollack, J.B., D. Yoskowitz, H.C. Kim, and P.A. Montagna, 2013. Role and Value of Nitrogen Regulation Provided by Oysters (*Crassostrea viginica*) in the Mission-Aransas Estuary, Texas, USA. PLoS ONE. http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0065314#pone.0065314-Piehler1, *accessed* September 2013.

Ribaudo, M.O., R. Heimlich, R. Claassen, and M. Peters, 2001. Least-Cost Management of Nonpoint Source Pollution: Source Reduction Versus Interception Strategies for Controlling Nitrogen Loss in the Mississippi Basin. Ecological Economics 37(2):183-197.

Ribaudo, M.O., R.D. Horan, and M.E. Smith, 1999. Economics of Water Quality Protection from Nonpoint Sources: Theory and Practice. Agricultural Economic Report No. 782, U.S. Department of Agriculture, Economic Research Service, Resource Economics Division, Washington, D.C.

Rutledge, A.T., 1998. Computer Programs for Describing the Recession of Ground-Water Discharge and for Estimating Mean Ground-Water Recharge and Discharge from Streamflow Records – Update. U.S. Geological Survey, Water-Resources Investigations Report 98-4148, 52 pp. http://pubs.usgs.gov/wri/wri984148/, *accessed* July 2012.

Schlesinger, W.H., 2008. On the Fate Anthropogenic Nitrogen. Proceedings of the National Academy of Sciences of the United States of America 106(1):203-208.

Sprague, L.A., M.J. Langland, S.E. Yochum, R.E. Edwards, J.D. Blomquist, S.W. Phillips, G.W. Shenk, and S.D. Preston, 2000. Factors Affecting Nutrient Trends in Major Rivers of the Chesapeake Bay Watershed. U.S. Geological Survey Water-Resources Investigations Report 00-4218, 109 pp. http://va.water.usgs.gov/online_pubs/WRIR/00-4218.htm, *accessed* August 2012.

Stephenson, K., S. Aultman, T. Metcalfe, and A. Miller, 2010. An Evaluation of Nutrient Trading Options in Virginia: A Role for Agricultural Nonpoint Sources? Water Resources Research 46:W04519, doi: 10.1029/2009WR008228.

Stuckey, M.H., 2006. Low-Flow, Base-Flow, and Mean-Flow Regression Equations for Pennsylvania Streams. U.S. Geological Service Scientific Investigations Report 2006-5130, 84 pp. http://pubs.usgs.gov/sir/2006/5130/pdf/sir2006-5130.pdf, *accessed* July 2012.

Tiruneh, N., 2007. Basin-Wide Annual Baseflow Analysis for the Fractured Bedrock Unit in the Potomac River Basin. Interstate Commission on the Potomac River Basin Report No. 07-6, 56 pp. http://www.potomacriver.org/2012/publicationspdf/ICPRB07-6.pdf, *accessed* July 2012.

USEPA (U.S. Environmental Protection Agency), 2010. Chesapeake Bay Total Maximum Daily Load for Nitrogen, Phosphorus and Sediment. http://www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html, *accessed* October 2012.

Vymazal, J., 2007. Removal of Nutrients in Various Types of Constructed Wetlands. Science of the Total Environment 380:48-65.

Vymazal, J., 2011. Long-Term Performance of Constructed Wetlands with Horizontal Sub-Surface Flow: Ten Case Studies from the Czech Republic. Ecological Engineering 37:54-63.