Science Assessment to Support an Illinois Nutrient Loss Reduction Strategy

Mark B. David
Gregory F. McIsaac
Gary D. Schnitkey
George F. Czapar
Corey A. Mitchell

University of Illinois at Urbana-Champaign
College of Agricultural, Consumer and Environmental Sciences
Urbana, Illinois

Peer reviewed and revised final report

May 6, 2014

Table of Contents

Introduction

Illinois is a highly agricultural state, but with several major metropolitan areas. There are more than 22 million acres of corn and soybeans (60% of the state's land area), much of it tile drained, and a human population near 13 million (fifth nationally). Consequently, both point and non-point sources of nitrogen (N) and phosphorus (P) are added to the streams and rivers of the state, with these nutrients being transported to the Mississippi River and the Gulf of Mexico (David and Gentry, 2000; David et al., 2010; Jacobson et al., 2011). The Mississippi River/Gulf of Mexico Watershed Nutrient Task Force has a goal to reduce the hypoxic zone in the northern Gulf of Mexico to a five-year running average of 5,000 km^2 (~1,900 square miles) by 2015 (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2008). To meet this goal, a 45% reduction in 1980 through 1996 stream loads of Mississippi River Basin (MRB) total N and total P was recommended by the USEPA Science Advisory Board in 2007 (USEPA, 2007). Because nitrate-N is thought to be the primary nutrient leading to formation of the hypoxic zone each summer, with total P secondary, the focus for reducing total N is to reduce nitrate-N loads in the MRB (USEPA, 2007). This report provides the scientific basis for a nutrient reduction strategy for Illinois by: 1) determining current conditions in Illinois of nutrient sources and export by rivers in the state from both point and non-point sources; 2) describing practices that could be used to reduce these losses to surface waters and along with estimates of practice effectiveness throughout Illinois; and 3) estimating the costs of statewide applications of these methods to reduce nutrient losses to meet Gulf of Mexico hypoxia goals.

In this analysis, we utilized United States Geological Survey (USGS) stream flow data and Illinois Environmental Protection Agency (IEPA) and USGS N and P concentrations to estimate major watershed stream loads for the state for the 1980 through 2011 water years. In addition, we directly estimated point source loads of N and P, and then calculated non-point sources by difference. These estimates were compared to previously published values by David and Gentry (2000) to provide perspective to earlier studies in Illinois. Urban non-point sources were estimated using published values and urban land areas in Illinois. We then applied a 45% reduction target or goal to the 1980 through 1996 stream load of nitrate-N and total P for the state, to determine the target or goal of a nutrient reduction strategy for Illinois. The next step was estimating nitrate-N and total P yields (both point and non-point) for the 8-digit HUC (hydrologic unit code, a USGS system of dividing the United States in hydrologic units at various scales) watersheds in Illinois, so that nutrient yields and listed waters (stream segments and lake acres listed for dissolved oxygen, total P, nitrate-N, aquatic plants, or aquatic algae) in the watersheds at this scale could be ranked to determine critical watersheds for nutrient reductions. We then estimated reductions in nutrient loads for various point and non-point practice changes, estimated costs per acre for agricultural practices and per pound (lb) of nutrient reduced for both point and non-point sources, and then scaled our estimates to the entire state. Finally, we developed various scenarios to reduce nitrate-N and total P loads by either 20 or 45%. Twenty percent was chosen to be roughly half of the 45% reduction target.

Current Conditions

Point Source Nutrient Loads

Nitrogen and P point source data are available through the United States Environmental Protection Agency (USEPA) Integrated Compliance Information System (ICIS). We began with IEPA's analysis (conducted by Mr. Robert Mosher of the Water Quality Standards Unit, Division of Water Pollution Control, Illinois Environmental Protection Agency) of total P data in ICIS, in which he received information on 1,660 point sources of P within Illinois for 2009. Mosher (2013) concluded that the ICIS tools did not allow an accurate estimation of point source P loadings in Illinois. As a result the IEPA utilized P data from 42 facilities [although 38 are referenced in Mosher (2013), Mosher provided us with a spreadsheet with data from 42 facilities because a few additional sources reported data after his report was written] provided by Illinois Association of Wastewater Agencies (IAWA) along with discussion with cooling water dischargers to recalculate P concentration and loads for the largest 108 dischargers listed in ICIS. For our analysis we added data from Decatur's publicly owned treatment works (POTW). IEPA found some important errors in the ICIS output, and recalculated the top 108 sources in the data from the ICIS output (Mosher, 2013). The 108 sources represented the largest 100 P sources in the state (and therefore most of the point source P load), as well as an additional 8 sources that provided data from the IAWA request (Mosher, 2013). Mosher (2013) used total P concentrations from either values reported by facilities in response to the IAWA request; 2) the IEPA's knowledge of the facility; or 3) the ICIS database. In our analysis of P, we examined the other major discharging facilities (hereafter referred to as "majors") in the ICIS database, a total of 263 facilities that included the top 108 previously analyzed. Majors are nearly all treatment works with design flows > 1 million gallons per day (MGD), but also include a few treatment works that score > 80 points on the NPDES Permit Rating Worksheet. As IEPA had done for the top 108 sources, we used IEPA's best estimate of the total P concentration for many of the industrial and agricultural facilities and a few POTW's that had very high total P concentrations in the ICIS database. For all others we used the USEPA ICIS value for total P, which typically was between 2.5 and 3 mg P/L. Similar to Mosher (2013), we found that the original ICIS output over-estimated the total P load by a large percentage. The ICIS estimate for the 263 majors was 29.4 million lb of total P yr^{-1}, whereas our estimate was 16.6 million lb of total P yr^{-1}. The ICIS major point source total P estimate was therefore 1.8 times too high (we believe our estimate is more accurate because we used actual data from dischargers in Illinois, instead of the modeled values used by USEPA). For the other 1,397 total P point sources in the ICIS database, we used the USEPA estimates multiplied by 0.565, based on this over-prediction. Because these 1,397 point sources were a small proportion of the overall point source P estimate and no other data were available for the wide range of sources in the data set, this was the best estimate we could make. A median of total P from the POTW's would not be appropriate to use for these varied point sources.

For N, there are fewer measurements available because many facilities have been monitoring only ammonia concentrations. We made a request through IAWA for N data for the period 2008-2012, and received data from 34 major facilities (requests went out by e-mail to all IAWA members on March 5, 2013, with a reminder on April 8, 2013 from Robin Ellison of IAWA). Three of the facilities only reported ammonia, but 31 reported total N and/or nitrate-N, with most reporting both. Some had five years of data, some only one. All reported flow as well. Facility size ranged from 1.3 to 712 MGD, with a median of 12 MGD. For typical plants (large Chicago plants excluded), the average total N concentration was 16.8 mg N/L, with a nitrate-N concentration of 14.9 mg N/L (26 facilities, most reporting data from 2008 through 2012). IEPA made a request to ICIS for all N data, and 392 sources were reported, all POTWs. They are the only point sources in Illinois with a permit for N, many fewer than the 1660 permitted for P. Because ICES reported only ammonia data for nearly all plants, only flow data could be used from this source and no ICIS N concentration data were utilized in our analysis. For the 31 plants that reported nitrate-N and/or total N data, annual loads were directly calculated. For the other 361 plants in the ICIS database (392 total sources minus 31 that reported concentrations to us), flow from ICIS and the average total N and nitrate-N concentrations reported above were used to estimate the source. Data were available for all seven Metropolitan Water Reclamation District of Greater Chicago (MWRDGC) plants for both the P and N estimates, which was important given MWRDGC has the largest plants in the state.

Table 1. Point source total P loads for entire state as well as by major river basin. The category "All other basins" includes point sources outside of the 8 major basins.

	All 1660 sources	Majors (263)
	million lb P yr^{-1}	
Rock River	1.01	0.89
Green River	0.03	0.02
Illinois River	14.6	13.8
Kaskaskia River	0.52	0.40
Big Muddy	0.21	0.17
Little Wabash	0.16	0.14
Embarras River	0.10	0.08
Vermilion River	0.22	0.20
All other basins	1.12	0.94
State Sum	18.0	16.6
State (David & Gentry, 2000)	14.7	

Table 2. Point source total N and nitrate-N loads for entire state as well as by major river basin (392 sources). The category "All other basins" includes point sources outside of the 8 major basins.

	Total N	Nitrate-N
	millions lb N yr^{-1}	
Rock River	3.94	3.48
Green River	0.11	0.09
Illinois River	75.2	64.4
Kaskaskia River	2.20	1.94
Big Muddy	1.21	1.08
Little Wabash	0.48	0.44
Embarras River	0.60	0.53
Vermilion River	1.54	1.37
All other basins	2.07	1.76
State sum	87.3	75.2
State (David & Gentry, 2000)	86.0	

Point source total P was estimated at 18.0 million lb of P yr^{-1}, with most from the major facilities in the state (16.6 million lb of P yr^{-1}) (Table 1). The estimated point source load of total N was 87.3 million lb per year, with 75.2 million lb as nitrate-N (Table 2). Most of the point source N is from northern Illinois, with large loads in the Illinois and Rock Rivers.

A comparison was made with previously published point source estimates for Illinois to see if our previous understanding of the importance of these nutrient sources was correct. Using completely different estimation techniques (per capita N in effluent) for the 1990s, David and Gentry (2000) reported a very similar estimate for total N from point sources of 86.0 million lb of N yr^{-1}. More recently David et al. (2010) estimated 123 million lb N yr^{-1} consumed in food, which would be expected to be larger than the N discharged due to gaseous N losses in wastewater treatment and N removed in biosolids. The estimate of point source P loads is larger than that predicted by David and Gentry (2000) or that predicted by Jacobson et al (2011) based on food consumption (13 million lb P yr^{-1}). This is likely due to the inclusion of industrial point sources in the current study that were not considered in the earlier work.

Urban Runoff Nutrient Loads

Urban runoff was estimated using land cover data for Illinois and published tables of nutrient loss per acre. We used two sets of land cover maps. The first was a 1991-1995 analysis by the Illinois Department of Natural Resources, published as the Illinois Land Cover: An Atlas (1996), from Landsat 4 and 5 Thematic mapper satellite imagery acquired during the 1991-1995 spring and fall seasons, with most of the data from 1992. This land cover dataset had 6 urban land uses, including: high density, medium/high density, medium density, low density, transportation, and urban grassland. The newer, 1999-2000 land cover data was a joint effort of the United States Department of Agriculture National Agricultural Statistics Service, Illinois Department of Agriculture, and the Illinois Department of Natural Resources using Landsat 5 TM and Landsat 7 RTM+ satellite imagery acquired during the spring, summer, and fall seasons of 1999 and 2000. However, these newer data only divided urban area into high density, medium/low density, and urban open space categories.

Nitrogen and P yields for urban areas were obtained from the report, "Preliminary data summary of urban storm water best management practices" (USEPA, 1999). Table 4-3 was used from this report, but is not shown here (the table number is given to provide the reader a clear reference to the source of data used). These estimates were derived from several different studies of typical urban area nutrient yields originally from Horner et al (1994). The table represents an estimate derived from several different studies of typical urban areas. We then multiplied the nutrient loads per acre by the appropriate acres for each land cover type. For the 1999-2000 land cover, we used averages of the nutrient yields from different land cover classes in Table 4-3 to match the three categories of land cover data that were available. We also used data for nitrate-N and total N urban runoff loads from a study conducted in Baltimore, Maryland (Groffman et al., 2004), total N from a study in Seattle, Washington (Herrera Environmental Consultants, 2011), total P from inputs estimated to the DuPage River, Illinois (DuPage River Salt Creek Workgroup, 2008) and an IEPA summary of total P in urban runoff from 1986 (IEPA, 1986). Each of these data sources were combined with the land cover data described above.

Land cover data indicated about 2.3 million acres of urban land in Illinois. We estimate that urban runoff is a source of about 1.5 million lb of total P yr^{-1}, 6.0 million lb of nitrate-N yr^{-1}, and 8.3 million lb of total N yr^{-1}. These are approximate values given the approach used, but are likely about the right order of magnitude. There was little difference in the estimates using the two land cover databases.

Riverine Nutrient Loads

We used stream flow and N and P concentrations for the eight major rivers in the state where data were available, which represents 74% of the state area (Table 3, Figure 1). USGS flow data and IEPA and USGS data were used to calculate annual N and P fluxes on a water year basis from 1980 through 2011 for nitrate-N, total N, dissolved reactive P, and total P. When these eight river loads were summed the results were extrapolated to represent all of the state (56,371 square miles). This generally follows methods used by David and Gentry (2000). For the Rock River, 54% of the drainage at Joslin, where the gage is located, is in Wisconsin. David and Gentry (2000) estimated the Illinois load as 46% of the load at Joslin, but we used a different method here. We calculated the load for the Rock River at Rockton, Illinois, which is mostly drainage from Wisconsin. We then subtracted the Rockton load from that at Joslin, giving us only a load from Illinois sources (3,187 square miles) that more accurately reflects Illinois sources.

Table 3. River systems, location and station number of discharge and water quality data, drainage area, and fraction of drainage area in Illinois used in estimating export of N and P by surface water from Illinois.

River system	Gage location	USGS Station number	Drainage area	Fraction in Illinois
			sq miles	%
Rock	Joslin	05446500	9,549	46
Rock	Rockton	05437500	6,362	
Green	Geneseo	05447500	1,003	100
Illinois	Valley City	05586100	26,743	93
Kaskaskia	Venedy Station	05594100	4,393	100
Big Muddy	Murphysboro	05599500	2,169	100
Little Wabash	Carmi	03381500	3,102	100
Embarras	Ste. Marie	03345500	1,516	100
Vermilion	Danville	03339000	1,290	100

To determine the annual load for a river from continuous daily flow and infrequent nutrient concentration measurements, a variety of methods can be utilized. There has been much discussion in the literature about the advantages and disadvantages of each method. Based on our assessment of the literature and current techniques available, interpolation is thought to be the best method for highly soluble nutrients such as nitrate-N in larger rivers, and because nitrate-N is a large percentage of total N interpolation can be used for total N as well. However, for P on smaller rivers there is a strong concentration response to flow and high flow loads can be underestimated with interpolation when sampling is infrequent. The USGS WRTDS (Weighted Regressions on Time, Discharge, and Season) technique (Hirsch et al., 2010) is a method that fits a relationship that includes flow, and therefore better estimates the high flow days that are critical to estimating P loads (Royer et al., 2006).

Linear interpolation to estimate daily nutrient concentrations between sampling days was conducted using SAS version 9.2 and the Proc Expand procedure. Daily flow and measured nutrient concentrations were the input data, with daily flow and daily concentration the output. With this procedure the observed values are present in the final data set, as they are not replaced with estimated values.

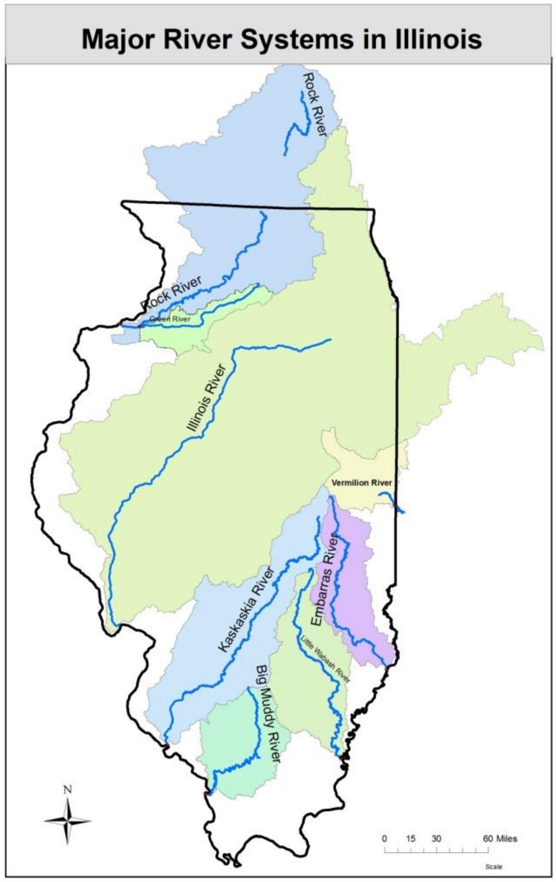

Figure 1. Eight major river systems used in estimating state nutrient loads. Note that gaging stations are up river from the state boundary so that the estimated area is smaller.

The WRDTS load estimates were calculated using software developed and provided by the USGS (https://github.com/USGS-CIDA/WRTDS/wiki). WRDTS estimates are based on regressions with discharge, time and seasonality and the user can specify the relative weightings for each of these factors by changing the value of three variables: windowY for time, windowQ for discharge and WindowS for seasonality. The model developers recommend default values of 10, 2 and 0.5, respectively, for these parameters. Daily load estimates produced by WRDTS with the default weightings were compared to the observed loads on the days when sample concentrations were measured by two different methods. First, linear regression was conducted between observed and model estimated load, with the intercept set to zero. If the slope of the regression line deviated substantially from 1.0, or if the coefficient of determination was less than 0.8, then alternative values for weightings were considered. Secondly, the WRDTS software calculates a flux bias statistic, which estimates the average deviation of the model load from the measured loads. If the flux bias statistic indicated a bias of 10% or greater, we used WRDTS to estimate loads with alternative values of the weighting parameters. The weighting values that produced the load estimates with the lowest flux bias statistic and the greatest correspondence between observed and model estimated daily loads were considered the best estimates.

This analysis was informed by communication with the USGS model developers. For the Illinois River, the seasonality parameter was reduced to 0.25. Sprague et al. (2011) had conducted and published an analysis of Illinois River nitrate-N loads and found a seasonality weighting of 0.25 was appropriate for the Illinois River, which was the value we used. The default weightings produced unusually large flux bias on the Embarras River, and in personal communication Hirsch recommended reducing the discharge weighting from 2.0 to 1.0. This substantially reduced the flux bias and improved the correspondence between the estimated and observed loads. This weighting was also found to reduce the bias and improve the correspondence for the Kaskaskia and the Vermilion rivers for both dissolved reactive P and total P loads, and for estimating dissolved reactive P loads for the Rock River at Joslin and the Green River.

For all eight rivers we calculated and compared annual loads (1980 through 2011) using both interpolation and WRDTS for nitrate-N, dissolved reactive P, and total P, and interpolation alone for total N. For nitrate-N, interpolation and WRDTS gave results that differed by less than 10% for most rivers. The Embarras River was an exception, for which WRDTS produced estimates that were 15 to 20% larger than interpolation. For dissolved reactive P and total P, both methods gave similar loads for the larger rivers, such as the Illinois. For smaller rivers, WRDTS gave larger loads in years with higher flows. This comparison supported our use of interpolation for nitrate-N and total N, and WRDTS for dissolved reactive P and total P.

Average annual riverine water estimated for the entire state, N, and P loads for the two periods of interest, 1980-1996 and 1997-2011 are presented in Table 4 (the 8 major rivers summed and then expanded to represent the entire state). There was a small (1%) increase in water flow from early to the later period, with a small increases in nitrate-N (1.4%) and total N (1.7%) loads. These changes are within the errors of our estimation methods and suggest little change with time. However, total P increased by 9.3%, with most of this increase in dissolved reactive P (20.1%). The David and Gentry (2000) estimate for total P load during 1980-1997 was likely lower due to interpolation being used to estimate loads. Total N and water loads were similar to what David and Gentry (2000) estimated. Point sources were 18.4% of the nitrate-N loads for 1997-2011, 16.3% of total N, and 48% of total P, nearly identical to previous estimates by David and Gentry (2000) for an earlier time period. Nutrient sources that contribute to riverine load for the state are shown as a percent of the total in Figure 2.

The increase in water flow was due to unusually high flows (compared to the long-term average) during 2008-2011 which followed relatively lower flows (compared to the long-term average) during 1997-2007 (Figure 3). Linear regression indicated no significant trend in annual flow for the 1980-2011 time period. Figure 3 includes a Locally Weighted Scatterplot Smooth (LOESS) curve calculated using SAS Ver. 9.2, that can be used to describe the relationship between Y and X without assuming linearity or normality of residuals, and is a robust description of the data pattern (Helsel and Hirsch, 2002). Annual nitrate-N and total P loads had different temporal patterns, with nitrate-N having no trend through time, but total P increasing (Figure 4). A linear regression of annual total P loads with annual water flux and year had an R^2 of 0.97, with both water and year significant at the $p < 0.0001$ level. Annual loads of dissolved reactive P had a similar result, with an R^2 of 0.96. Thus, the increase in annual P flux appears to be related to water flux and possibly other factors such as changing point source inputs or agricultural practices (e.g., fertilizer form, placement, and timing, manure practices, and tillage changes), although these were not evaluated.

For annual loads of nitrate-N, the Illinois, Embarras, Little Wabash, Big Muddy, and Vermilion all declined between 1980-1996 and 1997-2011, whereas the Rock, Green, and Kaskaskia increased, as did the state load (Figure 5). The greatest change was for the Rock, where the load increased 66% between these two time periods (while flow increased 12%). Because we estimated the load for the Rock by subtracting the station at Rockton from the load at Joslin, the resulting load is representative of Illinois only, and there the increase in annual nitrate-N loads was a result of greater losses from Illinois. For total P, all rivers except the Green, Vermilion, and Embarras increased, leading to an overall 10% increase for the state. This analysis of major rivers indicates that for nitrate-N the increase and decrease in riverine loads led to no change in the overall state export, but that there were differences through time within watersheds of Illinois.

We compared Illinois loads to the overall Mississippi River basin (MRB) loads available from the USGS. During the 1997-2011 time period. Illinois contributed about 20% of the nitrate-N load, and 11% of the total P load and 7% of the water flow to the Gulf of Mexico.

Table 4. Water, nitrate-N, total N, dissolved reactive P, and total P loads for Illinois for 1980-1996 and 1997-2011, along with David and Gentry (2000) estimates as a comparison. Point source loads are also shown, as well as point sources as a percent of the recent loads.

	Water	Nitrate-N	Total N	Dissolved reactive P	Total P
	10^{12} ft^3 yr^{-1}	----------------- million lb N or P yr^{-1} -----------------------			
David and Gentry (2000)	1.6		538		31.3
1980-1996	1.70	404	527	15.4	34.0
1997-2011	1.72	410	536	18.5	37.5
Urban runoff		6.0	8.3		1.5
Point sources		75.2	87.3		18.1
Point source percent of 1997-2011 load		18.4	16.3		48
David and Gentry (2000) point source percent of load			16		47

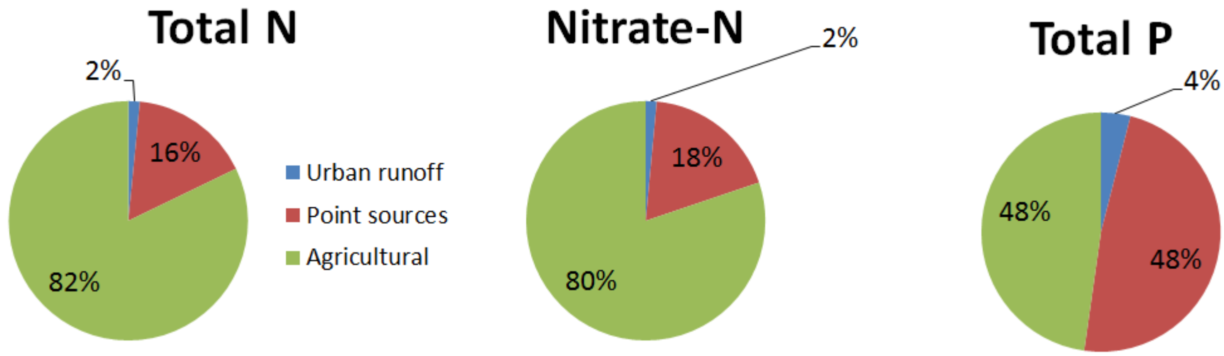

Figure 2. Sources of nutrients in Illinois contributing to riverine nutrient export from the state.

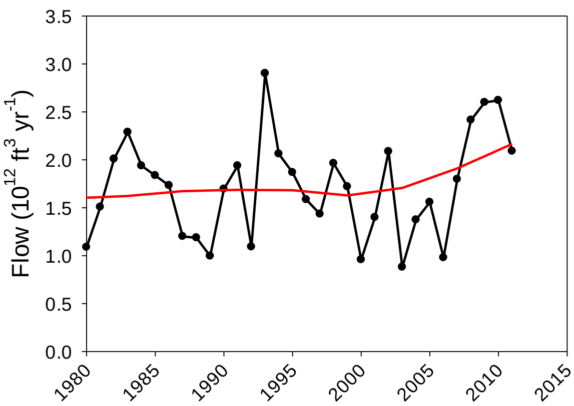

Figure 3. Annual water flows from Illinois for the 1980 through 2011 water years. LOESS trend fit shown in red.

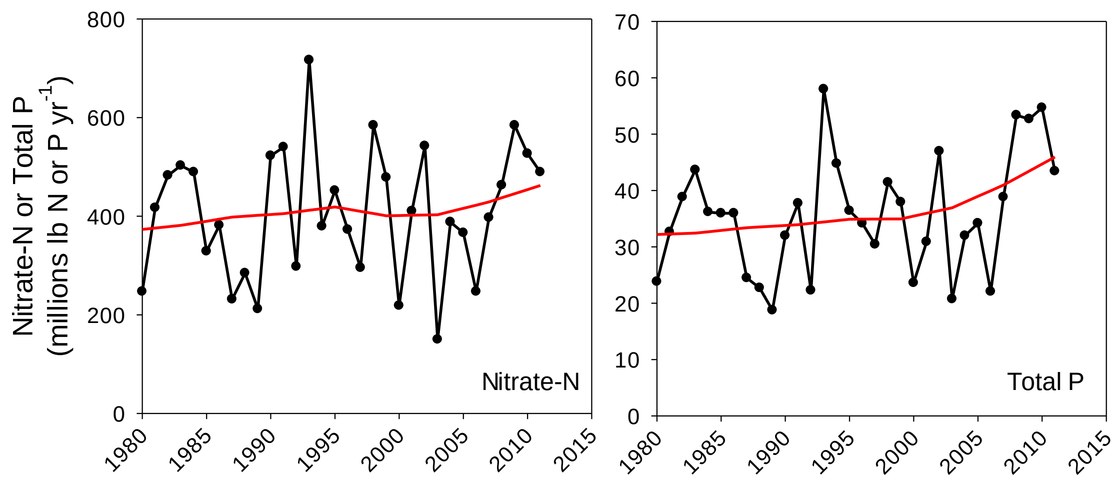

Figure 4. Annual nitrate-N and total P loads from Illinois for the 1980 through 2011 water years. LOESS trend fit shown in red.

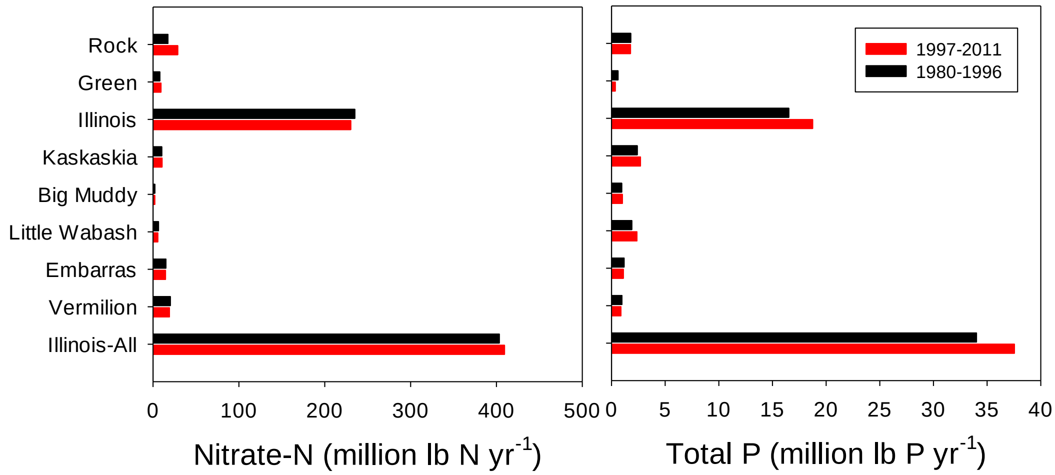

Figure 5. Riverine loads of nitrate-N and total P averaged for 1980-1996 and 1997-2011.

Dissolved reactive P was about half of total P, but is has increased as a percent of total P during the past 10 years (Figure 6). It had been consistently about 45% of total P during the 1980s to late 1990s, but has been greater and more variable since. Declines in particulate P loads are likely related to reduced erosion from adoption of conservation tillage and possibly increased tile drainage, whereas increases in dissolved reactive P could be due to reduced incorporation of P fertilizers (and more intense winter/spring storms), increased population, and more tile drainage.

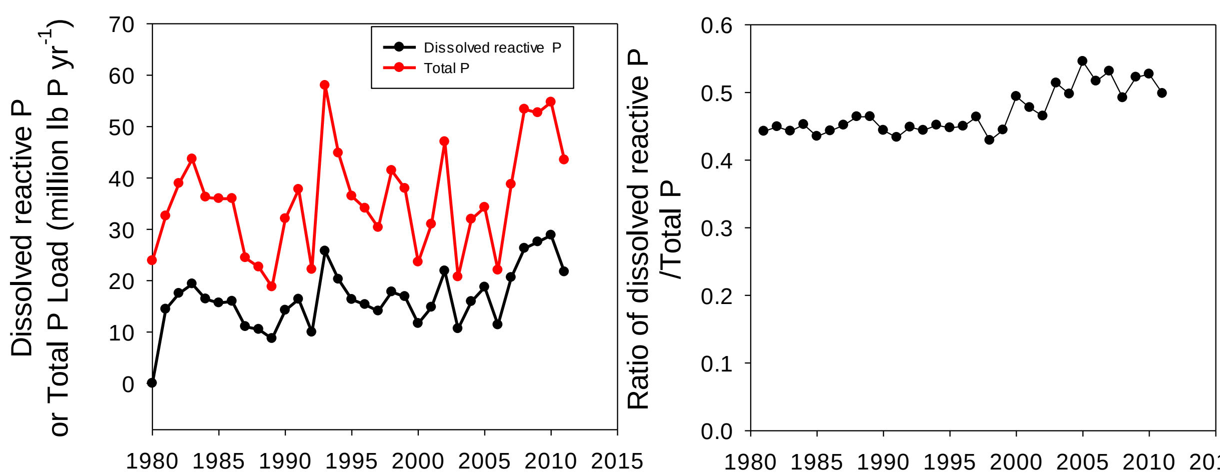

Figure 6. Dissolved reactive P and total P loads by water year from 1980 through 2011, along with the ratio of dissolved reactive P to total P.

Riverine Nutrient Yields

Riverine nutrient loads are influenced by the size of a watershed. Larger watersheds typically produce larger flows and nutrient loads. Another way to examine nutrient losses from a watershed, and to compare watersheds, is to divide the load of a nutrient at the outlet by the area of the watershed, which is called the yield. It allows watersheds of differing sizes to be compared for their nutrient loss per unit of land. Yields of total N and nitrate-N varied greatly across the state, with the tile drained watersheds having much larger yields than the non-tiled southern Illinois watersheds (Figure 7). In addition, some of the watersheds in southern Illinois are not as intensively agricultural. The state average nitrate-N yield was 11.3 lb N/acre/yr averaged for the 1997 to 2011 period, but this varied from 1.4 (Big Muddy) to 23 (Vermilion) lb N/acre/yr. Total P yields were less variable, and averaged 1.1 lb P/acre/yr, with a range of 0.55 to 1.18 lb P/acre/yr. When yields of nitrate-N and total P are viewed by source, the importance of point sources in the Illinois River and non-point in the other rivers is clearly shown (Figure 8).

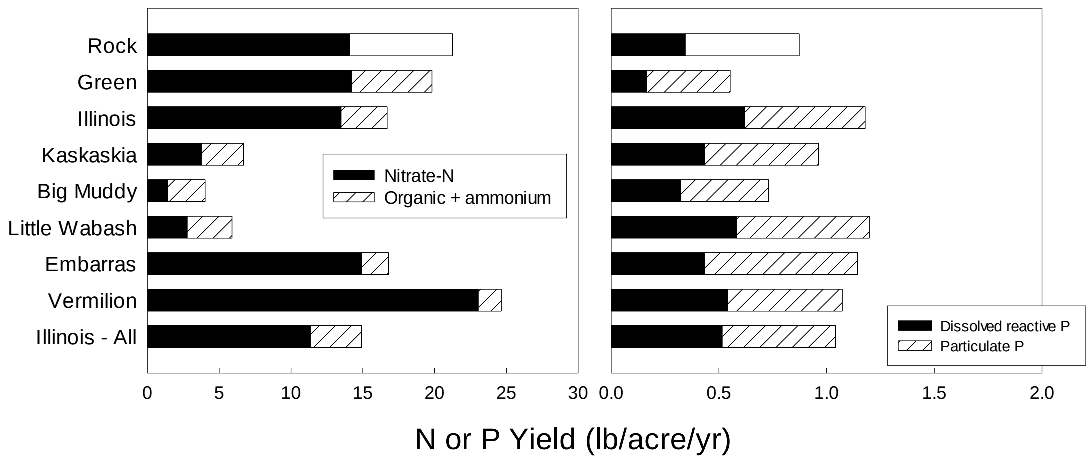

Figure 7. Nitrogen and P yields by watershed and for the state of Illinois as a whole, averaged for the 1997-2011 water years.

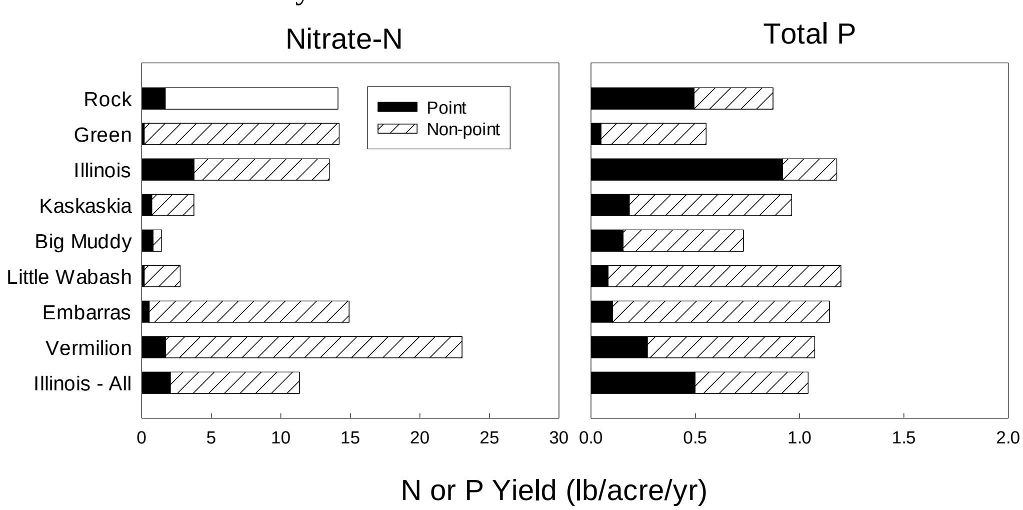

Figure 8. Nitrate-N and total P yields by source averaged for the 1997 through 2011 water years.

Riverine Nutrient Load Goal or Target

To meet a 45% reduction from 1980 to 1996 average riverine loads of nitrate-N and total P, the nitrate-N load target is 222 million lb N yr^{-1}, and total P is 18.7 million lb P yr^{-1}. Given that the 1997 to 2011 loads were greater than 1980-1996, this would require a 46% reduction from those loads for nitrate-N, and 50% for total P. Figure 9 shows loads by river and for the state, and the goal that is to be met. To meet the nitrate-N target, the focus must be on agricultural sources, mostly in northern and central Illinois. For total P, reductions in point sources could meet a large part of the target, but additional P reductions from agriculture throughout the state will probably be needed. The magnitude of the task ahead is shown in Figure 10, that indicates that only during low flow years was the target for nitrate-N met, and for total P only during the 1988 drought (although other dry years were close to the goal). To meet the target consistently will take major reductions from all sources.

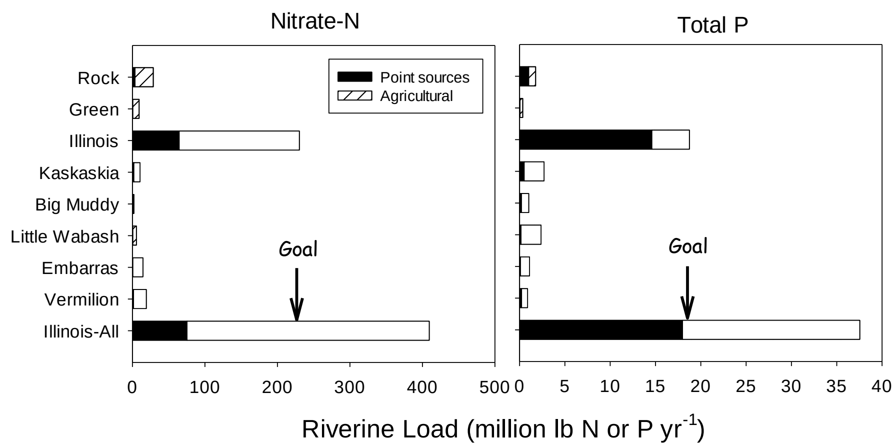

Figure 9. Riverine loads for 1997 through 2011 by source, along with the overall goal that leads to a 45% reduction in nitrate-N and total P loads based on the 1980 to 1996 riverine load average.

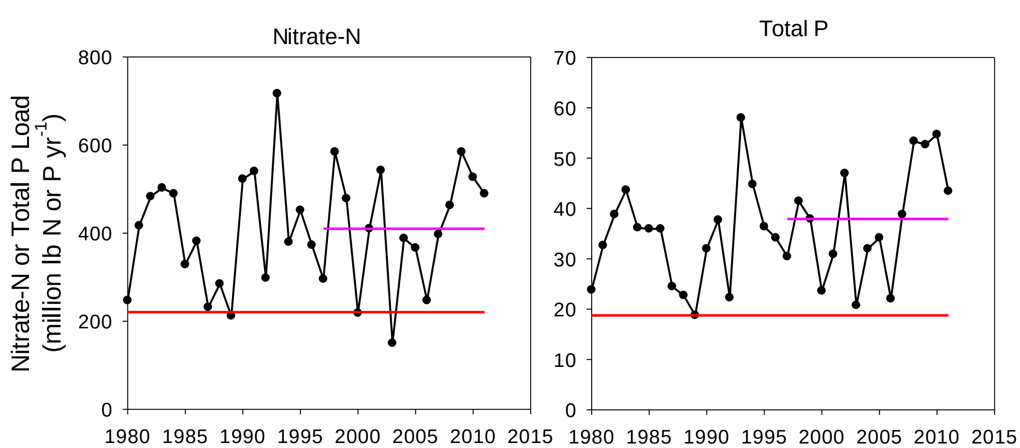

Figure 10. Riverine nitrate-N and total P loads by water year from 1980 through 2011, showing the goal or target load in red, and the average load during the last 15 years in purple.

Critical Watersheds

Methods Used to Facilitate Critical Watershed Identification

To support the determination of critical watersheds in the state, we evaluated nutrient yields at the HUC8 level (Figure 11). There are 50 8-digit HUC's in Illinois that drain into the Mississippi River, and one that drains into Lake Michigan. The HUC8s range in size from 17 to 2,436 sq miles, with an average size of about 1100 sq miles. The HUC8s with a small area in Illinois are actually larger, but they straddle two states and have only a small portion in Illinois. For each HUC8, we looked for available IEPA nutrient data combined with USGS stream flow gauges. In some of the HUC8s, flow and nutrient concentration data were available for a river that drained a large part of the HUC8. In others, the gauged drainage area was much smaller than the overall HUC8. In a few HUC8s, more than one river gauge was used, and in others, the estimate was made by difference between upstream and downstream monitoring sites. For the HUC8s without any available data, averages were taken from the surrounding HUC8s with estimated nutrient yields. For the HUC8s with both USGS stream flow gauges and IEPA nutrient data for nitrate-N and total P for 1997 through 2011, we used these data to estimate annual average nutrient yields in lb of N or P/acre/yr. As we did for the large rivers in the state, linear interpolation was used for the nitrate-N estimates, and WRDTS for total P. For nearly every site, little or no concentration data were available for 2007 through 2008, and those years were not included when concentration data were not available. This monitoring data allowed us to directly calculate an overall nitrate-N and total P yield for 39 of the 50 HUC8s, including 7 by difference between upstream and downstream sites. We also disaggregated point source nitrate-N and total P estimates by HUC8, and subtracted the point source nitrate-N or total P yield from the total nutrient yields to obtain estimates of both point and non-point source nutrient yields for each HUC. This worked well for all HUC8's except for three in northeastern Illinois, where the point source yields were quite high and not reflected in the river data we had available. In those HUC8s, we set the total nutrient yield equal to the point source yield, and assumed non-point sources were zero. Data were not available to allow us to determine the urban non-point source contribution in the three northeastern HUC8s, as the point source loads were so much greater than the stream loads estimated from water quality data. In addition, urban density values were not easily available by HUC8.

Figure 11. HUC8 subbasins in Illinois.

Nitrate-N and Total P Yields by HUC8

Total, point source, and non-point source nitrate-N and total P yields by HUC8 are shown in Figures 12-15. For nitrate-N, greatest yields were found in the tile-drained northern two thirds of Illinois where tile drainage is common and land-use is dominantly row crop agriculture. Statewide nitrate-N yields ranged from 0.7 to 42 lb nitrate-N/acre/yr, with an average of 13 lb nitrate-N/acre/yr. The non-point source yield average was 10 lb nitrate-N/acre/yr, with 27 HUC8s > 10 lb nitrate-N/acre/yr. For point sources, the Chicago and Des Plaines HUC8s had very large nitrate-N yields of 40.9 and 38.3 lb nitrate-N/acre/yr, respectively.

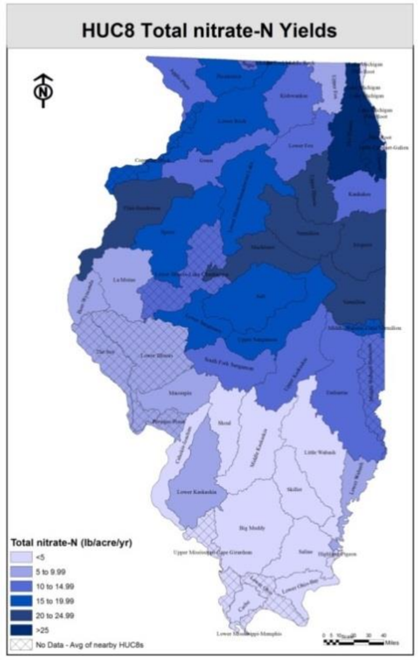

Figure 12. Total nitrate-N yields by HUC8 in Illinois.

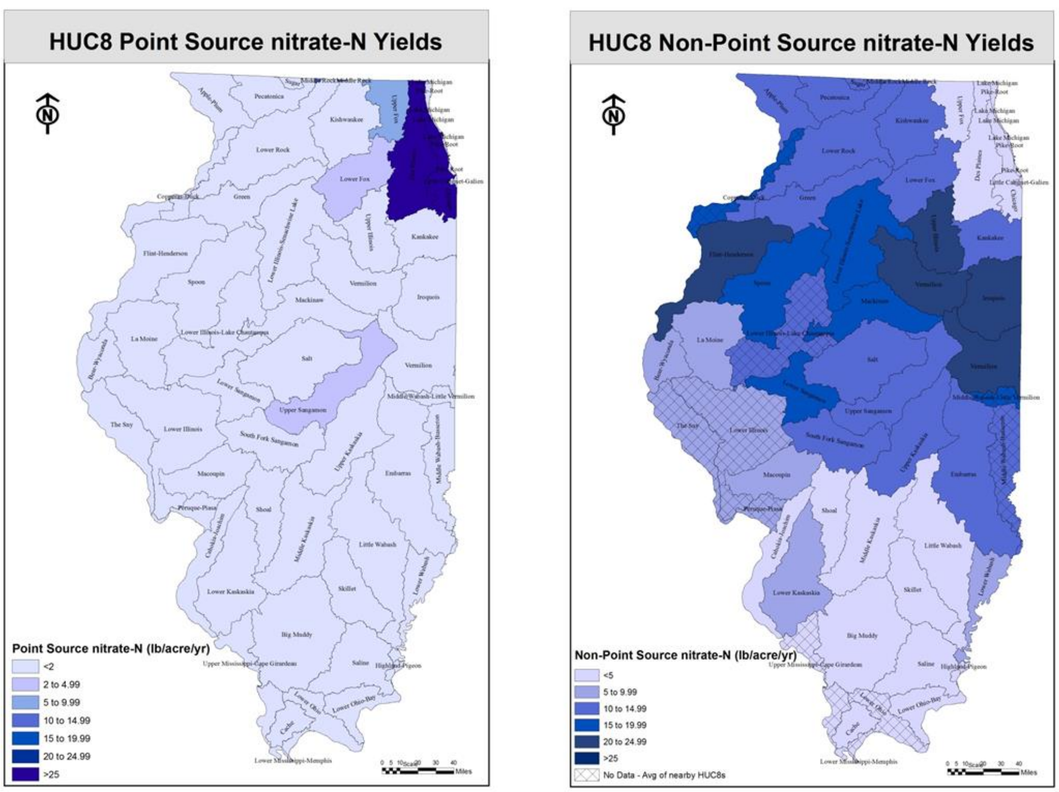

Figure 13. Point and non-point source nitrate-N yields by HUC8 in Illinois.

For total P, the average yield was 1.4 lb P/acre/yr, and ranged from 0.42 to 9.74 lb P/acre/yr across the HUC8s in the state (Figure 14). Non-point source total P yields were typically greater in southern Illinois HUC8s, with the smallest yields in northern Illinois (Figure 15). Point source total P yields were very large in the Chicago area, the Upper Sangamon HUC8 due to the sewage treatment plant discharge in Decatur, and in some HUC8s along the Mississippi River. The Chicago and Des Plaines HUC8 point source P yields were 9.74 and 6.65 lb P/acre/yr.

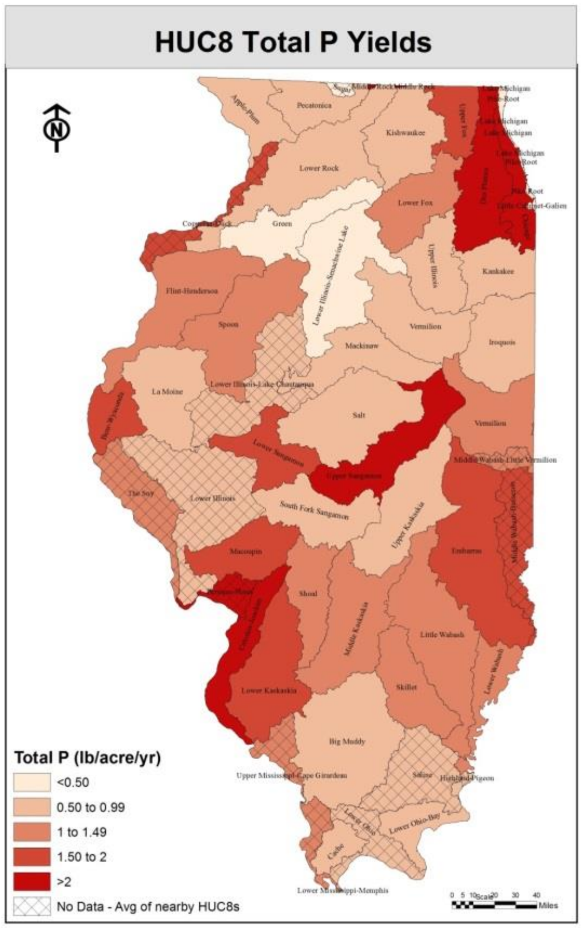

Figure 14. Total P yields by HUC8 in Illinois.

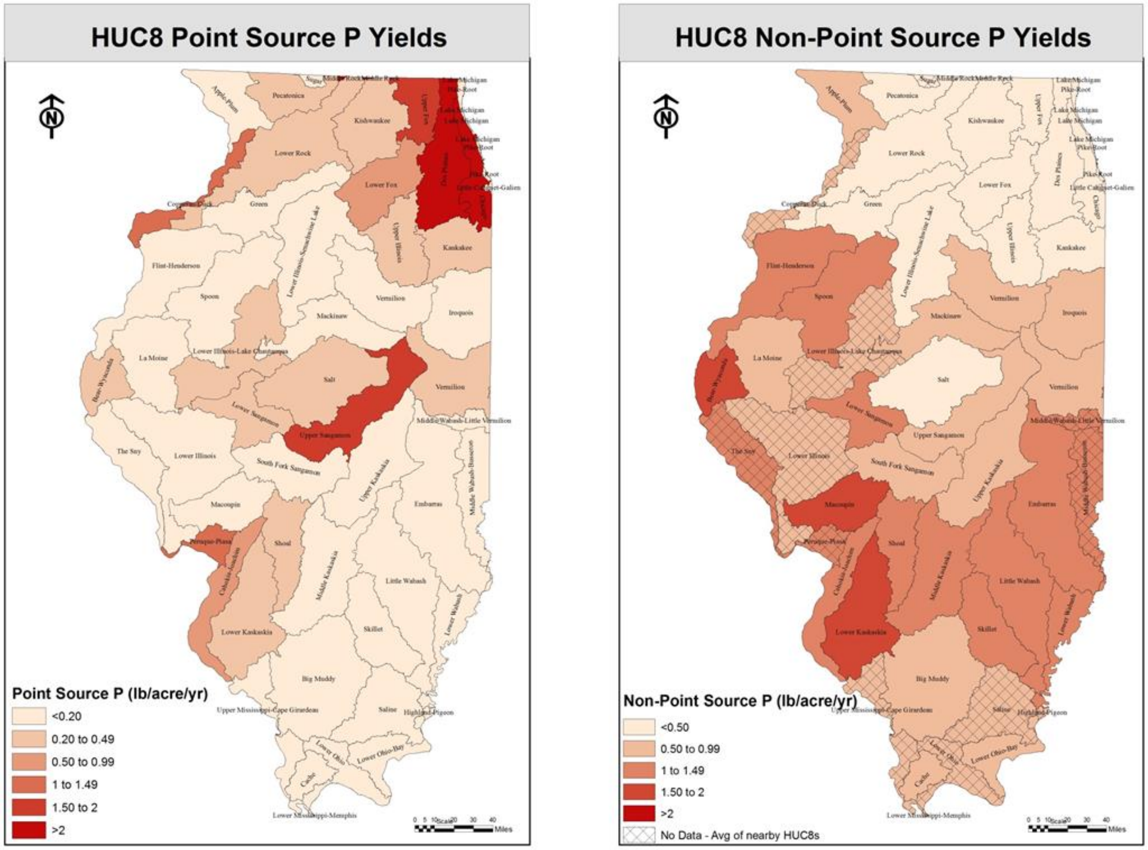

Figure 15. Point and non-point source total P yields by HUC8 in Illinois.

303d/305(b) Impaired Waters in 2012

We conducted an analysis to determine if yields of nitrate-N or total P were related to the miles of impaired streams or acres of lakes reported in the 2012 Integrated Report. Assessed 2012 303(d) listed streams, as well as 305(b) impaired streams and lakes by HUC8 were included if they were listed for dissolved oxygen, total P, nitrate-N, aquatic plants, or aquatic algae (Figure 16). There were 4,070 miles of streams listed on the 303(d) list, 4,346 on the 305(b), and 127,270 acres of lakes on the 305(b) assessment. We located them within each of the HUC8s, and summed the stream miles or acres of lakes by HUC8. Finally, a correlation analysis was conducted to determine if yields of nitrate-N or total P were related to the miles of impaired streams or acres of lakes. There were no strong relationships, with only HUC8 non-point source nitrate-N yield significantly related to 305(b) stream miles ($p=0.002$) and 305(b) lake acres ($p=0.011$). However, this relationship was negative, suggesting that the greater the nitrate-N yield from non-point sources, the fewer impaired stream miles or lake acres were present. Total P yield by HUC8 was not statistically correlated with impaired stream miles or lake acres.

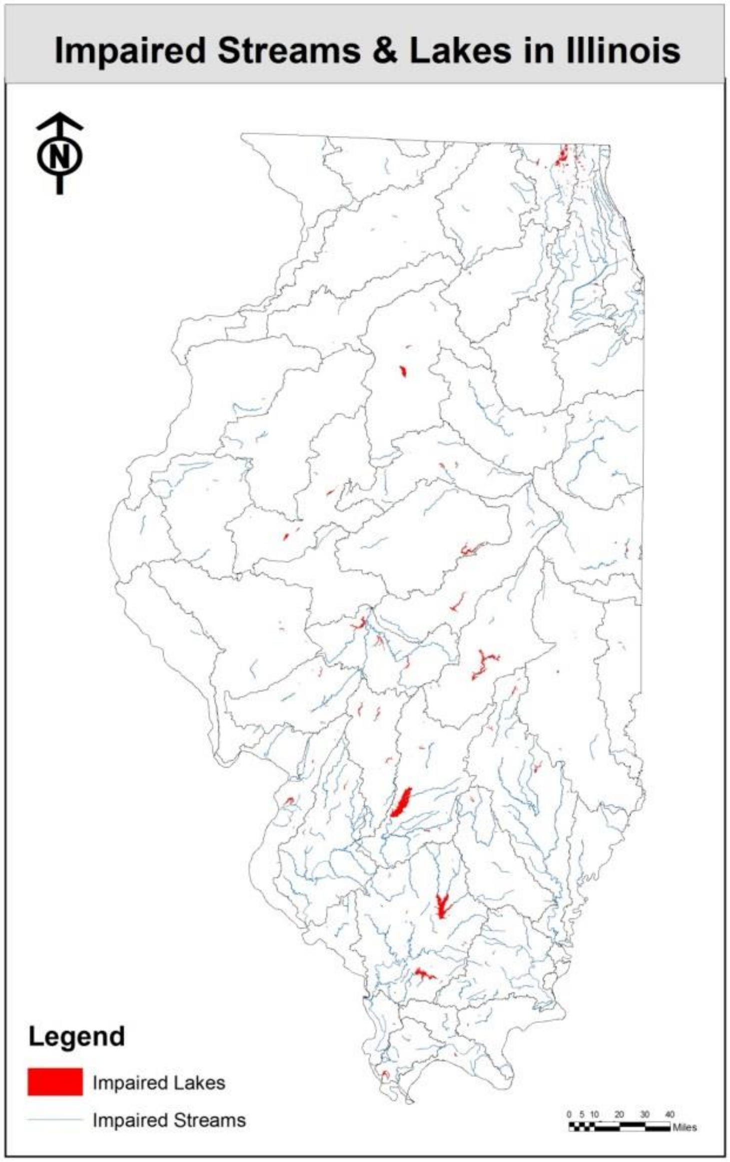

Figure 16. Impaired streams and lakes in Illinois, defined as those found on the 305(b) assessment for total P, dissolved oxygen, nitrate-N, aquatic plants, or aquatic algae.

Agricultural Practices and Nutrient Losses by MLRA in Illinois

To examine agriculture throughout Illinois, we used Major Land Resource Areas (MLRAs) from the USDA Natural Resources Conservation Service (NRCS), last published in 2006 (USDA-NRCS, 2006). MLRAs are geographically associated land resource units based on climate, soils and land use. There are 15 MLRAs in Illinois, but several have only a small area in the state. Therefore, we combined MLRAs into a total of nine for the state, as described in Table 5 and shown in Figure 17.

Information on crop acres planted and yields were obtained from the USDA National Agricultural Statistics Service (NASS) survey data for 2008 through 2012 by county, and then summed by MLRA (Table 6). More than 12 million acres of corn were planted each year in Illinois, and about 9 million acres of soybean. The third crop was wheat, but there was less than 1 million acres planted annually. Using tile drained acres per county from David et al. (2010), we estimated that 9.7 million acres in the state were tile drained, with most in MLRA groupings 2, 4, and 7. We assumed all drained acres were in corn and soybean production. Corn and soybean yields were greatest in northern and central Illinois, and the least in southern Illinois.

Fertilizer N usage per county was obtained from David et al. (2010), averaged for the time period 1997 through 2006. David et al. (2010) used state level fertilizer sales reported each year, with county usage broken out proportionately from USDA NASS Census of Agriculture fertilizer, lime, and soil conditioner expenditure county level data from 1997 and 2002, with other years estimated by interpolation. Because statewide N fertilizer sales were 4% greater during 2008 through 2012 compared to the David et al. (2010) average during 1997-2006, we increased the estimate we made by MLRA by this percentage. To estimate the amount of N fertilizer used per acre of corn, we assumed 100 lb N/acre/yr was used on wheat, and then the remainder was applied to corn (no reduction for non-farm N, which is a small percentage of N sales in Illinois). Manure N was from David et al. (2010), and was adjusted for pastured cattle since manure from this source is not applied to production crops. David et al. (2010) had been designed to look at overall N balances in a region, and the manure numbers included all cattle (both grain-fed and pastured). Cattle on feed data were obtained from USDA NASS by county to adjust the manure values for pastured cattle. All remaining manure N was then assumed to be applied to corn and all was assumed to be plant available. Depending on manure type, only some of the N would likely be available in the first year of application. However, because manure amounts were low in most of Illinois, this assumption has only a small effect on estimated N rates. Row crops were defined here as the sum of corn, soybean, and wheat acres.

Nitrogen application rates were then estimated for corn in rotation with soybean and for continuous corn, by adjusting the total fertilizer N application in combination with the acres of corn and soybean during 2008-2012 (Table 7). We assumed that corn after corn is typically fertilized at a rate 40 lb N/acre/yr greater than corn after soybean (Illinois Agronomy Handbook, 2012). The recommended amount of N fertilizer for both rotations was determined using the Maximum Return to N (MRTN) calculator (Iowa State University, 2013) assuming a ratio of corn price to N fertilizer price of 10:1. For central and northern Illinois MLRAs, the corn/soybean or continuous corn estimated fertilizer + manure rate was about the same or less than the MRTN, suggesting that on average Illinois farmers are applying the recommended N fertilizer rate for corn. For southern Illinois MLRAs, the rate was well above the MRTN. However, two of the MLRAs where this was true (8 and 9) have few corn acres and the estimates are subject to large errors.

To obtain nitrate-N yields per row crop acre, we used non-point source N yields in each HUC8 integrated across each MLRA, determining the average load of nitrate-N in the MLRA. These overall nitrate-N loads were then divided by row crop acres to get the nitrate-N yield per row crop acre (Table 8). We assume no nitrate-N loss from non-row crop agricultural lands in this analysis (e.g., pasture). These values ranged from 3.9 to 7.4 nitrate-N lb/acre/yr in southern Illinois, and 19.6 to 31.3 nitrate-N lb/acre/yr in central and northern Illinois. Tile drainage can explain this pattern, with the exception of MLRA 3, the northwestern corner of Illinois. This is a karst region with high livestock density, and the high nitrate-N yield may be explained by these factors. To develop the nitrate-N yield losses that will be used to determine the reduction in yields due to management practice change, we partitioned the nitrate-N yield per row crop acre into losses from tile drained land and losses from non–tiled land in MLRAs 1, 2, 4, 6, and 7. These are the MLRAs with substantial acres that are tile drained, and also had the larger nitrate-N yields due to this drainage. Nitrate-N yields from tile drained MLRAs ranged from 26 to 43 lb N/acre/yr, whereas yields from non-tiled land ranged from 3.9 to 11.8 lb N/acre/yr, with the exception of MLRA 3, the northwest corner of Illinois discussed earlier, which had a nitrate-N yield of 31 lb N/ac/yr.

Fertilizer P usage per county was from Jacobson et al. (2011), which was a mass balance study that estimated P usage on the basis of every acre in a county and was averaged for the 1997-2006 crop years. We adjusted this P rate by dividing the total P applied to a county by the sum of corn, soybean, wheat, and hay acres (Table 9). Manure P was also from Jacobson et al. (2011), and was adjusted the same as fertilizer P. In addition, as for manure N, cattle on feed data were obtained from USDA NASS by county to adjust the manure values, allowing pastured cattle manure P to be subtracted from overall manure P. Fertilizer P application rates on cropland ranged from 11.0 to 14.9 lb P/acre/yr, with little variation across the MLRAs, and manure between 1.2 and 5.4 lb P/acre/yr. The largest manure P rate was MLRA 3 in northwestern Illinois, where there was a high density of livestock. Total P yields per row crop acre ranged from 0.68 to 2.82 lb P/acre/yr, with greater losses in southern Illinois, and the least in northeastern Illinois.

These data will be basis for applying nutrient reduction practices by MLRA across Illinois.

Table 5. Major Land Resource Areas (MLRAs) in Illinois, showing combinations to be used for analysis (15 combined into 9). Bold MLRAs are the numbers that will be used throughout our analysis.

		Landscape		Climate		
MLRA	**Description**	**Elevation m (ft)**	**Local Relief m (ft)**	**Precipitation mm (inches)**	**Annual Temperature °C (°F)**	**Freeze Free Days**
95B	Southern Wisconsin and Northern Illinois Drift Plain	200 to 300 (660 to 980)	8 (25)	760 to 965 (30 to 38)	6 to 9 (43 to 48)	170
97	Southwestern Michigan Fruit and Truck Crop Belt	200 to 305 (600 to 1000)	2 to 5 (5 to 15)	890 to 1,015 (35 to 40)	8 to 11 (47 to 52)	200
98	Southern Michigan and Northern Indiana Drift Plain	175 to 335 (570 to 1,100)	15 (5)	735 to 1,015 (29 to 40)	7 to 10 (44 to 50)	175
110	Northern Illinois and Indiana Heavy Till Plain	200 (650)	3 to 8 (10 to 25)	785 to 1,015 (31 to 40)	7 to 11 (42 to 52)	185
105	Northern Mississippi Valley Loess Hills	200 to 400 (660 to 1,310)	3 to 6 (10 to 20)	760 to 965 (30 to 38)	6 to 10 (42 to 50)	175
108A	Illinois and Iowa Deep Loess and Drift, Eastern Part	200 to 300 (660 to 985)	1 to 3 (3 to 10)	890 to 1,090 (35 to 43)	8 to 12 (47 to 54)	195
108B	Illinois and Iowa Deep Loess and Drift, East-Central Part	200 to 300 (660 to 985)	1 to 3 (3 to 10)	840 to 990 (33 to 39)	8 to 12 (47 to 54)	185
113	Central Claypan Areas	200 (660)	1.5 to 3 (5 to 10)	915 to 1,170 (36 to 46)	11 to 14 (51 to 57)	205
115A	Central Mississippi Valley Wooded Slopes, Eastern Part	100 to 310 (320 to 1,020)	3 to 15 (10 to 50)	1,015 to 1,195 (40 to 47)	11 to 14 (53 to 57)	210
114B	Southern Illinois and Indiana Thin Loess and Till Plain, Western Part	105 to 365 (350 to 1,190)	3 to 15 (10 to 50)	940 to 1,170 (37 to 46)	11 to 14 (52 to 56	210
115C	Central Mississippi Valley Wooded Slopes, Northern Part	130 to 270 (420 to 885)	3 to 6 (10 to 20)	865 to 1,015 (34 to 40)	9 to 13 (48 to 55)	200
120A	Kentucky and Indiana Sandstone and Shale Hills and Valleys, Southern Part	105 to 290 (345 to 950)	Varies widely	1,145 to 1,370 (45 to 54)	13 to 14 (55 to 58)	210
115B	Central Mississippi Valley Wooded Slopes, Western Part	100 to 310 (320 to 1,020)	3 to 15 (10 to 50)	965 to 1,220 (38 to 48)	12 to 14 (53 to 57)	205
131A	Southern Mississippi River Alluvium	0 to 100 (0 to 330	Max 5 (15)	1,170 to 1,525 (46 to 60)	14 to 21 (56 to 69)	210 (North)
134	Southern Mississippi Valley Loess	25 to 185 (80 to 600)	3 to 6 (10 to 20)	1,195 to 1,525 (47 to 60)	14 to 20 (57 to 68)	215 (North)

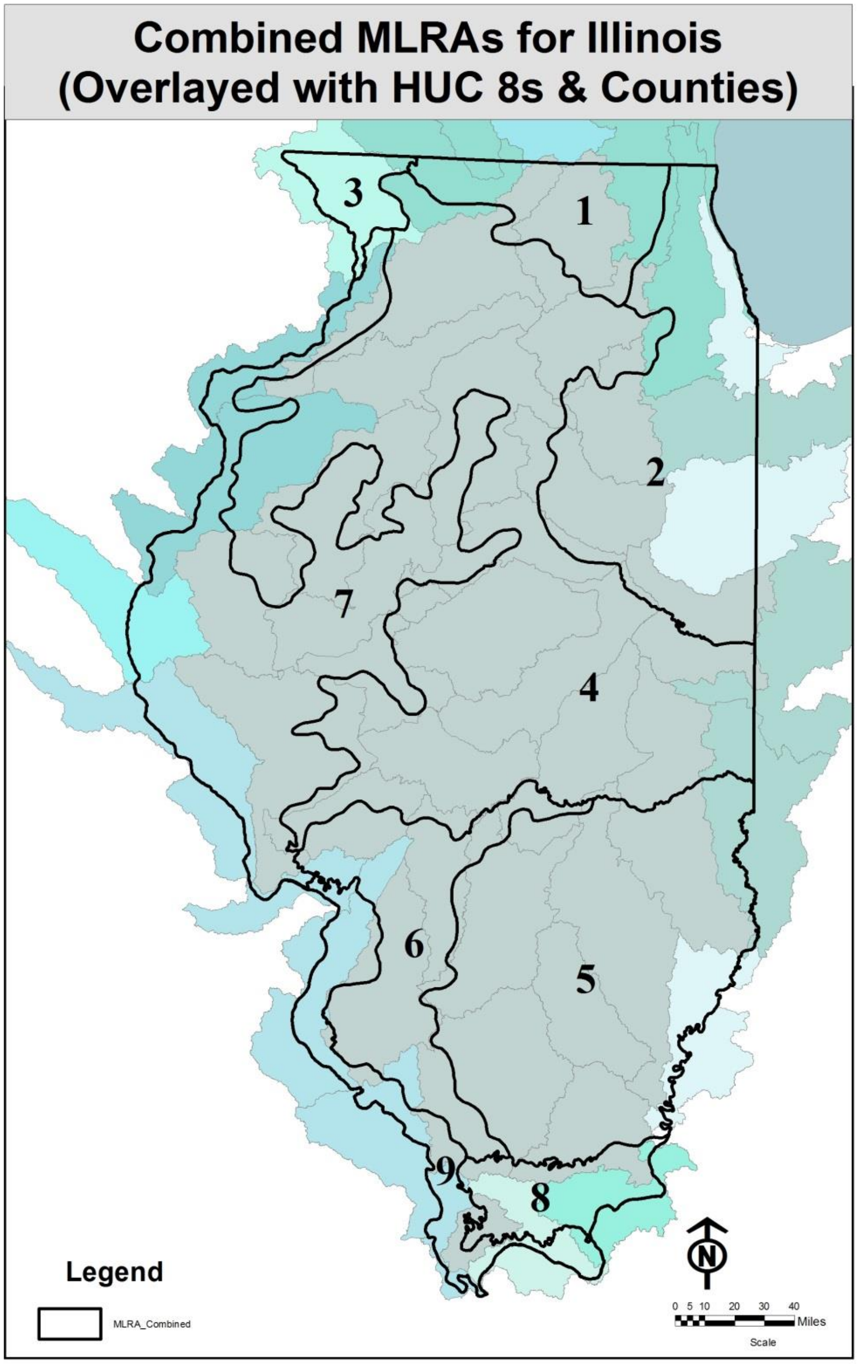

Figure 17. Combined MLRAs shown overlaying HUC8s in Illinois.

Table 6. Summary of agricultural management by MLRA in Illinois, averaged for 2008-2012, showing corn, soybean, and wheat acres, drained acres (with percent of crop acres), and corn and soybean yields.

Combined MLRA	Description	Corn (1000 acres)	Soybean (1000 acres)	Wheat (1000 acres)	Drained acres (% of crop acres)	Corn yield (bushels/acre)	Soybean yield (bushels/acre)
MLRA 1	Northern Illinois drift plain	516	224	20	288 (39)	161	48
MLRA 2	Northeastern Illinois heavy till plain	1,532	1,112	42	2,064 (78)	150	39
MLRA 3	Northern Mississippi Valley	164	52	2	2 (10)	160	50
MLRA 4	Deep loess and drift	5,580	3,343	76	5,438 (61)	164	52
MLRA 5	Claypan	1,610	1,992	353	310 (9)	128	39
MLRA 6	Thin loess and till	664	690	161	227 (17)	130	42
MLRA 7	Central Mississippi Valley, Northern Part	2,059	1,289	74	1,285 (38)	155	49
MLRA 8	Sandstone and shale hills and valleys	84	115	11	50 (25)	103	33
MLRA 9	Central Mississippi Valley, Western Part	204	315	78	24 (5)	125	39
Sum		12,412	9,132	817	9,706 (43)		

Table 7. Estimated corn fertilizer + manure N to corn in a corn/soybean rotation and continuous corn along with the recommended rate using the Maximum Return to N approach by MLRA in Illinois.

Combined MLRA	Description	Estimated CS fertilizer + manure (lb/acre/yr)	MRTN (10 to 1) CS (lb N/acre/yr)	Estimated CC fertilizer + manure (lb/acre/yr)	MRTN (10 to 1) CC (lb N/acre/yr)
MLRA 1	Northern Illinois drift plain	156	146	196	199
MLRA 2	Northeastern Illinois heavy till plain	151	155	190	197
MLRA 3	Northern Mississippi Valley	146	146	184	199
MLRA 4	Deep loess and drift	147	155	185	197
MLRA 5	Claypan	181	171	227	189
MLRA 6	Thin loess and till	157	171	198	189
MLRA 7	Central Mississippi Valley, Northern Part	156	163	197	194
MLRA 8	Sandstone and shale hills and valleys	202	171	254	189
MLRA 9	Central Mississippi Valley, Western Part	188	171	237	189

Table 8. Tile drained cropland acres and nitrate-N yields per row crop acre, along with yields divided into tile drained and non-tiled drained land by MLRA in Illinois.

Combined MLRA	Description	Drained cropland (1000 acres)	Nitrate-N yield per row crop acre (lb N/acre/yr)	Nitrate-N yield per tile drained acre (lb N/acre/yr)	Nitrate-N yield from non-tiled land (lb N/acre/yr)
MLRA 1	Northern Illinois drift plain	289	20.4	43	6.6
MLRA 2	Northeastern Illinois heavy till plain	2,064	25.0	29	10.8
MLRA 3	Northern Mississippi Valley	21	31.3		31.3
MLRA 4	Deep loess and drift	5,438	19.6	26	9.9
MLRA 5	Claypan	310	6.6		6.6
MLRA 6	Thin loess and till	227	7.4	30	3.5
MLRA 7	Central Mississippi Valley, Northern Part	1,285	24.5	46	11.8
MLRA 8	Sandstone and shale hills and valleys	50	3.9		3.9
MLRA 9	Central Mississippi Valley, Western Part	24	4.0		4.0

Table 9. Phosphorus fertilizer and manure inputs, row crop acres, and total P yields per row crop acre by MLRA in Illinois.

Combined MLRA	Description	Estimated fertilizer (lb P/acre/yr)	Estimated manure (lb P/acre/yr)	Row crops (1000 acres)	Total P yield per row crop acre (lb P/acre/yr)
MLRA 1	Northern Illinois drift plain	14.9	3.9	760	0.71
MLRA 2	Northeastern Illinois heavy till plain	13.4	1.3	2,686	0.68
MLRA 3	Northern Mississippi Valley	13.4	5.4	218	1.72
MLRA 4	Deep loess and drift	13.6	2.3	9,000	0.96
MLRA 5	Claypan	11.7	2.4	3,954	1.74
MLRA 6	Thin loess and till	11.3	2.5	1,515	2.09
MLRA 7	Central Mississippi Valley, Northern Part	13.6	3.4	3,421	1.45
MLRA 8	Sandstone and shale hills and valleys	11.3	1.3	210	2.82
MLRA 9	Central Mississippi Valley, Western Part	11.0	1.6	597	2.82
Sum				22,362	

Point Source Reductions and Cost Estimates

We estimated potential reductions in point sources for both total P and nitrate-N. It is important to keep in mind that point source nitrate-N was only estimated at 18% of the load for the state, whereas total P was 48% of the state load. For total P, we estimated various reductions that would occur with the major point sources reducing P discharge concentrations to either 1.0 or 0.3 mg P L^{-1}, as well as only the top 20, 30, or 50 majors (Table 10). From these results it is clear that most of the potential reduction is from reducing the current discharge concentrations of 2.5 to 3.0 mg P L^{-1} to 1.0 mg P L^{-1}, with less of a reduction from reducing concentrations from 1.0 to 0.3 mg P L^{-1}. In addition, reductions in total P of just the top 20 majors provide about 58% of the estimated reduction. In Illinois, a few very large plants produce a large portion of the point source total P and reductions from these facilities lead to large reductions in the statewide estimated total P loads.

After reviewing other states' nutrient reduction plans and various reports on possible reductions in nutrients from sewage effluent, we made estimates of the costs and reductions that would occur from all majors reducing their effluent concentrations to 10 mg nitrate-N L^{-1}, and 1.0 mg total P L^{-1} . For point source reduction costs, we used the following sources:

USEPA Municipal Nutrient Removal Technologies Reference Document (USEPA, 2008)
Nutrient Removal Study North Shore Sanitary District, Illinois (Donohue & Associates, 2010)
Utah Statewide Nutrient Removal Cost Impact Study (CH2MHILL, 2010)
Minnesota nutrient reduction strategy (Minnesota, 2013)
Iowa nutrient reduction strategy (Iowa, 2013)
Water Policy Working Paper #2005-011 (Jiang et al., 2005)
Cost/Benefit Study of the Impacts of Potential Nutrient Controls for Colorado Point Source Discharges (Camp Dresser & McKee Inc., 2012)

USEPA (2008) provided annualized costs per lb of N or P reduced for eight plants in the United States and Canada. We used the median values of N and P costs for these eight plants. For the three North Shore Plants in Illinois, we were able to calculate a number of estimates for each plant depending on the technology that might be used, and took the median value for our overall estimate. There were twenty plants in the Utah study where they had calculated a net present value cost that we adjusted to an annualized $ per lb reduced, and again we took the median value from this study. The 2005 construction costs in the Jiang et al. (2005) study were escalated to 2013 dollars using the construction cost index. Iowa and Minnesota costs came from their 2013 nutrient reduction plans, with estimates made of the lb of N or P reduced since these values were not given. We assumed current total P concentrations were 2.78 mg of total P L^{-1} and nitrate-N concentrations of 16.8 mg N L^{-1} in all plants that would be affected. These values were the statewide averages from our evaluation of point source nutrient concentrations in Illinois. We also used the statewide average from a Colorado study that we adjusted to annualized costs (from the reported net present values). Finally, we took the mean of each of these costs estimates from the seven different studies (range $2.42 to $33.23 per lb for total P, and $1.34 to $5.67 per lb for total N), for overall cost estimates of $13.71 per lb of total P reduced (to a 1 mg total P L^{-1} standard), and $3.30 for each lb of nitrate-N (to a 10 mg of nitrate-N L^{-1} standard). When applied to all major point sources in the state, 14 and 8.3 million lb of nitrate-N and total P, respectively, would be reduced at annualized costs of $46 million per year for nitrate-N, and $114 million for total P. There would be substantial first year construction costs, but these were annualized over a 20-year period at an interest rate of 4.5%.

These costs are averages and the cost would vary greatly dependent on a plant's current configuration and treatment processes. Plants that could effectively do biological nutrient removal would likely have much lower costs than those that would do chemical treatment alone. In addition, larger plants can typically reduce nutrients at a cheaper per lb cost compared to small plants. As plant size approaches 1 MGD or less, costs increase greatly. As a final comment, point source reductions to lower standards (0.5 or 0.1 mg P L^{-1}, 3 mg nitrate-N L^{-1}) would cost much more per lb reduced. We were not able to find enough information on these costs to make good estimates, but the literature suggests as much as 10 times greater costs per lb.

Table 10. Point source P estimates at two limits and number of majors reduced, along with the % of target reduction (the target is a reduction of 18.8 million lb P).

Point Source Limit (mg P L^{-1})	Million lb of P reduced	% of target (18.8 million lb P)
All majors to 1 mg P L^{-1}	8.3	44
Top 20 majors to 1.0	4.8	26
Top 30 majors to 1.0	5.4	29
Top 50 majors to 1.0	6.1	32
All majors to 0.3 mg P L^{-1}	13.1	70
Top 20 majors to 0.3	8.0	42
Top 30 majors to 0.3	8.9	47
Top 50 majors to 0.3	9.9	52

Non-point Source Nitrate-N Reduction Practices

We considered a full range of practices that could be applied in Illinois to reduce nitrate-N losses from agricultural fields. We utilized the Iowa nutrient reduction strategy literature review (Iowa, 2013) and the Lake Bloomington Study (David et al., 2008) as the core of understanding of what might work well in Illinois, and then made modifications based on Illinois conditions. Practices are divided into three groups: in-field, edge-of-field, and land use change (Table 11). We took the per acre costs presented in Appendix 1 and estimated what the overall cost would be in $/lb nitrate-N removed if the practice or scenario were fully implemented in the state. The results at this stage of the analysis cannot be added together, because one practice may affect the removal effectiveness in another.

In-Field Practices

Our analysis of fertilizer use suggested that in most areas of the state producers apply N fertilizer at rates similar to what is recommended through the MRTN calculator. However, it is likely that not all producers are following this guideline, and so we assumed 10% are well above the MRTN and reducing their N rate to MRTN would result in 10% reduction in nitrate-N losses per acre (reduction % from Iowa, 2013). When applied to 10% of fertilized corn acres in the state, this would reduce the overall nitrate-N load by 2.3 million lb yr^{-1}, or 0.6% of the baseline. This isn't a large reduction, but the cost is negative, meaning that producers would save money.

The impact of nitrification inhibitors was estimated at 4.3 million lb yr^{-1} of nitrate-N losses reduced. Assumptions included a 10% per acre reduction in loss (from Iowa, 2013) by using an inhibitor for fall applied N compared to no inhibitor, and that 50% of the N in the northern two thirds of Illinois (MLRAs 1, 2, 4, 6, and 7) is applied in the fall, and that currently 50% of that N includes an inhibitor. These assumptions come from analysis of fertilizer sales information, surveys, and discussions with industry representatives. Cost is $2.33/lb N removed.

Two estimates were made for changing fertilizer timing. The first was that no fall N was applied to tile drained acres in the state. Based on results from Clover (2005) and Gentry et al. (2014), we used a 20% reduction in nitrate-N losses in central Illinois, and a 15% reduction in northern Illinois. Central Illinois has warmer temperatures and typically greatly winter and early spring tile flow, leading to potentially greater losses from fall applied N. Iowa (2013) estimated a 6% reduction using data from both Iowa as well as surrounding states, but their temperatures are lower and they have less precipitation in their tile-drained region compared to Illinois. These assumptions led to a 26 million lb nitrate-N yr^{-1} reduction, or 6.4% of the baseline. We also estimated a split application of 50% fall, and 50% spring for a given field. Because there are no measurements for this N system, we assumed it would be 50% as effective in reducing nitrate-N losses as moving all fall N to spring. Therefore, the estimated reduction in loads was 13 million lb nitrate-N yr^{-1}. We did make an estimate for a fertilizer system that includes three applications [i.e., some in fall with inhibitor (40%), at planting (10% as a carrier for herbicides or as a started fertilizer), and then side-dressed (50% in mid-June], although there are no data available on what the nitrate-N response in tile drains would be. The Clover (2005) data did show a 20% reduction in nitrate-N losses from tile-drains when spring and side dressed were compared. Therefore, given the three-way split, we have included an estimate that is similar to the reduction for the fall to spring change in timing. Costs for timing changes ranged from $3.17 to $6.22/lb N removed.

The other major in-field management change is the use of cover crops. There have been many studies of the effectiveness of cover crops, but fewer on tiled-drained fields. Iowa (2013) calculated from the literature a 31% reduction in nitrate-N losses from a rye cover crop, and 28% from oat. We therefore assumed a 30% reduction in nitrate-N losses using a generic, grass cover crop. When applied to all tile-drained acres in Illinois, a cover crop led to the largest nitrate-N reduction of any practice, 84 million lb N yr^{-1}, or 20.5% of the baseline. When applied to all non-tiled acres, the reduction was 32 million lb N yr^{-1}. The costs for cover crops were quite different between tile-drained ($3.21/lb N removed) and non-drained lands ($10.62/lb N removed), because nitrate-N loss per acre is so much greater on the tile-drained lands, reducing costs per lb.

This calculation doesn't mean that cover crops take up more N on tile-drained fields, only that the leaching losses are larger and therefore the reduction is greater, reducing the cost per lb.

Edge-of-field Practices

We estimated the effectiveness of three edge-of-field practices: bioreactors, wetlands, and buffers. Bioreactors are trenches filled with wood chips that are located on the edge of fields and intercept tile flow. Iowa (2013) estimated their effectiveness at a 43% reduction in nitrate-N loss from a field. We used a value of 40%, and assumed that as much as 50% of all tile-drained land could receive a bioreactor, reducing nitrate-N loads by 56 million lb N yr^{-1} (13.6% of baseline). Bioreactors have a large upfront cost, but when estimated on a $/lb N removed, were the cheapest practice we evaluated at $1.38/lb N.

For constructed wetlands, we assumed a 40% effectiveness, whereas Iowa (2013) assumed a 52% reduction in nitrate-N losses. Our constructed wetlands are different than Iowa's in that ours are typically put at the end of individual tile lines at a wetland/field ratio of 5% (wetland area to drainage area) and are small in size (0.5 to 2 acres). In Iowa the wetlands are typically many acres in size fed by drainage areas of 1000 to 2000 acres. They intercept tile mains that we don't have here in Illinois. Kovacic et al. (2000) conducted the most complete constructed wetland study in Illinois, and measured a 37% reduction in tile nitrate-N loads to the river. When they included seepage reductions, the overall estimate increased to 45%. Considering the increase in intense precipitation events in winter and spring we now have, we conservatively estimated that constructed wetlands in Illinois would be 40% effective in reducing nitrate-N loads. We assumed that due to landscape and other limitations, no more than 25% of tile-drained land could accommodate a wetland. This would lead to a 28 million lb N yr^{-1} (6.8% of baseline) reduction at a cost of $5.06/lb N removed.

Buffers along agricultural ditches and streams can reduce nitrate-N losses by plant uptake and denitrification, but only for the water that seeps through them. In tile-drained landscapes much of the drainage water bypasses buffers, and estimating the water that does flow through them is difficult. In non-tile drained regions of the state, they can be effective at reducing nitrate-N losses to streams, although the current loads in the streams are much lower. To estimate the potential reduction from planting grass riparian buffers along streams, we first conducted a GIS analysis to identify the miles of streams that currently have buffers (defined as vegetation other than a row crop within 100 feet of the stream) and those that do not. Approximately 64% of the agricultural stream miles in the state were determined to not have a buffer, and therefore nitrate-N loads could be reduced if a buffer were planted. Iowa (2013) used studies conducted in the state and a complex analysis to determine the amount of water and nitrate-N that would pass through buffers. This analysis was beyond the data we had available for Illinois. To estimate nitrate-N removal by buffers, we used the ratio of total P to nitrate-N removed in Iowa (2013) with the total P estimate we made for Illinois (see below). By assuming buffers were installed on all agricultural streams currently without buffers, the estimate is that nitrate-N would be reduced by 36 million lb N yr^{-1} statewide, or 8.7% of baseline at a cost of $1.63/lb N removed. This is a crude estimate, but we believe it is the correct magnitude although it is likely to vary throughout the state due to soils and lateral flow path differences.

One edge-of-field practice we did not include in our cost estimates is drainage water management (DWM). This practice involves raising the outlet of the tile system with a control structure to as little as 6 inches below the soil surface during periods of the year when the field doesn't need to be worked, such as winter and early spring (Frankenberger et al., 2006; Skaggs et al., 2012). This practice works best on flat fields (< 0.5% slope) with new patterned tile systems, but can be retrofitted on existing systems. Research has shown that reductions in nitrate-N loss are nearly the same as the reduction in water that occurs from raising the tile outlet, and can be as much as 82% (Skaggs et al., 2012). In Illinois Cooke and Verma (2012) found nitrate-N reductions with DWM of 37 to 79%, similar to reductions measured by Woli et al. (2010), also for a field in central Illinois. Limitations of the results to date is that most of the studies have been on small fields, often just a few acres, and a poor understanding of the fate of the held back water and nitrate. When the tile outlet is lowered, nearly all studies have shown that most of the held back water doesn't drain out. Therefore, the question is where did it move to? If the water and nitrate-N move through lateral seepage due to the tile being raised to a nearby ditch or tile system, then the effectiveness at the watershed scale would be greatly reduced. A recent study by Sunohara et al. (2014) showed that DWM could increase seepage both to groundwater and laterally, and that this seepage could limit the effectiveness. However these authors also indicated more research was needed because their research with DWM was only during the growing season, rather than during winter and early spring, when we really expect this practice to be utilized. Given these uncertainties with this practice, we did not include it any scenarios.

Land-use Change

Two estimates were made for land-use changes. The first was putting cropland that was converted to row crops from pasture/hay from 1987 through 2007 by MLRA (1.1 million acres from NASS Census of Agriculture data) into perennial crops. This estimate is essentially returning the last land converted into row crops back into perennial crops. We estimated a 90% reduction in nitrate-N losses from this conversion, based on results from Iowa (2013) and from recent work with biofuels on the University of Illinois South Farms (Smith et al., 2013). The estimated nitrate-N reduction would be 10 million lb N yr^{-1}, a 2.6% reduction from baseline at a cost of $9.34/lb N removed.

As an additional estimate, we calculated what the nitrate-N reduction would be if 10% of corn/soybean tile-drained land were converted to perennials, again with a 90% per acre reduction. This 1 million acre change would lead to a 25 million lb N yr^{-1} reduction from the state, of 6.1% of baseline at a cost of $3.18/lb N removed. This cost per lb N removed much less than the other land-use change described above, because the land is 100% tile-drained, leading to much larger reduction per acre.

Table 11. Example statewide results for nitrate-N reductions by practice/scenario with shading to represent in-field, edge-of-field, land use change, and point source practices or scenarios.

Practice/Scenario	Nitrate-N reduction per acre (%)	Nitrate-N reduced (million lb N)	Nitrate-N reduction from baseline (%)	Cost ($/lb N removed)
Reducing N rate from background to the MRTN (10% of acres)	10	2.3	0.6	-4.25
Nitrification inhibitor with all fall applied fertilizer on tile-drained corn acres	10	4.3	1.0	2.33
Split (50%) fall and spring (50%) on tile-drained corn acres	7.5 to 10	13	3.1	6.22
Fall to spring on tile-drained corn acres	15 to 20	26	6.4	3.17
Split 40% fall, 10% pre-plant, 50% side dress	15 to 20	26	6.4	
Cover crops on all corn/soybean tile-drained acres	30	84	20.5	3.21
Cover crops on all corn/soybean non-tiled acres	30	33	7.9	11.02
Bioreactors on 50% of tile-drained land	40	56	13.6	1.38
Wetlands on 25% of tile-drained land	40	28	6.8	5.06
Buffers on all applicable crop land (reduction only for water that interacts with active area)	90	36	8.7	1.63
Perennial/energy crops equal to pasture/hay acreage from 1987	90	10	2.6	9.34
Perennial/energy crops on 10% of tile-drained land	90	25	6.1	3.18
Point source reduction to 10 mg nitrate-N/L		14	3.4	3.30

Non-point Source Total P Reduction Practices

Total P Losses and Soil Erosion Estimates

Phosphorus tends to be strongly adsorbed to soil particles, and consequently non-point source total P losses from agriculture tend to be associated with surface runoff and soil erosion. Under certain conditions, leaching and subsurface flow can be significant pathways of dissolved P losses, which we will not address because it is difficult to estimate and thought to be a minor source on average. For the statewide assessment, we obtained the IDOA estimates of cropland erosion for approximately 50,000 points in the state, based on the tillage transect survey conducted in the spring of 2011 by soil and water conservation districts across the state. This survey collects information on residue cover, crop planted, slope and other factors in order to estimate long term average sheet and rill erosion rates from each survey point using the Universal soil loss equation (USLE) and the Revised USLE (RUSLE). USLE was developed in the 1950s. RUSLE, released in the 1990s, is based on more extensive data than USLE, and often produces lower erosion estimates, especially for steeper slopes, due to a modification in the slope steepness factor. We considered the RUSLE erosion estimate to be the more accurate estimate of cropland sheet and rill erosion, but we also used the USLE estimate in conjunction with a compatible sediment delivery formula to estimate total P losses at the watershed scale. Neither method estimates erosion or total P loads from ephemeral gullies, large gullies, or stream channel erosion.

Average sheet and rill erosion rate (ton/ac) estimated with RUSLE within each of the nine modified MLRAs was highly correlated with the riverine non-point source total P yields leaving the MLRAs (Figure 18). The non-point source total P yields are calculated from the riverine total P loads minus the point source inputs and divided by the total land area in the MLRA.

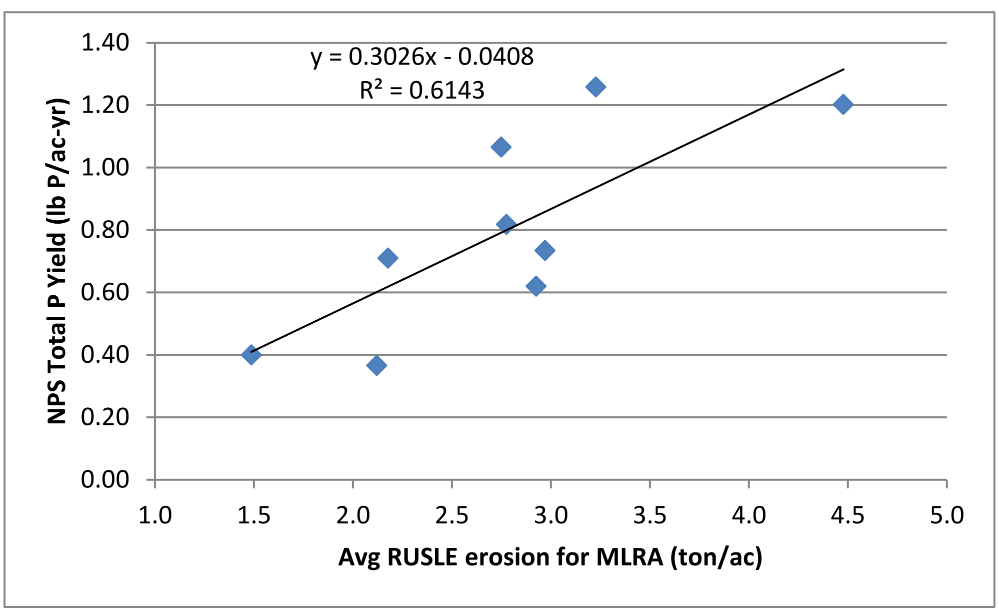

Figure 18. Non-point source total P yield (1997-2011) from the nine modified MLRAs plotted as a function of average cropland RUSLE erosion rates within the MLRAs.

Not all of the riverine non-point source total P leaving the MLRA originates from cropland erosion, and not all soil eroded in a given year reaches the watershed outlet. As surface runoff travels from cropland to larger rivers, some of the eroded soil is deposited in lower adjacent fields, buffers and grass waterways as well as flood plains, stream beds and rivers. The fraction of eroded soil that arrives at a watershed outlet has been estimated by using sediment delivery ratios (SDR). An early and widely used equation for estimating SDR in conjunction with USLE estimates of erosion is

$$SDR = 0.42*(Area)^{-0.125} \qquad \text{Equation [1]}$$

where Area = watershed area in square miles.

Actual SDR will depend on more factors than area (such as slope, rainfall characteristics and vegetation), but limitations of time and resources do not allow us to evaluate these other factors. For the purpose of calculating rough approximations, the above equation was used to estimate SDR for each modified MLRA by using the area of the MLRA. This equation was developed when the USLE was used to estimate erosion, and therefore we used it only in conjunction with USLE estimates of erosion. We used the RUSLE estimates of erosion to calculate the proportion of fields that were eroding at rates greater than soil loss tolerance (T), and to develop estimates of total P load under different nutrient reduction scenarios described below.

Based on observed sediment and total P loads in Illinois from small watersheds (Russell, 2013) ranging from (11 to 67 square miles), we used an average of 1.5 lb of total P load per ton of sediment load. The 1997-2011 average annual non-point source total P loads draining the MLRAs were highly correlated with the total P loads estimated from using the average USLE erosion rates times the corresponding cropland area (including hay), SDR and 1.5 lb P per ton of sediment (Figure 19). The 1.5 lb total P per ton of sediment includes an unknown fraction of dissolved reactive P (DRP) that may have originated from a variety of sources, including desorption from cropland soils that were not eroded (discussed below), desorption from deposited sediments, leaching and subsurface transport of P, or other non-point source P sources.

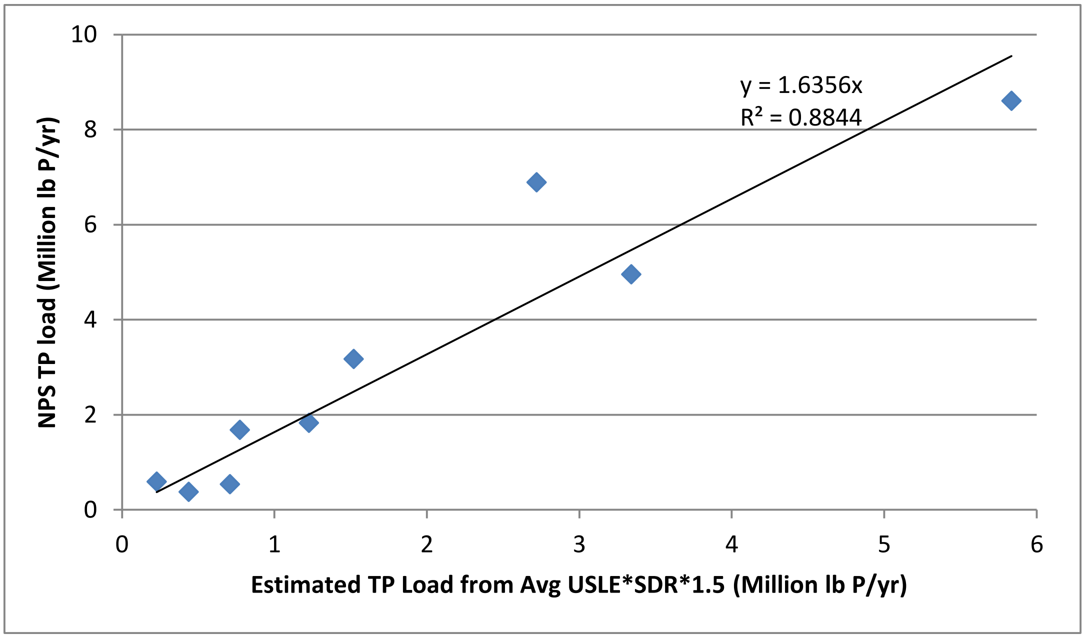

Figure 19. Annual average non-point source total P loads from the modified MLRAs plotted as a function of loads estimated from average USLE erosion rates, cropland area, SDR and assuming 1.5 lb of total P per ton of sediment load.

The high correlation in Figure 19 is partly a result of the variation in size of the MLRAs. The highest loads come from the largest MLRAs. However, the correlation between total P yield and soil erosion rates (Figure 18) is also a factor. If we accept the estimates of USLE and SDR, the slope of the line (1.64) indicates that the non-point source total P loads carried by the rivers draining the MLRAs are 64% more than estimated from cropland erosion. In other words, approximately 60% of the non-point source total P appears to be associated with cropland sheet and rill erosion. The other 40% could conceivably come from ephemeral gully erosion, stream bank erosion, leaching of dissolved P, desorption of soil P to runoff, other non-point source sources and/or estimation inaccuracy.

There is considerable uncertainty about these estimates of total P associated with erosion. For an alternative estimate, we ignored the SDR and used the average RUSLE estimates of erosion and found strong correlation with the non-point source total P load from the MLRAs (Figure 20). In this case, the slope of 0.32 which implies an average SDR of 0.32, if we assume all other non-point sources of total P were negligible. In comparison, the SDRs calculated by Equation (1) ranged from 0.12 to 0.18 for the different MLRAs. A portion of the total P entrained with eroded sediment is DRP desorbed from non-eroded soil during runoff events. Sharpley et al. (2003) estimated that approximately 20% of total P in runoff from cropland is desorbed DRP. Controlling erosion by reducing tillage is not likely to reduce this movement of P, and may actually increase it. Thus, in estimating the impacts of erosion reduction measures on total P loads, we assume that 80% of the estimated total P associated with erosion is attached to eroded soil and can be reduced by erosion control measures. We apply this assumption only to the total P estimate based on the RUSLE estimates of erosion. For our total P load estimate based on the

USLE, we assume the DRP desorbed from soils is a portion of the 40% of the observed non-point source total P load that was in excess of our estimate based on the USLE.

Cropland Erosion Estimates

According to the data collected in the IDOA tillage transect survey, the statewide average RUSLE erosion rate from all cropland (including hay) was 2.4 ton/ac in 2011. On cultivated cropland (corn, soybean and wheat) the RUSLE average was 2.6 ton/ac and the USLE average was 3.6 ton/ac. This latter value is similar to the 2007 USDA-NRI estimate of 3.9 ton/ac of sheet and rill erosion from cultivated cropland in Illinois estimated using USLE. The USDA-NRI continues to use the USLE estimates to allow consistent comparisons to earlier NRI erosion estimates.

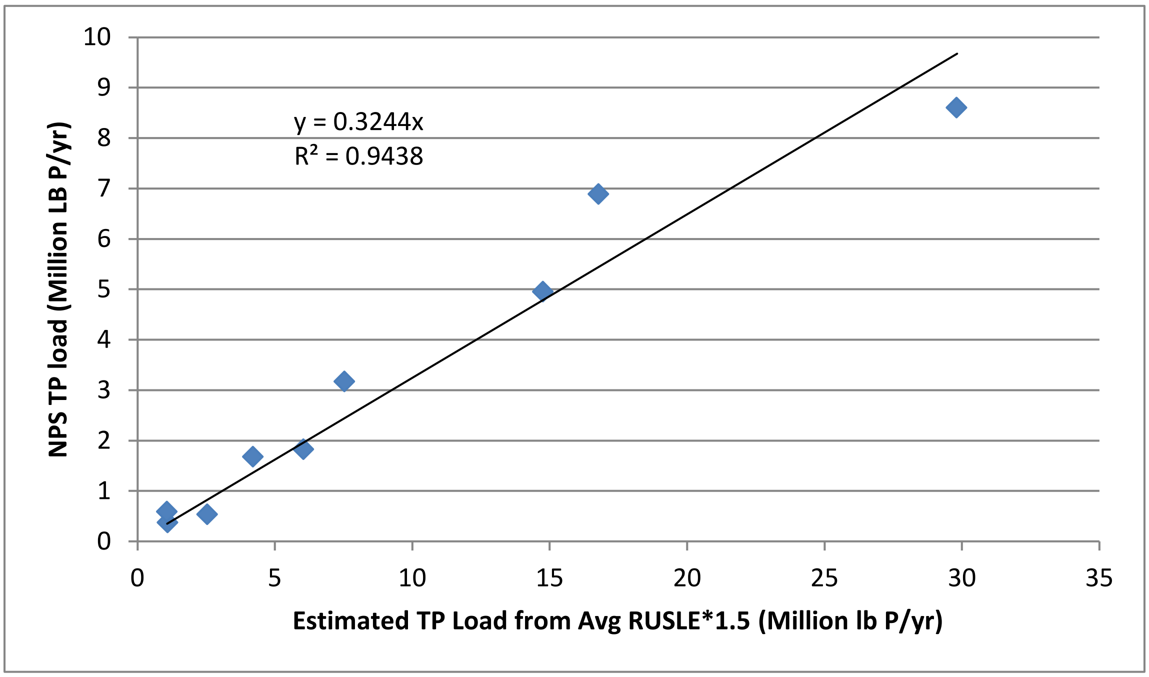

Figure 20. Average annual total P loads from the modified MLRAs plotted as a function of average RUSLE erosion estimates from all cropland in the MLRAs (including hay) times 1.5 lb P per ton of sediment.

Federal and state policies have been enacted to discourage landowners from allowing soil erosion on their cropland to exceed soil loss tolerance (T), which is considered the maximum rate of erosion that does not damage the productive potential of the soil. Values of T vary from 1 to 5 ton/ac/yr depending on soil characteristics. According to the IDOA analysis of the 2011 tillage transect data, 15.8% of Illinois cropland was eroding at rates in excess of T (IDOA, 2011). In their analysis, IDOA excluded erosion estimates of zero, assuming these to be the result of erroneous or incomplete data entry. When we included these zero estimates, we found that 15.4% of the sampled cropland was eroding in excess of T (Figure 21). Many of these zero values included cropland in hay or CRP which would have low erosion rates. Furthermore, there is a likelihood that incomplete or incorrect data entry also resulted in some erosion estimates that were erroneously high. Eliminating only erosion estimates of zero may produce an upward bias in aggregate erosion statistics, although that bias appears to be relatively small.

Our estimates of reducing non-point source total P loads by changing in-field management practices will focus on practices that could reduce erosion rates to T or less on the acres that appear to be eroding greater than T. It is interesting to note that the percentage of cropland acres having RUSLE estimated erosion rates greater than T has increased from 13.5% in 1997 (Figure 21). The highest average erosion rates and the largest percentage of cropland with erosion rates in excess of T were in the southern portion of the state (Table 12), where rainfall erosivity is higher and slopes tend to be steeper than other parts of the state.

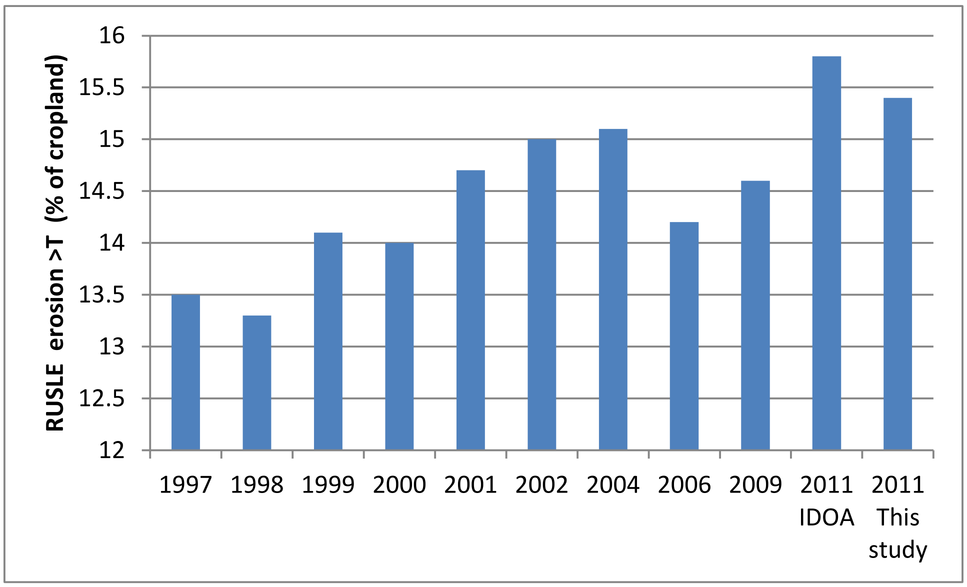

Figure 21. Percentage of Illinois cropland with RUSLE erosion estimates in excess of soil loss tolerance (T); IDOA results for 2011 eliminated observations with erosion rates of zero while our analysis included those values (Data from IDOA, 2011).

Table 12. Extent of cropland area with RUSLE erosion estimates in excess of T, the average amount of erosion in excess of T on these acres, and estimated riverine load of total P associated with erosion in excess of T.

	Cropland with RUSLE Erosion >T		Avg. RUSLE	Est. Riverine total P Load associated with erosion>T	
MLRA			Erosion>T	USLE*SDR*1.5	RUSLE*0.32*1.5*0.8
	(%)	(1000 Acres)	(ton/ac)	(Million lb total P/yr)	
1	9	76	2.7	0.15	0.08
2	7	178	2.2	0.13	0.15
3	21	51	3.8	0.18	0.08
4	9	805	3.2	1.07	0.98
5	22	911	3.4	0.70	1.19
6	27	418	3.8	0.52	0.61
7	15	542	4.0	1.01	0.83
8	18	44	7.3	0.09	0.12
9	27	171	7.0	0.34	0.46
Total		3,196		4.2	4.5

Approximately 3.2 million cropland acres in the state have RUSLE erosion estimates in excess of T. If erosion on these fields were reduced to T, we estimate that the total P leaving the state in its rivers might be reduced between 4.2 and 4.5 million lb P per year, depending on the method used to estimate the relationship between cropland erosion and riverine P loads. Although the highest erosion rates in excess of T are in MLRAs 8 and 9, these MLRAs are relatively small and don't contribute a large portion of the statewide total P loads from cropland with erosion greater than T. MLRAs 4, 5 and 7 contribute a larger portion of this total P load because of their size and high average rate of erosion.

The sites with high erosion tend to have higher than average slopes, and 1.8 million acres are in annual cropland with conventional tillage (Table 13). If these acres were converted to some form of reduced or conservation tillage that would reduce erosion by 50% on average, we estimate a reduction in total P load on the order of 1.8 to 2.8 million lb P per year, at a negative cost of -$16.60/lb P removed (Table 14), assuming no reduction in crop yields.

Table 13. Area of corn, soybeans and wheat in various tillage systems with RUSLE erosion estimates greater than T, and estimated reduction in total P loads from converting the conventional tilled area to a combination of reduced, mulch or no-till resulting in a 50% reduction in erosion.

	Acres of corn, soy & wheat in various tillage systems and with RUSLE erosion estimates greater than T				Estimated reduction in total P load from converting conventional tillage to reduced, mulch or no-till assuming an avg. 50% erosion reduction	
MLRA	Conventional	Reduced	Mulch	No-till	USLE est.	RUSLE est.
	(1000 acres)				(Million lb P/yr)	
1	50	14	10	6	0.08	0.07
2	105	52	21	6	0.07	0.12
3	9	16	26	9	0.02	0.02
4	419	249	88	63	0.39	0.64
5	634	175	89	113	0.45	0.87
6	242	87	40	67	0.24	0.37
7	254	202	78	68	0.34	0.47
8	30	9	5	30	0.04	0.07
9	86	36	21	54	0.12	0.21
State total	1,829	841	377	414	1.8	2.8

Among the acres that are eroding greater than T, there are approximately 1.6 million acres of corn, soybeans and wheat that are already in some form of reduced tillage (reduced, mulch or no-till in Table 13). If these acres included winter cover crops in their rotation, causing a 50% reduction in erosion, we estimate that total P loads would be reduced between 1.9 and 2.3 million lb P per year. Alternatively, if these acres were converted to perennial crops, such as for biofuels, hay or CRP, resulting in a 90% reduction in estimated soil erosion, we estimate that total P loads may be reduced on the order of 3.5 to 6.5 million lb P per year (at a cost of $40.40/lb P for the lower estimate), depending on the method of estimation used.

If winter cover crops were planted on all 21.5 million acres in corn and soybeans, we estimate an average total P loss reduction of 30% on these acres which translates to a statewide load reduction in the neighborhood of 4.8 to 6.1 million lb P per year at a cost of $130.40/lb P removed at the lower estimate. The 30% average reduction assumed for the 21.5 million acres is less than the 50% reduction assumed for the sites eroding greater than T because there is less average erosion and total P loss to reduce. However, the aggregate load reduction is greater because of the larger area considered (21.5 million acres compared to 1.6 million acres).

If 1.1 million acres of current corn-soybean that had been in hay or pasture in 1987 were converted to perennial hay or energy crops, assuming a 90% reduction in erosion from the average erosion rates from corn-soybean in each MLRA, we estimate the statewide reduction in total P load would be approximately 0.9 – 1.1 million lb P per year depending on the method used. For the lower estimate this would cost $102.30/lb P removed.

If 10% of corn-soybean acres on tile drained land were converted to perennial hay or energy crops, assuming a 50% reduction in P loss per acre, the statewide reduction would be approximately 0.3 million lb P per year as a cost of $250.07 per lb P removed.

For all scenarios described above, our assumptions of 30, 50% or 90% reductions in erosion are rough approximations. More precise estimations will require more detailed assessments of the characteristics of the current landscapes and the cropping systems that would replace the current practices. Additionally the impact of ephemeral gulley erosion, associated total P load and the potential of grassed waterways to reduce this source of sediment and total P is highly uncertain. The Illinois office of USDA-NRCS is in the process of producing estimates of ephemeral gully erosion that are expected to be available in the coming months (personal communication from Mr. Kerry Goodrich, State Resource Conservationist, USDA NRCS).

It should also be noted that according to the USDA NRI, average USLE soil erosion rates on cropland in Illinois (including hay) declined from 6.2 ton/ac in 1982 to 4 ton/ac in 1997. Between 1997 and 2007, average erosion rates have declined only slightly to 3.8 ton/ac. Applying our analysis to the NRI data would suggest that the decline in USLE erosion estimates between 1982 and 1997 would have been accompanied by a decline in total P loads of about 14.6 million lb P/yr, but we see no such decline in the state aggregate total P loads (including point sources) in the seven major river basins (Figure 22). We do not have reliable historical data on point source loads, but it is possible that a decline in non-point source total P loads due to erosion control may have been partially offset by an increase in point source inputs. Non-point source total P reductions may have also been offset by legacy effects in which P in stream bed sediments was desorbed as non-point source inputs declined. Almost all of the increase in total P loads in recent years was due to DRP rather than particulate P loads, which may implicate urban point sources as well as leaching from cropland, and surface runoff of unincorporated P fertilizers. Furthermore, the years after 2007 included some record setting rainfall and flooding events, which increased erosion and total P losses more than is reflected in the USLE estimates, which are calculated based on long-term average annual rainfall erosivity values from prior decades.

Trends in total P loads are often difficult to detect because of large year to year variations in river discharge. USGS has developed a method of flow normalization that attempts to eliminate the influence of year to year variations in flow on concentration and constituent flux. Using this technique to examine individual river basins reveals that the flow normalized total P concentrations and loads in the Rock, Green and Embarras Rivers declined between 30 and 60% between 1985 and 2011. For the Illinois, Kaskaskia, Little Wabash and Big Muddy Rivers, flow normalized concentrations and fluxes increased between about 4 and 20%. For the Vermilion River at Danville, flow normalized concentration declined 30% but flow normalized flux increased only 1%. Almost all the decline in the Rock River occurs above Rockton, which is the Wisconsin portion of the basin. When the flow normalized flux at Rockton is subtracted from that at Joslin, and added to the other fluxes, the state total increased by 10% from 1985 to 2011. Further research is needed to understand this variation on patterns across the state and why there was no decline in total P loads despite the large reductions in USLE erosion estimates of the 1980s and early 1990s.

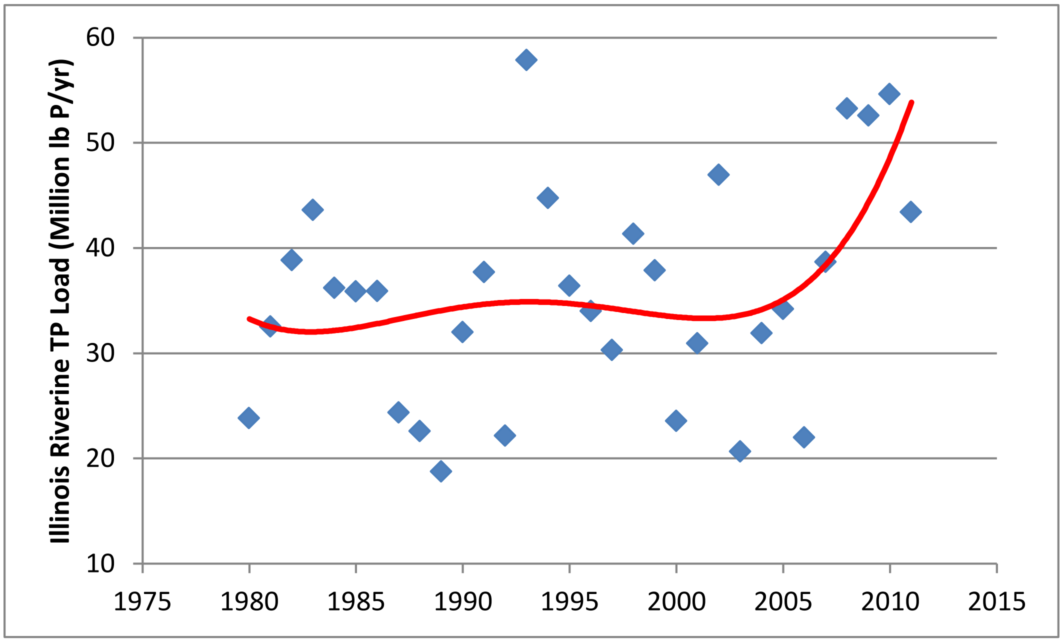

Figure 22. Estimated annual riverine total P loads leaving Illinois based on observed stream flows, periodic concentration measurements and the WRDTS method of load calculation. (Blue diamonds indicate loads of individual years and the red line is a polynomial best fit trend line intended to illustrate unusually high P loads after 2007, and not intended to forecast future trends.)

Soil Test Phosphorus

The vast majority of the P in soil is in organic forms that are not immediately available for crop uptake. A variety of soil tests have been developed to estimate the relatively small portion of total soil P that is readily available to crops. This is generally referred to as soil test P (STP) or specifically to Olsen, Bray, or Mehlich forms of P depending on the specific laboratory procedures used. Since STP is usually a small fraction of the total soil P, normal variations in STP have little influence on the magnitude of P transported with eroded sediment. On the other hand, many studies have shown a strong linear relationship between STP and dissolved reactive P (DRP) in surface runoff (Sharpley et al., 2003). Sharpley et al. (2003) also estimated that about 20% of total P loss in runoff from cropland occurred in the form of DRP as influenced by STP. For the purpose of estimating the influence of STP levels on total P loss in Illinois rivers, we assumed that 20% of the riverine non-point source total P loads from the MLRAs was from desorption of soil P to DRP in surface runoff.

Fernandez et al. (2012) measured STP in 547 Illinois corn fields prior to harvest in September and October of the 2007 and 2008. They reported that 59% of fields sampled were above the level requiring additional P fertilizer to achieve maximum crop yields, and on average, these fields could achieve maximum crop production for 6 years without additional P fertilizer applications. This would reduce the STP and reduce the DRP in runoff from these fields. We estimated a reduction in DRP loss in runoff from cropland that might occur if the average STP values reported by Fernandez et al. (2012) were reduced to the STP maintenance levels (30, 33 and 35 ppm P for the high, medium and low P supplying regions, respectively). This corresponds to a 50% reduction in STP in the high P supplying region, a 37% reduction in the medium P supplying region and a 12% reduction in the low P supplying region. If DRP in runoff is a linear function of STP concentration as indicated by Sharpley et al. (2003), reducing STP would lead to a proportional reduction in DRP loss from cropland. Assuming that this DRP contribution is 20% of the average observed non-point source total P loads for each MLRA, we calculated a reduction of DRP loss for each MLRA based on the proportion of the MLRA in the high, medium and low P supplying regions, and assuming the average STP values reported by Fernandez et al. (2012) were representative of these regions. The total reduction for the state was 1.9 million lb P per year, at a negative cost of -$48.750/lb P reduced.

The STP maintenance levels mentioned above are considered appropriate for fields that are likely to include wheat, oats or alfalfa in the rotation. According to the Agronomy Handbook, STP levels of 20 ppm P can produce maximum yields in a corn-soybean rotation. Assuming STP values from Fernandez et al. (2012) largely represent fields in a corn-soybean rotation, reducing average STP values to 20 ppm P, and following the same procedures as described above, we estimate a reduction in DRP in runoff of 3.2 million lb P/yr.

Edge-of-field Practices

To estimate the potential reduction from planting grass riparian buffers along streams, we first conducted a GIS analysis to identify the miles of streams that currently have buffers and those that do not. Approximately 64% of the stream miles in the state were determined to not have a buffer, and therefore P loads could be reduced if buffers were established on these streams. We assumed that an average of 70% of the non-point source total P load from each MLRA originated as subsurface drainage and surface runoff, and riparian buffers could reduce the P in surface runoff but not subsurface drainage. For non-tile drained land, we assumed that new riparian buffers 35 feet wide would reduce total P loads from cropland without buffers by 50%. In tile drained land, there would be considerably less surface runoff interacting with the buffer, so we assumed a 25% reduction in total P loads from adding 35 foot wide buffers in tile drained areas. From these assumptions, we estimate that planting 35 foot wide riparian buffers on all streams lacking any buffer would reduce total P loss by 4.8 million lb P per year at a cost of $11.97/lb P reduced. Measured total P reductions by perennial buffers have been highly variable. Iowa (2013) used an average reduction of 58%, but reported a range from -10% (i.e., an increase in total P load) to 98% reduction. The estimated reductions we assumed were based on professional judgment informed by empirical results published in relevant literature.

The literature on wetland removal of total P reports highly variable results (e.g., Kovacic et al., 2000). Unlike nitrate-N that can be removed from the aquatic system by conversion to gaseous forms in anaerobic conditions, total P cycles through wetland vegetation and can be released as DRP from decaying wetland vegetation and bottom sediments. Consequently, we assumed no net reduction in total P loads from additional constructed wetlands.

Although not estimated, there are other edge-of-field practices that can be used to reduce sediment losses, including strip cropping, terraces, and WASCOBS (water and sediment control basins). See Czapar et al. (2008) for a summary and discussion of the effectiveness of these practices.

Table 14. Example statewide results for total P reductions by practice/scenario with shading to represent in-field, edge-of-field, land use change, and point source practices or scenarios.

Practice/Scenario	Total P reduction per acre (%)	Total P reduced (million lb P)	Total P reduction from baseline (%)	Cost ($/lb P removed)
Convert 1.8 million acres of conventional till eroding >T to reduced, mulch or no-till	50	1.8	5.0	-16.60
P rate reduction on fields with soil test P above the recommended maintenance level	7	1.9	5.0	-48.75
Cover crops on all corn/soybean tile-drained acres	30	4.8	12.8	130.40
Cover crops on 1.6 million acres eroding>T currently in reduced, mulch or no-till	50	1.9	5.0	24.50
Wetlands on 25% of tile-drained land	0	0	0.0	
Buffers on all applicable crop land	25-50	4.8	12.9	11.97
Perennial/energy crops equal to pasture/hay acreage from 1987	90	0.9	2.5	102.30
Perennial/energy crops on 1.6 million acres>T currently in reduced, mulch or no-till	90	3.5	9.0	40.40
Perennial/energy crops on 10% of tile-drained land	50	0.3	0.8	250.07
Point source reduction to 1 mg total P/L (majors only)		8.3	22.1	13.71

State-wide Scenarios with Costs

The final steps in our analysis was to combine practices and scenarios for nitrate-N and total P reductions to develop overall statewide N and P scenarios to reach either 20 or 45% reductions for each nutrient individually, and then a final combined scenario. These scenarios take into account that one practice may alter the effectiveness of another. Table 15 presents the nitrate-N scenarios that we developed, three for 45% reductions, and three for 20% reductions. All nitrate-N scenarios include the MRTN rate applied to all MLRAs, because that is a negative cost to farmers. Scenario N1 also includes all spring N application with cover crops, bioreactors, wetlands, and buffers to reach a 45% reduction in nitrate-N at a cost per lb of $3.67, and annual costs of $678 million per year. This scenario has the lowest cost, as it maximizes the practices that have the lowest costs per lb N removed such as bioreactors, buffers, and cover crops on tile-drained acres.

Scenario N2 uses mostly cover crops, perennials, and bioreactors, as well as point source reductions to reach a 45% reduction in nitrate-N, but the costs are increased to $4.34/lb N reduced, or $811 million per year. The final scenario is N3, which replaces the fall to spring and bioreactors with wetlands, but adds in buffers. Costs are the greatest at $4.51/lb N reduced, or $833 million per year. There is some total P reduction with many of the N practices, and they ranged from 20 to 33% for the three N scenarios.

Scenarios N4-6 all estimate a 20% reduction in nitrate-N, with N4 the cheapest at $1.99/lb N. This scenario includes MRTN, all spring N application, a small % of cover crops, and full implementation of bioreactors. Because bioreactors are the cheapest practice, this scenario can lead to a 20% reduction in nitrate-N at a relatively low cost of $162 million per year. Scenario N5 has a similar cost, keeping the bioreactors but replacing the fertilizer timing with 35% cover crops on tile drained acres. The final scenario, N6, shows the level of cover cropping needed for a 20% reduction, and that the cost would be substantially greater. However, there is much greater P removal with this scenario, compared to N4 or N5.

Table 15. Example statewide nitrate-N scenarios.

Name	Combined practices and/or scenarios	Nitrate-N (% reduction)	Total P (% reduction)	Cost of reduction ($/lb N)	Annualized costs (million $/yr)
N1	MRTN rate, all spring N application, cover crops 70% tile-drained & 45% non-tiled, bioreactors 50%, wetlands 25%, all ag streams have buffers	45	20	3.67	678
N2	MRTN rate, all spring N application, cover crops 100% tile-drained & 70% non-tiled, bioreactors 50%, perennial crops non-tiled, point source to 10 mg nitrate-N/L	45	33	4.34	811
N3	MRTN rate, cover crops 100% tile-drained & 70% non-tiled, wetlands 25%, perennial crops non-tiled, all ag streams have buffers, point source to 10 mg nitrate-N/L	45	24	4.51	833
N4	MRTN rate, all spring N application, cover crops 5% tile-drained, bioreactors 50%	20	0.3	1.99	162
N5	MRTN rate, cover crops 35% tile-drained, bioreactors 50%	20	2	2.00	161
N6	MRTN rate, cover crops 75% tile-drained, 55% non-tiled	20	8	4.55	370

In Table 16 we present statewide scenarios for total P. Scenario P1 includes reducing P fertilizer application, reduced tillage, buffers, and point source reductions to reach a 45% reduction in total P, for a cost of $48 million per year. Because most of the agricultural practices save money, the overall cost is reduced. However, there is a positive cost of $114 million per year for point source P reduction. Scenario P2 includes the reduction in P fertilizer and reduced tillage, but then adds cover crops so all agricultural practices are in-field. Costs at $36.44/lb P reduced, or $615 million per year. Scenario P3 adds in perennials (land-use change) on many acres as an alternative, and demonstrates the increased cost ($41.24/lb P reduced). However, this scenario has the greatest nitrate-N reductions, at 37%, compared to only 7% with scenario P1.

Table 16. Example statewide total P scenarios.

Name	Combined practices and/or scenarios	Nitrate-N (% reduction)	Total P (% reduction)	Cost of reduction ($/lb P)	Annualized costs (million $/yr)
P1	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, buffers on all applicable lands, point source to 1.0 mg total P/L	7	45	2.84	48
P2	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on all CS, point source to 1.0 mg total P/L	29	45	36.44	615
P3	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on 87.5% of CS, buffers on all applicable lands, perennial crops on 1.6 million ac >T, and 0.9 million additional ac.	38	45	41.24	696
P4	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, buffers on 80% of all applicable land	6	20	-10.40	-78
P5	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, point source to 1.0 mg total P/L on 45% of discharge	0	20	--9.73	-73
P6	No P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on 1.6 million ac eroding >T and 40% of all other CS	11	20	22.93	172

For a 20% reduction in total P, scenarios P4 and P5 use fertilizer reductions and reduced tillage along with either buffers or point source reductions to reach the intermediate target. Costs are about -$73 to -$78 million dollars per year. The final scenario, P6, uses all in-field practices (i.e., there is no reduction of row crop acres) and has costs of $172 million per year for a 20% reduction in total P.

The final analysis was to develop scenarios that met both nitrate-N and total P targets of either 20 or 45% reductions. Again, we had three scenarios that approached the 45% reduction, and three for 20% reductions to illustrate the range of practices and costs that would be involved (Table 17). Scenario NP1 is the cheapest at $383 million per year, but only achieves a 35% nitrate-N reduction. We present it to show that getting both nitrate-N and total P reductions of 45% is quite a bit more expensive than this scenario. Scenario NP2 does hit both targets, and includes fertilizer N and P changes, reduced tillage, cover crops on all corn and soybean land, and point source reductions. These are extensive practice changes, and the costs are $810 million per year. Scenario NP3 achieves the 45% reductions by including land use changes, and costs $791 million per year.

A 20% reduction in nitrate-N and total P can be achieved for $48 million per year (scenario NP4), due to the cost saving from less N and P fertilizer. Scenario NP5 used both agricultural and point source reductions to reach 20% reductions, for a cost of $66 million per year. Finally scenario NP6 demonstrates the costs for only agricultural practices, which is substantially greater at $244 million per year.

Costs by practice for each of the combined scenarios are presented in Table 18 to illustrate how the total costs were calculated. Three practices have negative costs (MRTN, reduced P fertilizer, and reduced tillage), whereas cover crops and point P have some of the higher costs.

None of these scenarios include any potential changes in yields, which might occur in some cases. This is beyond the scope of what we can estimate. Many of the practices also include large first year costs (e.g., bioreactors, wetlands) that would likely require a phased implementation. There are also many other considerations that could affect any of the scenario estimates that we have made, and we refer readers to the excellent summary of these considerations discussed in section 2.4 of the Iowa Science document (Iowa, 2013).

Table 17. Example statewide nitrate-N and total P scenarios.

Name	Combined practices and/or scenarios	Nitrate-N (% reduction)	Total P (% reduction)	Cost of reduction ($/lb)	Annualized costs (million $/yr)
NP1	MRTN, all spring N application, bioreactors 50%, wetlands 25%, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, buffers on all applicable lands, point source to 1.0 mg total P/L and 10 mg nitrate-N/L	35	45	**	383
NP2	MRTN, all spring N application, bioreactors 50%, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on all CS, point source to 1.0 mg total P/L and 10 mg nitrate-N/L	45	45	**	810
NP3	MRTN, all spring N application, bioreactors 15%, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on 87.5% of CS, buffers on all applicable lands, perennial crops on 1.6 million ac >T, and 0.9 million additional ac	45	45	**	791
NP4	MRTN, all spring N application, bioreactors 35%, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, buffers on 80% of all applicable land	20	20	**	48
NP5	MRTN, all spring N application, bioreactors 30%, wetlands 15%, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, point source to 1.0 mg total P/L and 10 mg nitrate-N/L on 45% of discharge	20	20	**	66
NP6	MRTN, all spring N application, no P fert. on 12.5 million ac above STP maintenance, reduced till on 1.8 million ac conv. till eroding > T, cover crops on 1.6 million ac eroding >T and 40% of all other CS	24	20	**	244

Table 18. Combined nitrate-N and total P scenarios with costs by practice (million $ per year).

Practice	NP1	NP2	NP3	NP4	NP5	NP6
MRTN	-9.6	-9.6	-9.6	-9.6	-9.6	-9.6
N fertilizer timing	82	82	82	82	82	82
Bioreactors	77	77	23	54	46	
Wetlands	140					
Buffers	58		58	46		
Cover crops		625	547			296
Perennials			215			
Reduced P fertilizer	-94	-94	-94	-94	-94	-94
Reduced tillage	-30	-30	-30	-30	-30	-30
Point source P	114	114			51	
Point source N	46	46			21	
Sum	383	810	791	48	66	244

Conclusions

This report has described the current state and long-term trends in nutrients in Illinois rivers, as well as the sources of those nutrients. Illinois had annual riverine nutrient loads of 536 and 37.5 million lb of nitrate-N and total P, respectively, leaving the state for 1997-2011. Agricultural sources contributed about 80% of the nitrate-N exported by Illinois rivers, with 18% from point sources, and 2% from urban runoff. For total P, agriculture and points source each contributed 48% of riverine loads, whereas 4% was from urban runoff. To meet a 45% reduction target would take major changes in both point and agricultural sources. Because several of the practices recommended save money, a 35% reduction in nitrate-N and a 45% reduction in total P could be met at an annualized cost of $383 million per year. To get the additional 10% in nitrate-N reduction leads to annualized costs of about $800 million per year.

It should be noted that the estimates of potential nutrient reductions provided in this report are rough estimates based on existing research literature, data, and professional judgment. Limitations of time, resources and data did not allow us to conduct more detailed analysis, but future refinements are encouraged. To make our reduction estimates we used a single percentage effectiveness for a given practice. However, there are many uncertainties in the effectiveness of a specific practice in reducing N or P losses in any one field or year and this needs to be taken into account when our results are interpreted. We point out that more research is needed to provide these estimates, and is currently underway in the state. The P assessment did not include stream bed and bank erosion as sources of P, nor did we include losses of P from ephemeral gulley erosion. Data are not currently available to estimate these potential sources of P throughout Illinois. Reliable data on fertilizer and manure management practices would greatly enhance the reliability of future assessments of nutrient losses and the likely impacts of conservation efforts.

References

Camp Dresser & McKee Inc. 2008. Report Cost/Benefit Study of the Impacts of Potential Nutrient Controls for Colorado Point Source Discharges.

CH2MHILL. 2010. Statewide nutrient removal cost impact study prepared for Utah Division of Water Quality. October, 2010. Salt Lake City, Utah.

Christianson, L., J. Tyndall, and M. Helmers. 2013. Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage. Water Resources and Economics 2-3:30-56.

Clover, M.W. 2005. Impact of nitrogen management on corn grain yield and nitrogen loss on a tile drained field. M.S. thesis, University of Illinois at Urbana-Champaign. 44 pages.

Cooke, R., and S. Verma. 2012. Performance of drainage water management systems in Illinois, United States. Journal of Soil and Water Conservation 67: 453-464.

Czapar, G.F., J.M. Laflen, G.F. McIsaac, and D.P. McKenna. 2008. Effects of erosion control practices on nutrient loss. p. 117–127. In Upper Mississippi River Subbasin Hypoxia Nutrient Committee (editors) Final report: Gulf Hypoxia and Local Water Quality Concerns Workshop. American Society of Agricultural and Biological Engineers, St. Joseph, MI.

David, M.B. and L.E. Gentry. 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois, USA. Journal of Environmental Quality 29:494-508.

David, M.B., G.F. McIsaac, and C.A. Mitchell. 2008. Water quality improvement case study: assessment of the Lake Bloomington watershed. Report submitted to Illinois Department of Agriculture, March 10, 2008. 17 pages.

David, M.B., L.E. Drinkwater, and G.F. McIsaac. 2010. Sources of nitrate yields in the Mississippi River basin. Journal of Environmental Quality 39:1657-1667.

Donohue & Associates. 2010. Nutrient removal study North Shore Sanitary District. North Short Sanitary District, Illinois.

DuPage River Salt Creek Workgroup. 2008. WinSLAMM3 model outputs for West and East Branch of the DuPage River.

Fernández, F.G., B.S. Farmaha, and E.D. Nafziger. 2012. Soil fertility status of soils in Illinois. Communications in Soil Science and Plant Analysis 43: 2897-2914.

Frankenberger, J., E. Kladivko, G. Sands, D. Jaynes, N. Fausey, M. Helmers, R. Cooke, J. Strock, K. Nelson, and L. Brown. 2006. Drainage Water Management for the Midwest, Questions and Answers About Drainage Water Management for the Midwest. WQ-44, Purdue Agriculture.

Gentry, L.E., M.B. David, and G.F. McIsaac. 2014. Variation in riverine nitrate flux and fall nitrogen fertilizer application in east-central Illinois. Journal of Environmental Quality (in press).

Groffman, P.M., N.L. Law, K.T. Belt, L.E. Band, and G.T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393-403.

Helsel, D.R., and R.M. Hirsch. 2002. Statistical methods in water resources: U.S. Geological Survey Techniques of Water-Resources Investigations, Book 4, Chapter A3, available at http://pubs.usgs.gov/twri/twri4a3/

Herrera Environmental Consultants. 2011. Toxics in Surface Runoff to Puget Sound: Phase 3 Data and Load Estimates. Prepared for the Washington State Department of Ecology by Herrera Environmental Consultants, Inc., Seattle, Washington. May 2011. Publication Number 11-03-010.

Hirsch, R.M., D.L Moyer, and S.A. Archfield. 2010. Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay River inputs. Journal of the American Water Resources Association 46:857-880.

Horner, R.R., J.J. Skupien, E.H. Livingston, and H.E. Shaver. 1994. Fundamentals of Urban Runoff Management: Technical and Institutional Issues. Terrene Institute, Washington, DC in cooperation with U.S. Environmental Protection Agency.

Illinois Environmental Protection Agency. 1986. Phosphorus: A Summary of Information Regarding Lake Water Quality. IEPA/WPC/86-010.

Iowa State University. 2013. Corn Nitrogen Rate Calculator. http://extension.agron.iastate.edu/soilfertility/nrate.aspx

Iowa. 2013. Iowa Nutrient Reduction Strategy. A Science and technology-based framework to assess and reduce nutrients to Iowa waters and the Gulf of Mexico. Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources, and Iowa State University College of Agriculture and Life Sciences.

Jacobson, L.M., M.B. David, and L.E. Drinkwater. 2011. A spatial analysis of phosphorus in the Mississippi River basin. Journal of Environmental Quality 40:931-941.

Jiang, F., M.B. Beck, R.G. Cummings, K. Rowles, and D. Russell. 2005. Estimation of costs of phosphorus removal in wastewater treatment facilities: adaptation of existing facilities. Water Policy Working Paper #2005-011.

Minnesota. 2013. The Minnesota Nutrient Reduction Strategy. WQ-S1-80.

Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. 2008. Gulf Hypoxia Action Plan 2008 for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico and Improving Water Quality in the Mississippi River Basin. Washington, DC.

Mosher, R. 2013. Reality checking USEPA's nutrient loading tool. Illinois Environmental Protection Agency White Paper. 3 pp.

Royer, T.V., M.B. David, and L.E. Gentry. 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: implications for reducing nutrient loading to the Mississippi River. Environmental Science and Technology 40:4126-4131.

Russell, A. 2013. Suspended sediment and total phosphorus loadings from small agricultural watersheds in western Illinois. M.S. thesis, University of Illinois at Urbana-Champaign.

Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.J. Gburek, P.A. Moore, and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the United State. Journal of Soil and Water Conservation 58:137-152.

Sharpley, A.N., T.C. Daniel, J.T. Sims, J. Lemunyon, R.A. Stevens, and R. Parry. 2003. Agricultural Phosphorus and Eutrophication. Second Edition. USDA-ARS Report 149. Washington, DC: US Government Printing Office.

Skaggs, R.W., N.R. Fausey, and R.O. Evans. 2012. Drainage water management. Journal of Soil and Water Conservation 67:167-172.

Smith, C.M., M.B. David, C.A. Mitchell, M.D. Masters, K.J. Anderson-Teixeira, C.J. Bernacchi, and E.H. DeLucia. 2013. Reduced nitrogen losses following conversion of row crop agriculture to perennial biofuel crops. Journal of Environmental Quality 42:219-228.

Sprague, L.A., R.M. Hirsch, and B.T. Aulenbach. 2011. Nitrate in the Mississippi River and its tributaries, 1980 to 2008: Are we making progress? Environmental Science & Technology 45: 7209–7216.

Sunohara, M.D., E. Craiovan, E. Topp, N. Gottschall, C.F. Drury, and D.R. Lapen. 2014. Comprehensive nitrogen budgets for controlled tile drainage fields in eastern Ontario, Canada. Journal of Environmental Quality 43:617-630.

USDA-NRCS. 2006. Land resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin. USDA Handbook 296.

USEPA. 1999. Preliminary data summary of urban storm water best management practices. EPA-821-R-99-012. USEPA Office of Water, Washington, DC.

USEPA. 2007. Hypoxia in the northern Gulf of Mexico: An update by the EPA Science Advisory Board. EPA-SAB-08-004. USEPA, Washington, DC.

USEPA. 2008. Municipal Nutrient Removal Technologies Reference Document, EPA 832-R-08-006, US Environmental Protection Agency.

Woli, K.P., M.B. David, R.A. Cooke, G.F. McIsaac, and C.A. Mitchell. 2010. Nitrogen balance in and export from agricultural fields associated with controlled drainage systems and denitrifying bioreactors. Ecological Engineering 36:1558-1566.

Glossary

Effluent – The liquid or gas discharged from a process or chemical reactor, usually containing residues from that process.

Non-point source – A diffuse source of chemical and/or nutrient inputs not attributable to any single discharge (e.g., agricultural surface runoff, agricultural subsurface drainage by tile systems, urban runoff, atmospheric deposition).

Nutrients – Inorganic chemicals (particularly nitrogen, phosphorus, and silicon) required for the growth of plants, including crops and phytoplankton.

Point source – Readily identifiable inputs where treated wastes are discharged from municipal, industrial, and agricultural facilities to the receiving waters through a pipe or drain.

Watershed – The drainage basin contributing water, organic matter, dissolved nutrients, and sediments to a stream or lake.

Watershed load – total mass or amount of a nutrient or chemical that leaves the watershed in stream flow during a given amount of time, typically tons or lb per year. Nutrient loads from larger watersheds are often greater than loads from smaller watersheds.

Watershed yield – load of a nutrient or chemical for a given time period divided by the drainage area of the watershed (in square km, ha, square miles, or acres). Because nutrient loads usually increase with watershed size, the yield indicator facilitates comparisons between watersheds of different sizes. Units are typically kg ha^{-1} yr^{-1}, or lb/acre/year.

Appendix 1 - Non-point Source Cost Estimates

This section provides cost estimates for each practice that can be used in strategies to reduce nitrate-N and total P losses from agricultural fields. These cost estimates are stated on an annual basis in dollars per acre, where the acre is on farmland on which the practice is implemented. For example, a practice may have a $6 per acre cost. This indicates that it will cost $6 per acre to implement the practice each year. If the practice is projected to be implemented on ten million acres, the total cost of the practice would be $60 million per year ($6 per acre x 10 million acres). If that practice is implemented each of the next ten years, there will be a $60 million cost per year.

The following subsections provide detail on methods used to estimate costs for each practice. Before proceeding to each practice, five points germane to all practices are made.

First, the method used in generating cost estimates is a partial budgeting approach. In this approach, a "base" case is specified that represents general agricultural practices. Then, a nutrient reduction practice is specified. Changes in costs from the current practice and the reduction practice then are estimated. Hence, cost estimates represent a change from the current, general practice.

Second, some of the following practices have initial investments. These investments are made in the first year, with the benefits of the investment accruing over many years. In these cases, the initial investment cost is amortized over the life of the investment using an annualized equivalence approach. A discount factor of 6 percent is used in calculating annualized investment costs. A 20 year life is used on most practices.

Third, some of the practice changes could have impacts on yields. For example, some of the proposed changes are to shift N applications from the fall to the spring, and others split applications of N over more than one application period. Research continues on these practices, with some research indicating that these practices can increase yields or reduce nutrient applications. In determining whether to include a yield change for a practice, *the Illinois Agronomy Handbook* was consulted (http://extension.cropsci.illinois.edu/handbook/), with particular emphasis given to Chapter 9 entitled *Managing Nitrogen*. If that publication indicated that a yield change was warranted, then a yield change is included in the following practice. The *Illinois Agronomy Handbook* is treated as a guide to standard agronomic practices in Illinois. Some of the newer timing and split application strategies may prove beneficial. As of yet, however, they are not standard, and there are likely economic and agronomic reasons that they are not standard. In addition, research-based results do not necessarily translate to the general situations, as general practices differ from those of the research setting.

Fourth, the magnitude of the costs needs to be kept in context of the earning potential from farmland. Some gauge of these costs can be gained by looking at per acre net returns to farmers where the farmland is rented at average cash rent (see management section of *farmdoc*). Net returns to the farmer in central Illinois averaged $56 per acre from 2000 through 2006. Returns from 2007 through 2013 were higher, averaging $195 per acre. The 2007 through 2013 period likely will be remembered as a high profit area, as returns were higher than historical averages. Current projection place estimated returns over the next five-years around $55 per acre, with the potential for 2014 and 2015 to be much lower than $55 per acre. Using $55 per acre as a guideline, a practice with a $10 cost represents an 18% reduction in returns to the farmer. On a percentage basis, many of the following strategies represent a significant reduction in agricultural returns.

Fifth, adoption of any of these practices by farmers will depend on more than simply the costs of the practices. Two factors are particularly critical. High initial investment costs will have the potential to reduce incentives to implement the practice, as debt capital may be required to implement the practice, and adding debt increases the risk exposure farms face. The second is timing risks. Many of the strategies change the timing of field operations, particularly moving field operations to the spring and post planting period. There are limited numbers of days suitable for field work during these periods. Placing more field operations in the spring time frame increases the risk of not planting in a timely manner, thereby reducing profits from agricultural operations. These concerns are listed as "caveats" to each practice.

Table A1 includes the costs of each of the practices, along with a listing of some of the concerns that may impede adoption. The following sub-sections describe each of the practices in more detail.

Reduced Tillage

The base practice includes a "heavy" tillage pass. The alternative is to eliminate that tillage pass, but this alternative would still include tillage and is not a no-till system. The cost of this strategy is the reduction of one tillage pass. The cost of the tillage pass is taken M*achine Cost Estimates: Field Operations* published by the University of Illinois, May 2012 (http://www.farmdoc.illinois.edu/manage/index.asp). The particular implement used in the cost is a horizontal disk, drag, rolling basket. The cost is negative (-$17 per acre), indicating savings.

Caveats:

Many farmers undertake a tillage pass on fields that were corn in the previous year and will be corn in the next year. There is reason to believe that breaking up residue aids in stalk decomposition, potentially leading to higher yields on corn in the following year. A yield reduction would reduce savings indicated for this practice.

Eliminate Phosphorus Applications

The base strategy assumes that a soil has high soil P levels that were obtained through a combination of inherently high P levels in the soil, commercial applications of fertilizers, and applications of manure. Not all soils meet these conditions. The nutrient reduction scenario estimated here is eliminating P applications for six years to bring down the soil test levels.

In costing this scenario, it is assumed that 170 pound lb diammonium phosphate (DAP, a fertilizer that is 18% N, 46% P, and 0% K) is the amount of fertilizer that is eliminated during several rotations to bring the soil test P level down. The 170 pound application rate represents a maintenance level of fertilizer application, and is equal to 34 lb P per acre per year. It is typically applied at this rate every other year. This compares well to the 11 to 13 lb P per year P fertilizer application previously estimated for the state (Table 9). A DAP fertilizer price of $540 per ton is used in calculating the cost. A credit is given for N in DAP, as the N in DAP likely will have to be replaced by some other commercial N source. The credit is based on $680 per ton anhydrous ammonia price. Given the credit, elimination of P fertilizers in one year will result in a cost savings of $34 per year.

The present value of three applications of $34 per year is calculated, which is what would typically be applied during a six-year period. With the present value of P application of $34 per year, the annualized yearly value during a 20 year period is then calculated. A 6% discount factor is used in calculating both the present value and the annualized cost. The annual yearly cost for reducing P fertilizer is -$7.50 per year, indicating that this practice reduces costs.

Stream Buffers

This practice takes acres out of production so as to provide a buffer near streams and other water bodies. The base practice is an acre in production. The nutrient reduction practice takes the acre out of production, and plants grass on that acre.

The buffer eliminates all income potential from the acre. The cost of this will be represented by the cash rent from the acre. Implementing the practice will be planting grass, a one-time investment costs. In addition, maintenance of the buffer acre must occur. Hence, the total costs of the buffer include three items:

1. The cash rent value of the farmland is given up. The central Illinois average cash rent is $280 per acre and this area would likely would be a target for buffers, but this could be adjusted for other areas of the state. For example in southern Illinois, the cash rent value would be much less. (see *Cash Rents in Illinois* in management section of farmdoc)
2. There will be $50 per acre of grass planting costs (including seed and tillage). This is amortized over a 20 year life using a 6% interest rate. The cost is $4.36 per acre.
3. There are $10 per acre in maintenance costs per year

Total costs are $294 per acre.

Caveats:

Taking farmland out of production may have negative impacts on farmland price, as tillable acres typically sell for higher values than do non-tillable acres. These costs are not included in the estimates.

Reducing N Rates

Under this practice, the base case assumes that N in being over-applied by 10% of producers. Our analysis discussed previously indicated that on average farmers are adding the recommended rate of N fertilizer, but we are making the assumption here that 10% are likely over applying. The nutrient reduction practice is to reduce N applications by 20 pounds per acre (to MRTN), resulting in a savings of $8 per year.

Nitrification Inhibitor

This practice adds a nitrification inhibitor to all fall applied fertilizer. The cost of this practice is placed in a conservative manner ($7 per acre) because it is likely that use of N inhibitor will result in a reduction of N applications. Use of a N inhibitor is a relatively standard practice for fall applied N in Illinois.

Split N Fertilizer Applications

The base case for this practice is fall applied N. The N reduction strategy is to split N application with ½ the N applied in the fall, and one-half the N applied in the spring. Under both the base and nutrient reduction practice a total of 160 pounds of actual N are applied. Under the base case, the N application in the fall is applied as anhydrous ammonia. Under the reduction practice, the fall application still is made as anhydrous ammonia. The spring application is divided: with one-half made as anhydrous ammonia and one-half made as 28% N solution. The N solution is used to speed up application. Under the N reduction practice, the following costs are incurred:

1. There is an additional anhydrous ammonia pass on one-half of acres. From the University of Illinois *Machinery Cost Estimates*, an anhydrous ammonia pass costs $12.20 per acre. Since this pass is on one-half the acres, this cost equals $6.10 per acre.
2. There is an additional N solution pass on one-half of acres. From the University of Illinois *Machinery Cost Estimates*, an N solution pass costs $9.10 per acre. Since this is on one-half the acres, this cost equals $4.55 per acre.
3. Nitrogen solutions have average $.16 per pound higher in active N between 2010 and 2012. The cost is $.16 per acre x 40 pounds of active N, of $6.40 per acre

Total cost of this practice is $17 per acre

Caveats:

This practice adds another operation pass to the spring time period. There are only a limited number of days available during spring, with slightly less than one-half of all days typically being suitable for field work. Adding a field operation could potentially delay planting, which could lower yields. Costs associated with delayed planting and additional risks are not included in the costs. The risk of delayed planting could limit adoption of this practice.

Fall to Spring N Fertilizer Application

The base practice is fall application of N. The reduction practice moves the fall application to the spring. In both cases, 160 pounds of N are assumed to be applied. This will have two impacts on costs. First, due to timeliness concerns, one-half the spring application will be switched from anhydrous ammonia to N solutions. Second, switching all tile drained soils to a spring application will result in the need for additional N application and storage structure, so as lower chances of late N applications. This will increase costs, which will be reflected in higher N and application costs. This practice change will result in the following cost changes.

1. Spring applications will result in a shift away from anhydrous ammonia applications to N solutions. One-half of applications are assumed to switch to 28% N solutions. The

additional cost of liquid N (28%) over anhydrous ammonia has averaged $.16 per pound of actual N between 2010 and 2013. This switch increases costs to $12.80 per acre

1. Anhydrous ammonia prices have averaged $.02 higher in spring than in fall. It is assumed that this will be a permanent increase in price, resulting in $1.60 per acre in increased costs.
2. Movement to spring of all tile-drained soils will require an increase in N infrastructure. It is estimated that will increase costs of N by $20 per ton, or about $.017 per pound of actual N. This will increase costs by $2.70 per acre.
2. A switch from fall to spring would require additional equipment due to timeliness constraints. Based on work-day probability analysis, a shift would result in a 20% loss of aces covered per machine, leading to an increase in per acre application costs of $.80 per acre

Total costs of this practice change are $18 per acre.

Caveats:

Movement will result in significantly larger chances of timeliness concerns, and late planting. The additional costs associated with late planting are not included in the above analysis. Timeliness concerns could prove to be a hindrance to adoption of this practice.

Cover Crops

The base case is no cover crop. The reduction practice is to plant a cover crop. Costing is based on planting rye into standing corn by aerial applying seed. The cover crop then is chemically killed in the spring. However, the cost of the herbicide application is not included, as an herbicide application or tillage pass would occur under the base case of no cover crop. However, a $5 per acre partial spray is included to cover any additional problems.

Costs for this strategy are:

1. Aerial application costs of $16 per acre,
2. Seed costs of $8 per acre ($16 per bushel x one-half bushel per acre), and
3. Partial spray costs of $5 per acre.

Total costs of this strategy are $29 per acre.

Caveats:

Farmer's use of cover crops may introduce additional management problems, particularly in adverse years. Establishing cover crops may be difficult in years of dry summers and falls. Cover crops may lead to a reduction in crop yields in adverse years, and as indicated earlier, this is not included in our cost estimates. For example, the cover crop may reduce available moisture in dry springs, leading to lower yields. Performing planting operations may introduce logistical issues on farms.

Bioreactors

This base practice is no bioreactor. The reduction practice is construction of a bioreactor at the end of the tile line. Cost estimates for the practice are taken from Christianson et al. (2013). Costs for this strategy are:

1. Investment costs of $133 per acre. These are stated on an annual basis using a 6% discount rate and a 20 year life. Annualize costs are $12 per acre
2. Yearly maintenance costs equal $5.00 per acre

Total costs for his strategy are $17 per acre.

Caveats:

A bioreactor represents a significant investment, likely requiring debt capital, thereby increasing the risk to farmers. In addition, there are no cash flows from this project, providing funds for payback on capital. In addition, large amount of construction of bioreactors could increase the investment costs in bioreactors.

Wetlands

The base practice is no wetland. The reduction practice is the construction of a wetland, taking up 5 acres for the wetland per 100 acres of production. Design of the wetlands were taken from Christianson et al. (2013).

The major cost of the wetland is farmland taken out of production. Costs are:

1. The major cost is farmland taken out of production. A farmland value of $12,500 per acre is used in costing. In addition, $60 per acre of designing the wetland are included. The 5% of farmland are taken out of production is charged against the 95% of farmland remaining in production. The $12,560 of investment costs is amortized over a 20 year horizon at a 6% interest rate. This results in a cost of $1,095.04 per acre of wetland. These costs are stated on a per acre of production, or $57.63 per year ($1.095.04 per acre of wetland x 5 acres of wetland / 96 acres of production)
2. A maintenance cost of $3 per acre is included for the wetland .

Total costs are $60.63 per acre

Caveats:

This practice represents a large decrease in income generating potential. Adoption of this practice will be slow due to the costs of wetland. Also, reductions in the value of the property may occur beyond those included here.

Moving to a Perennial Crop

This base practice is corn and soybean production in a 50% corn – 50% soybean rotation. The reduction practice is moving to a perennial crop. Alfalfa is the perennial crop modeled for this practice. This cost change equals returns from corn and soybean production minus returns from alfalfa production

Corn and soybean returns are taken from *2014 Illinois Crop Budgets* for high-productivity farmland. Operator and farmland returns equal $309 for corn and $265 for soybeans. These are averaged to arrive at a $287 per acre return.

Returns for alfalfa production are modeled from 2013 Iowa budgets. The following changes were made:

1. The land charge was taken out of Iowa budgets to be consistent with Illinois budgets.
2. The hay stand life was increased from 4 year to 5 years. This increases alfalfa profitability.
3. Alfalfa yield was increased to 5 tons per acre. This increased alfalfa profitability.

These changes result in a $201 per acre to alfalfa production.

Costs of this strategy are $86 per acre ($287 return from corn-soybean rotation – $201 return for hay).

Caveat:

Large scale movement to perennial crop production would require dramatic changes in agricultural structure. In particular, alfalfa (and most forages) are fed to ruminant livestock (predominately beef or dairy production). Increases in alfalfa acres, by necessity, imply increases in ruminant livestock. Without this change, hay prices likely would decline, leading to higher costs for this practice.

Table A1. Costs of agricultural practices and other economic concerns with practices.

Practice	Cost per acre	Other economic concerns
Reduced tillage	-$17	Yield reduction may occur because tillage is not included
P rate reduction	-$7.50	
Stream buffer	$294	Cost is per acre of buffer; negative impacts on farmland
Reducing N rates from background to MRTN	-$8	
Nitrogen inhibitor with all fall applied fertilizer	$7	
Split N fertilizer application on tile-drained soils (50% fall, 50% spring)	$17	
Fall to spring N fertilizer application on tile-drained tiles	$18	Timeliness concerns
Cover crops	$29	Difficulty in planting; potential impact on yields
Bioreactor	$17	Large investments costs; increasing costs with large adoption
Wetland	$61	Large investment costs
Moving to a perennial crop	$86	Large shifts to forages likely would result in much lower forage prices