# Section 2.1
# Executive Summary – Iowa Science Assessment of Nonpoint Source Practices to Reduce Nitrogen and Phosphorus Transport in the Mississippi River Basin

Prepared by the Iowa State University Science Team
May 2013

## Introduction

The 2008 Hypoxia Action Plan calls for states along the Mississippi River to develop nutrient reduction strategies to reduce, mitigate, and control hypoxia in the Gulf of Mexico and improve overall water quality. In October 2010, the Iowa Department of Agriculture and Land Stewardship and the College of Agriculture and Life Sciences at Iowa State University partnered to conduct a technical assessment needed for the development of a statewide strategy to reduce nutrient to streams and the Gulf of Mexico. The team working on this effort consisted of 23 individuals representing five agencies or organizations. Within the overall team, sub-group science teams were formed to focus on nitrogen, phosphorus and hydrology.

The goals of the process were to assess nutrient loading from Iowa to the Mississippi River and the potential practices needed to achieve desired environmental goals. As per the 2008 Gulf Hypoxia Action Plan, these goals are a 45% reduction in riverine N and P load. In conjunction with this non-point source assessment, the Iowa Department of Natural Resources (IDNR) has been conducting an assessment of nutrient loads from point sources.

Based on IDNR estimates, nonpoint source load reductions for nitrate-N would need to achieve 41% load reduction in nitrate-N with the remaining 4% coming from point sources (Table 1). For phosphorus, the nonpoint source load reductions would need to achieve 29%, with the remaining 16% coming from point sources.

**Table 1. Estimated percent load contributions from point and non-point sources.**

| Estimated % of Loads and Load Reduction | Nitrogen | Phosphorus |
|---|---|---|
| % of Total Load from Point Sources | 7 | 21 |
| % of Total Load from Non-point Sources | 93 | 79 |
| % of Overall Load Reduction from Point Sources to meet 45% Total Load Reduction Goal | 4 | 16 |
| % of Overall Load Reduction from Nonpoint Sources to meet 45% Total Load Reduction Goal | 41 | 29 |

## Process

The assessment was conducted in the following steps:

1. *Establish baseline conditions*
   Available information was used to estimate existing conditions relative to nutrient application, timing of nutrient application, existing soil test phosphorus conditions, land use, crop rotations, extent of current

tillage practices, estimated extent of land benefitting from tile drainage, and estimated extent of existing conservation practices. These conditions were aggregated by Major Land Resource Area (MLRA). **Based on this review, it is clear there is a lack of information on existing conditions, and a need for greater on-going documentation and reporting of this information.**

2. *Review scientific literature to assess potential performance of practices*
   A comprehensive list of practices potentially reducing nitrate-N or phosphorus export was assembled and refined based on practices expected to have the greatest potential impact and for which there was research data on the impact to water quality. An extensive review of scientific literature was conducted to assess the potential impact on nitrate-N and phosphorus reductions. Studies included were limited to those conducted in Iowa or surrounding states so climatic conditions would be similar to Iowa conditions. Initial documents on baseline conditions and practice performance were subjected to outside blind peer review.
3. *Estimate potential load reductions of implementing nutrient reduction practices (scenarios)*
   The potential for nitrate-N and phosphorus load reduction with implementation of individual practices or a combination of practices was assessed using the baseline data and information on practice performance. Scenarios of practice combinations where the water quality goals could potentially be achieved were identified. **It is important to note these scenarios represent EXAMPLES of practice combinations and are not the recommendations of the science team.**
4. *Estimate cost of implementation and cost per pound of nitrogen and phosphorus reduction*
   Economic costs of combination scenarios were computed considering the cost for implementing the practice and any potential impact on crop yield, specifically corn grain yield. An equal annualized cost (EAC) was computed so those practices with annualized costs and those with large initial capital costs could be appropriately compared.

## Nutrient Reduction Practices

### *Nitrogen*

Nitrogen reduction practices ranging from in-field nitrogen management practices to edge-of-field practices to land use change were reviewed to assess the potential for nitrate-N reduction and impacts on corn yield (Table 2). Based on this review, practices related to the timing of nitrogen application resulted in less than a 10% reduction in nitrate-N, no matter the timing of nitrogen application. In addition, all of these timing practices had high standard deviations (20% or greater), indicating that certain years there could be a fairly dramatic increase in nitrate-N.

For example, moving from fall to spring pre-plant nitrogen application, the percentage of nitrate reduction plus or minus one standard deviation is -19% to 31%. Inclusion of a nitrification inhibitor with fall-applied nitrogen had slightly higher nitrate-N reduction than the timing practices (9% reduction) but the standard deviation was still 19%. For the nitrogen management practices that consider nitrogen rate, timing, or source, the rate of nitrogen application and, specifically, reducing the average nitrogen application rate to the Maximum Return to Nitrogen Rate (MRTN) shows greatest potential for nitrate-N reduction. It should be noted some of the nitrogen timing or inhibitor practices show potential to increase corn yield. Overall, for the practices categorized as a nitrogen management practice, cover crops and living mulches show the greatest potential for nitrate-N reduction. However, both a rye cover crop and kura clover living mulch have the potential for reduced corn yield. Reducing potential negative corn yield impacts when utilizing a cover crop or living mulch is an area where future research is needed.

Land use change through conversion of corn-soybean systems to perennial vegetation or extended rotations show potential to dramatically reduce nitrate-N, but conversion to these perennial-based systems would reduce the acreage of corn-soybean. Edge-of-field practices also show potential for substantial reduction in nitrate-N and require little land to be taken out of row crop production.

***Phosphorus***

Phosphorus reduction practices ranging from in-field phosphorus management practices to erosion control to edge-of-field practices to land use change were reviewed to assess the potential for phosphorus reduction and impacts on corn yield (Table 3). Based on this review, phosphorus management practices have the potential to reduce phosphorus loss, but in all cases the standard deviations associated with these reductions were fairly large - greater than 27%. Reducing tillage intensity has the potential to significantly reduce phosphorus loss, especially when no-till is compared to a chisel plow system (90% reduction in phosphorus load).

Land use change through conversion of row crop systems to perennial vegetation shows potential to dramatically reduce phosphorus but conversion to these perennial-based systems would reduce the acreage of corn-soybean. Edge-of-field practices through buffers or sedimentation basins show potential for dramatic reductions in phosphorus load, 58% and 85% respectively. However, the realized performance of edge-of-field practices will be dependent upon the characteristics of the contributing area and design of the buffers or sedimentation basins.

## Estimated Potential for Nutrient Load Reduction

***Nitrogen***

To estimate the baseline nitrate-N load, estimates of existing land use, literature estimates of nitrate-N concentrations in tile and subsurface water, and estimates of water yield to streams were used to compute a baseline nitrate-N load. The loads were calculated for each MLRA in Iowa and loads were accumulated for a statewide load. To assess the impact of the nitrogen practice implementation, the baseline nitrate-N concentrations were adjusted based on literature estimates for each practice. These concentrations were used to compute a scenario load of nitrate-N, which was compared to the baseline load. From this comparison, the estimate of potential nitrate-N load reduction for each standalone practice was developed (Table 4). **It is important to note the computed reductions for standalone practices are not additive. In other words, it's not possible to add together reductions from multiple practices.**

From Table 4, the nitrogen management practices with the greatest potential for nitrate-N reduction are a reduction in nitrogen application rate or planting cover crops. Currently, the estimated average nitrogen application (commercial fertilizer and manure) to corn in a corn-soybean rotation is 151 lb-N/acre and 201 lb-N/acre to corn in continuous corn rotation. The MRTN for corn following soybean is 133 lb-N/acre and 190 lb-N/acre for corn following corn ($5.00/bushel corn and $0.50/lb nitrogen). In addition, sidedressing nitrogen rather than just a spring pre-plant application has some potential for nitrate-N reduction (4%). Moving nitrogen that is currently fall applied (estimated to be about 25% of the total fertilizer nitrogen for corn) to spring application shows little potential for overall nitrate-N reduction (less than 1%).

The edge-of-field practices of wetlands targeted for water quality benefits and subsurface drainage bioreactors show the greatest potential for nitrate-N reduction, 22% and 18% reductions, respectively. The potential for nitrate-N reductions for controlled drainage are limited by land area applicable for this practice (slopes less than 1%). Also, while nitrate-N concentration in water moving through the shallow groundwater below a buffer has been shown to be dramatically reduced (approximately 91%), the overall potential for nitrate-N load reduction by buffering all agricultural streams is limited (approximately 7%). This load reduction is limited by water interception and shallow groundwater movement below the buffer. Land use change also shows potential for nitrate-N reductions but the level of reduction will be dependent on the overall amount of land converted to a perennial based system or extended rotation.

A review of Table 4 shows no single practice would achieve nutrient reduction goals other than major land use changes. Instead, a combination of practices will be needed. There are endless combinations, but a few combined scenarios are highlighted in Table 5 that would reach goals for both nitrate-N and phosphorus. These represent a range of initial investments and annualized cost and benefits. Economic costs of these combination scenarios were computed considering the cost for implementing the practice and any potential impact on crop yield, specifically corn grain yield. An equal annualized cost (EAC) was computed so those practices with annualized costs and those with large initial capital costs could be appropriately compared. For the capital costs, a design life of 50 years and a discount rate of 4% was used. The price of corn was assumed to be $5/bushel and the cost of nitrogen was assumed to be $0.50/lb N. It is evident a range of scenarios are possible to achieve the nitrate-N and phosphorus reduction goals and that combinations of practices would be needed, with potential costs varying dramatically depending on which practices are implemented.

***Phosphorus***

The Iowa P Index is a quantitative assessment tool intended to assess risk of P loss from individual agricultural fields, allow for comparisons of conservation and P management practices in relation to potential P loss, and estimate P delivered to the nearest stream or water body. This model is comprehensive and estimates P loss, taking into account location in the state, soil type, soil test phosphorus, P application rate, tillage practices, source, timing and incorporation practices, runoff, erosion, and distance to the nearest stream or water body. To achieve the objectives of this effort, the science team adapted this tool to estimate P loads from MLRAs. To assess the impact of phosphorus reduction practice implementation, scenarios were developed within the P Index representing the number of acres being implemented with each practice or combination of practices. From this comparison, the estimate of potential P load reduction for each standalone practice or combination of practices was computed. It is important to note the computed reductions for standalone practices are not additive. In other words, it's not possible to add together reductions from multiple practices.

Alternatives for reducing P loading to receiving waters fall into three main groups: P management practices, edge-of-field and erosion control practices, and land use change. Phosphorus management practices focus on the most effective or efficient use of P, or those that otherwise reduce its availability for transport to receiving waters. As shown in Table 6, the P management strategies of cover crops (50% reduction) and conversion of all tillage to no-till (39% reduction) have the potential to substantially reduce P loss. Converting all acres of intensive tillage (<20% residue) to conservation tillage (>30% residue) would potentially reduce P loss by 11%. Injecting or banding of P within current no-till acres has little potential impact on P loss (<1%).

Edge-of-field technologies are designed primarily to settle sediment, or, in some cases, to retain dissolved P. These provide opportunities to remove P either in combination with the above practices or as stand-alone P reduction strategies. While the potential reduction of many erosion control practices could not be estimated due to lack of data, streamside buffers were estimated to have the potential to reduce P loss by 18%.

A third option is changing land use, with major focus on cropping systems that involve perennial vegetation cover or rotations of row crops with perennial forage crops for hay, pasture, or bioenergy production. As shown in Table 6, scenarios were developed that would change land use to perennial crops (energy crops), or pasture and land retirement equal to the acreage of pasture, hay, and Conservation Reserve Program land in 1987. Of these two scenarios, conversion to perennial energy crops would have the greatest potential to reduce P loss (29%). Doubling the amount of current extended rotation acres would have little potential impact on P loss (3%).

A review of Table 6 shows that only a few single practices would achieve P reduction goals without significant land use change. Instead, a combination of practices, likely in conjunction with N reduction practices, will be needed. As discussed above, these combinations are highlighted in Table 5.

## Future Needs

While significant research has been conducted on the potential performance of various nutrient reduction practices, there is a need for development of additional practices, testing of new practices, further testing of existing practices, and verifying practice performance at implementation scales. Many of the studies used in this evaluation were conducted at the plot scale. While these provide critical information and studies of this kind should continue, there also is a need for studies that scale up the area of practice implementation to better assess water quality impacts across landscapes and with multiple practices. Additional research also likely would improve the predictability of practice performance and improve the understanding of practice uncertainty.

In addition, to assess potential landscape-scale changes, there is a need for better tracking of practices currently in place, including but not limited to land use, crop rotations, nutrient applications, tillage, and conservation practices. In this analysis, the practices and existing conditions were aggregated on a MLRA scale, but actual implementation would be at a much finer scale. This highlights the need for actual practice information at the field level in order to better inform future assessments on potential gains or actual gains being made in achieving nitrogen and phosphorus nutrient reductions to surface waters.

**Table 2. Nitrogen reduction practices – potential impact on nitrate-N reduction and corn yield based on literature review.**

| | Practice | Comments | % Nitrate-N Reduction+ | % Corn Yield Change++ |
|---|---|---|---|---|
| | | | Average (SD*) | Average (SD*) |
| Nitrogen Management | Timing | Moving from Fall to Spring Pre-plant Application | 6 (25) | 4 (16) |
| | | Spring pre-plant/sidedress 40-60 split Compared to Fall Applied | 5 (28) | 10 (7) |
| | | Sidedress - Compared to Pre-plant Application | 7 (37) | 0 (3) |
| | | Sidedress – Soil Test Based Compared to Pre-plant | 4 (20) | 13 (22) |
| | Source | Liquid Swine Manure Compared to Spring Applied Fertilizer | 4 (11) | 0 (13) |
| | | Poultry Manure Compared to Spring Applied Fertilizer | -3 (20) | -2 (14) |
| | Nitrogen Application Rate | Reduce to Maximum Return to Nitrogen value 149 kg N/ha (133 lb N/ac) for CS and 213 kg N/ha (190 lb N/ac) for CC | 10‡ | -1‡‡ |
| | Nitrification Inhibitor | Nitrapyrin – Fall - Compared to Fall-Applied without Nitrapyrin | 9 (19) | 6 (22) |
| | Cover Crops | Rye | 31 (29) | -6 (7) |
| | | Oat | 28 (2)** | -5 (1) |
| | Living Mulches | e.g. Kura clover - Nitrate-N reduction from one site | 41 (16) | -9 (32) |
| Land Use | Perennial | Energy Crops Compared to Spring- Applied Fertilizer | 72 (23) | -100 ᵞ |
| | | Land Retirement (CRP) Compared to Spring- Applied Fertilizer | 85 (9) | -100 ᵞ |
| | Extended Rotations | At least 2 years of alfalfa in a 4 or 5 year rotation | 42 (12) | 7 (7) |
| | Grazed Pastures | No pertinent information from Iowa - Assume similar to CRP | 85*** | NA |
| Edge-of-Field | Drainage Water Mgmt. | No impact on concentration | 33 (32)^ | |
| | Shallow Drainage | No impact on concentration | 32 (15)^ | |
| | Wetlands | Targeted Water Quality | 52† | |
| | Bioreactors | | 43 (21) | |
| | Buffers | Only for water that interacts with active zone below the buffer - a small fraction of all water that makes it to a stream. | 91 (20) | |

+ A positive number is nitrate concentration or load reduction and a negative number is increased nitrate.
++ A positive corn yield change is increased yield and a negative number is decreased yield. Soybean yield is not included as the practices are not expected to affect soybean yield.
* SD = standard deviation.
‡ Reduction calculated based on initial application rate for each Major Land Resource Area (MLRA).
‡‡ Calculated based on the Maximum Return to Nitrogen (MRTN) relative yield at the given rates.
** Based on 1 study with 3 years of corn and 2 years of soybean.
*** This number is based on the Land Retirement number – there are no observations to develop a SD.
^ These numbers are based on load reduction since there is no impact on concentration with these practices
† Based on one report looking at multiple wetlands in Iowa (Helmers et al., 2008a).

**Table 3. Practices with the largest potential impact on phosphorus load reduction.**
**Notes: Corn yield impacts associated with each practice also are shown as some practices may be increase or decrease corn production. See text for information on value calculations.**

| | Practice | Comments | % Phosphorus Load Reduction[a] | % Corn Yield Change[b] |
|---|---|---|---|---|
| | | | Average (SD[c]) | Average (SD[c]) |
| Phosphorus Management Practices | Phosphorus Application | Applying P based on crop removal - Assuming optimal soil-test P level and P incorporation | 0.6[d] [70[e]] | 0[f] |
| | | Soil-Test P – Producer does not apply P until soil-test P drops to the optimal level | 17[g] [40[h]] | 0[f] |
| | | Site-specific P management | | 0[f] |
| | Source of Phosphorus | Liquid swine, dairy, and poultry manure compared to commercial fertilizer – Runoff shortly after application | 46 (45) | -1 (13) |
| | | Beef manure compared to commercial fertilizer – Runoff shortly after application | 46 (96) | |
| | Placement of Phosphorus | Broadcast incorporated within one week compared to no incorporation – Same tillage | 36 (27) | 0[f] |
| | | With Seed or knifed bands compared to surface application without incorporation | 24 (46) [35[i]] | 0[f] |
| Erosion Control and Land Use Change Practices | Tillage | Conservation till – chisel plowing compared to moldboard plowing | 33 (49) | 0 (6) |
| | | No till compared to chisel plowing | 90 (17) | -6 (8) |
| | Crop Choice | Extended rotation | [j] | 7 (7)[k] |
| | Perennial | Energy crops | 34 (34) | NA |
| | | Land retirement (CRP) | 75 | NA |
| | | Grazed pastures | 59 (42) | NA |
| | Terraces | | 77 (19) | |
| Edge-of-Field Practices | Wetlands | Targeted water quality | [l] | |
| | Buffers | | 58 (32) | |
| | Sediment Control | Sedimentation basins | 85 | |

a - A positive number is phosphorus reduction and a negative number is increased phosphorus.
b - A positive corn yield change is increased yield and a negative number is decreased yield. Practices are not expected to affect soybean yield.
c - SD = standard deviation.
d - Maximum and average estimated by comparing application of 200 and 125 kg $P_2O_5$/ha, respectively, to 58 kg $P_2O_5$/ha (corn-soybean rotation requirements) (Mallarino et al., 2002).
e - This represents the worst case scenario as data is based on runoff events 24 hours after P application. Maximum and average were estimated as application of 200 and 125 kg $P_2O_5$/ha, respectively, compared to 58 kg $P_2O_5$/ha (corn-soybean rotation requirements), considering results of two Iowa P rate studies (Allen and Mallarino, 2008; Tabbara, 2003).
f - Indicates no impact on yield should be observed.
g - Maximum and average estimates based on reducing the average STP (Bray-1) of the two highest counties in Iowa and the statewide average STP (Mallarino et al., 2011a), respectively to an optimum level of 20 ppm (Mallarino et al., 2002). Minimum value assumes soil is at the optimum level.
h - Estimates made from unpublished work by Mallarino (2011) in conjunction with the Iowa P Index and Mallarino and Prater (2007). These studies were conducted at several locations and over several years but may, or may not, represent conditions in all Iowa fields.
i - Numbers are from a report by (Dinnes, 2004) and are the author's professional judgment.
j - There is scarce water quality data for P loss on extended rotations in Iowa compared to a corn-soybean rotation.
k - This increase is only seen in the corn year of the rotation – one of five years.
l - Specific conditions are important in wetlands with regards to P as with changing inflow loads.

**Table 4. Example Statewide Results for Individual Practices at Estimated Nitrate-N Reduction.**
**Notes: Research indicates large variation in reductions not reflected in this table and some practices interact such that the reductions are not additive.**

| | Name | Practice/Scenario* | Nitrate-N Reduction % (from baseline) | Total Load (1,000 short ton) | N Reduced from baseline (1,000 short ton) |
|---|---|---|---|---|---|
| | BS | Baseline | | 307 | |
| Nitrogen Management | CCb | Cover crops (rye) on ALL CS and CC acres | 28 | 221 | 79 |
| | RR | Reducing nitrogen application rate from background to the MRTN 133 lb N/ac on CB and to 190 lb N/ac on CC (in MLRAs where rates are higher than this) | 9 | 279 | 28 |
| | CCa | Cover crops (rye) on all no-till acres | 6 | 288 | 18 |
| | SN | Sidedress all spring applied N | 4 | 295 | 12 |
| | NI | Using a nitrification inhibitor with all fall applied fertilizer | 1 | 305 | 2 |
| | FNb | Move all liquid swine manure and anhydrous to spring preplant | 0.3 | 306 | 1 |
| | FNa | Moving fall anhydrous fertilizer application to spring preplant | 0.1 | 307 | 0 |
| Edge-of-Field* | W | Installing wetlands to treat 45% of the rowcrop acres | 22 | 238 | 69 |
| | BR | Installing denitrification bioreactors on all tile drained acres | 18 | 252 | 55 |
| | CD | Installing Controlled Drainage on all applicable acres | 2 | 300 | 7 |
| | BF | Installing Buffers on all applicable lands | 7 | 284 | 23 |
| Land Use Changes | EC | Perennial crops (Energy crops) equal to pasture/hay acreage from 1987. Take acres proportionally from all row crop. This is in addition to current pasture. | 18 | 253 | 54 |
| | P/LR | Pasture and Land Retirement to equal acreage of Pasture/Hay and CRP from 1987 (in MLRAs where 1987 was higher than now). Take acres from row crops proportionally | 7 | 287 | 20 |
| | EXT | Doubling the amount of extended rotation acreage (removing from CS and CC proportionally) | 3 | 297 | 10 |

* These practices include substantial initial investment costs.

**Table 5. Example Statewide Combination Scenarios that Achieve Both the Targeted Nitrate-N and Phosphorous Reductions, Initial Investment and Estimated Equal Annualized Costs based on 21.009 Million Acres of Corn-Corn and Corn-Soybean Rotation.**
**Note: Research indicates large variation in reductions from practices that is not reflected in this table. Additional costs could be incurred for some of these scenarios due to industry costs or market impacts.**

| | | Nitrate-N | Phosphorus | Cost of N Reduction from baseline | Initial Investment | Total EAC* Cost | Statewide Average EAC Costs |
|---|---|---|---|---|---|---|---|
| **Name** | **Practice/Scenario**** | **% Reduction from baseline** [xx] | | **($/lb)** | **(million $)** | **(million $/year)** | **($/acre)** |
| NCS1 | Combined Scenario (MRTN Rate, 60% Acreage with Cover Crop, 27% of ag land treated with wetland and 60% of drained land has bioreactor) | 42 | 30 | 2.95 | 3,218 | 756 | 36 |
| NCS3 | Combined Scenario (MRTN Rate, 95% of acreage in all MLRAs with Cover Crops, 34% of ag land in MLRA 103 and 104 treated with wetland, and 5% land retirement in all MLRAs) | 42 | 50 | 4.67 | 1,222 | 1,214 | 58 |
| NCS8 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 70% of all tile drained acres treated with bioreactor, 70% of all applicable land has controlled drainage, 31.5% of ag land treated with a wetland, and 70% of all agricultural streams have a buffer) - Phosphorus reduction practices (phosphorus rate reduction on all ag land, Convert 90% of Conventional Tillage CS & CC acres to Conservation Till and Convert 10% of Non-No-till CS & CC ground to No-Till) | 42 | 29 | *** | 4,041 | 77 | 4 |

* EAC stands for Equal Annualized Cost (50 year life and 4% discount rate) and factors in the cost of any corn yield impact as well as the cost of physically implementing the practice. Average cost based on 21.009 million acres, costs will differ by region, farm and field.
** Scenarios that include wetlands, bioreactors, controlled drainage and buffers have substantial initial investment costs.
*** N practices and cost of N reduction are the same as NCS7 (Section 2.2). Reducing P application meets the P reduction goal and lowers the cost of the scenario.
xx Baseline load includes both point and nonpoint sources.

**Table 6. Example Statewide Results for Individual Practices at Estimated Phosphorous Reduction.**
**Notes: Research indicates large variation in reductions not reflected in this table and some practices interact such that the reductions are not additive.**

| | Name | Practice/Scenario | Phosphorus Reduction (% from baseline) | Total Load (1,000 short ton) | P Reduced from baseline (1000 Short ton) |
|---|---|---|---|---|---|
| | BS | Baseline | | 16.8 | |
| Phosphorus Management | CCa | Cover crops (rye) on all CS and CC acres | 50 | 8.3 | 8.5 |
| | Tnt | Convert all tillage to no-till | 39 | 10.3 | 6.5 |
| | Tct | Convert all intensive tillage to conservation tillage | 11 | 14.9 | 1.9 |
| | RR | P rate reduction in those MLRAs that have high to very high soil test P | 7 | 15.6 | 1.2 |
| | CCnt | Cover crops (rye) on all no-till acres | 4 | 16.1 | 0.7 |
| | IN | Injection within no-till acres | 0.3 | 16.8 | 0.05 |
| Edge-of-Field* | BF | Buffers (35 ft) on all crop land | 18 | 13.7 | 3.1 |
| Land Use Changes | EC | Perennial crops (Energy crops) equal to pasture/hay acreage from 1987. Take acres proportionally from all rowcrop. This is in addition to current pasture. | 29 | 11.9 | 4.9 |
| | P/LR | Pasture and Land Retirement to equal acreage of Pasture/Hay and CRP from 1987 (in MLRAs where 1987 was higher than now). Take acres from rowcrops proportionally | 9 | 15.3 | 1.5 |
| | EXT | Doubling the amount of extended rotation acreage (removing from CS and CC proportionally) | 3 | 16.3 | 0.5 |

* These practices include substantial initial investment costs.

# Section 2.2
# Iowa Science Assessment of Nonpoint Source Practices to Reduce Nitrogen Transport in the Mississippi River Basin

Prepared by the Nitrogen Science Team
May 2013

# Table of Contents

## Introduction

Nationally, the main reason for reducing nitrogen coming from agricultural regions of the Midwest is to reduce the size of the hypoxic zone in the Gulf of Mexico. The main emphasis is nitrate-N. Locally, nitrate-N levels also exceed the maximum contaminant level for drinking water of 10 mg N/L, resulting in increased water treatment costs in some cases and overall concern for aquatic ecosystems. Corn and soybean row crop production is extensive in Iowa, occupying the majority of agricultural managed land. Since the soil is an open system, that is, there is water drainage from the soil profile, and more rainfall is received than can be held within the soil profile, practices to lessen nitrate loss must work within these constraints. In addition, nitrogen can leave the land surface with runoff and erosion. Some of the practices discussed below will additionally have an impact on surface runoff and erosion, however, these were not addressed with this reduction effort.

In late 2010, the Iowa Department of Agriculture and Land Stewardship and the College of Agriculture and Life Sciences at Iowa State University partnered to develop a statewide nutrient reduction strategy for Iowa. Reducing nutrient loading to the Mississippi River is to be consistent with goals of a 45% reduction in riverine nitrogen and phosphorus transport. The science team working on this effort has 23 individuals representing five agencies or organizations. Within the overall team, sub-group science teams were formed to focus on nitrogen and phosphorus.

Included in this document are results from the nitrogen team. This work was focused on determining practices that would be expected to provide the greatest opportunity for reduction in nitrate-N export, and then estimating the potential for load reduction with practice implementation or combination of practice implementation. Since nitrogen export is primarily in the nitrate form, the work focused on nitrate-N reduction. The science team assembled a list of potential practices for greatest reductions, and the subgroup nitrogen team refined the list based on practices expected to have the greatest potential impact. The overall team then reviewed the list of practices and provided additional input.

Nitrate reduction practices being considered have a range of implementation and treatment scales. The primary reduction strategies fall into three main groups: nitrogen management, land use, and edge-of-field.

The nitrogen management practices focus on the most effective or efficient use of nitrogen, including nitrogen application timing (moving application from fall to spring); sidedressing nitrogen sometime after plant emergence (attempting to apply nitrogen closer to crop uptake); nitrogen source (commercial fertilizer, liquid swine manure, and poultry manure); nitrogen application rate; and a nitrification inhibitor (for fall-applied anhydrous ammonia); adding cover crops (cereal rye or oats) to row crop systems; and adding a living mulch to row crop systems (e.g. growing kura clover with continuous corn).

The land use options are intended to physically change the nitrogen dynamics by changing crops produced to varying degrees. These practices include moving to perennial crops used for energy production (e.g. switchgrass for ethanol); land retirement (e.g. CRP); converting row-crop land to pasture; and moving from a corn-soybean or continuous corn rotation to an extended four or five year rotation that includes multiple years of alfalfa.

Edge-of-field technologies provide opportunities to remove nitrate from water leaving production fields, either in combination with nitrogen management or land use practices or as standalone nitrate reduction systems. These practices include drainage water management (controlling tile water); shallow drainage (installing tile drains closer together but nearer the soil surface than conventional drainage); wetlands (targeted for water quality enhancement); denitrification bioreactors (treating tile-flow water from fields); and vegetated buffers along streams.

The list of specific nitrogen reduction practices could be very long when considering variations and combinations of practices. The following section outlines only those practices that have the potential to make a significant impact on reducing nitrate-N. Additionally, the practices are applicable to large portions of Iowa.

## Nitrogen Reduction Practices

After the science team determined the list of reduction practices, appropriate literature was assembled (see "Appendix A – Literature Reviewed") to determine the applicability of the practice and the likely benefit or detriment of implementation. Since this is a reduction effort focused on Iowa and conditions within the state, most of the studies selected for evaluation were conducted in or near Iowa. This was because a large portion of nitrate-N leaving the state is due to subsurface tile drainage, which typically has a region-specific influence due to differences in soils, climatic conditions, etc. One example is potentially long periods of wintertime frozen soil conditions in Iowa but open winter periods in other regions. However, if future precipitation amounts increase in Iowa, nitrogen export is likely to increase as well and it may be necessary to re-evaluate research from other regions.

The order of practices outlined in the text below or presented in **Table 1** does not represent a prioritized list. However, it is organized into nitrogen management, land use, and edge-of-field practices. There are wide performance ranges for all practices, which indicate spatial, temporal, and climactic influences, with those effects not directly considered here. In order to attempt to show the variability in practice performance, the minimum, maximum, and average (arithmetic mean) along with the standard deviation are given in **Table 1.** Large standard deviations indicate uncertainty, and when considering practices with single digit averages, may mean the practice will have little measureable impact on nitrate-N concentrations or reduction.

### Nitrogen Management

#### Timing

An estimated 12.9 million acres out of 50.6 million acres in the Midwest Corn Belt have fertilizer nitrogen applied in the fall (Randall and Sawyer, 2008). If this fractional estimate is applied to Iowa, approximately 3.12 million acres have fertilizer applied in the fall. The research summary showed there could be an average 6% reduction in nitrate-N concentration in tile drainage water when moving from fall to spring-applied nitrogen fertilizer, considering the same application rate. Any additional fertilizer application in the fall to compensate for anticipated losses is not accounted for here, but moving from fall to spring, in conjunction with a rate reduction, would be a larger benefit.

#### Sidedress

Sidedressing nitrogen can be done in different ways and with different sources of nitrogen, yet the concept of applying fertilizer after corn emergence is consistent. This strategy includes applying nitrogen during plant uptake, as well as timing to reduce the risk of loss from early spring rainfall/leaching events. The research summary showed an average 5% reduction in nitrate-N concentration in tile drainage water when moving from fall to spring/split-applied nitrogen fertilizer, and 4-7% reduction with sidedress compared to spring pre-plant, considering the same application rate. Sidedressing also allows the N rate to be optimized by either soil sampling or crop canopy sensing. For this reduction practice, sidedressing is considered only as early sidedress timing (corn height below 24-inch) or application based on soil nitrate sampling.

One note relative to the results shown in Table 1. The 13% yield increase for sidedress with soil testing should be viewed with some caution as the sidedress treatment from one of the main studies had 110 kg-

N/ha (95 lb-N/acre) for the preplant treatment but 123 kg-N/ha (110 lb-N/acre) to 225 kg-N/ha (200 lb-N/acre) for the sidedress with soil test treatment. As a result the corn yield impact may be due to nitrogen application rate differences. To date in Iowa, adjusting N rates with crop sensing has not been shown to be optimal as crop N deficiencies may not be detectable until mid-season and delaying N application in rain-fed corn does not always result in optimum yield or a water quality benefit. Thus, sidedressing with rates guided by crop sensing is not included in this practice. To confidently suggest all sidedressing practices for nitrate loss reduction, more research would be needed directly comparing the practices to pre-plant systems.

**Source**

Research suggests there is little, if any, difference in nitrate leaching or corn yield when using different sources of fertilizer nitrogen provided similar plant-available nitrogen application rates are used and management is appropriate for the source. Using slow or controlled-release fertilizer sources may have an impact on nitrate-N leaching, but no water quality data is available to quantify this and therefore those technologies are not included. The research summary indicated on average a small reduction (4%) in nitrate-N concentration when comparing liquid swine manure to fertilizer nitrogen, considering the same crop-available application rate. Besides potential impact on nitrate leaching, some manure sources high in solids content may have a positive impact on soil organic carbon, soil structure, and runoff.

**Nitrogen Application Rate**

Nitrogen rate is dynamic due to wide variation in potential nitrogen applications, including differences due to crop rotations and prices. However, rate has a predictable impact on nitrate-N concentrations leaving the crop root zone and in tile flow. The on-line Corn Nitrogen Rate Calculator tool is used in Iowa to determine the Maximum Return To Nitrogen (MRTN) for continuous corn and corn rotated with soybean, which provides the optimal rate based on the economic relationship between nitrogen cost and corn grain price. The Corn Nitrogen Rate Calculator also provides a profitable range around the MRTN which is within $1/acre net return of the MRTN. The MRTN and the most profitable range do provide an estimated statewide N fertilization rate needed for Iowa corn production.

**Nitrification Inhibitor**

Nitrification inhibitors slow the microbial conversion of ammonium-nitrogen to nitrate-N (nitrification). If more ammonium is present at the time of a loss event (leaching or denitrification), then more of the applied ammonium remains for crop use. This nitrification inhibitor practice specifically includes only nitrapyrin, the active ingredient in N-Serve®, and applied with fall anhydrous ammonia. For this practice, and in the literature reviewed, anhydrous was applied when soil temperatures were 10°C (50°F) and cooling and used other best practices for applying anhydrous ammonia. Nationally, research has found an average yield increase of 7% (Wolt, 2004) with use of nitrapyrin, but within and nearby Iowa yield benefits average 6% (with a standard deviation of 22%).

Nitrate-N loss benefits are mixed, but the average nitrate-N reduction from the research summary is 9% (with a standard deviation of 19%) when compared to fall-applied without an inhibitor. Nitrapyrin can also be used with spring applied anhydrous ammonia, but little relevant water quality data is available and research has not shown positive yield improvement. Due to limited data with use of nitrapyrin with other nitrogen fertilizers, or other products that slow nitrification, these were not included in this practice.

**Cover Crops**

The intent when using a cover crop is to reduce soil erosion and limit the amount of nitrate-N leaching from the system. Cover crops can be seeded in the fall using a variety of methods including drilling the seed after crop harvest, broadcasting the seed after crop harvest, or aerial broadcasting the seed before harvest.

Aerial application works best with cover crops that establish in a variety of conditions. Although there may be poor germination with aerial application, there is potential for extending the growing season of the cover crop with seeding before row crop harvest. This would enhance water quality benefits. Winter cover crops have the potential to reduce nitrate leaching in continuous corn and the corn-soybean rotation by taking up water and nitrate during the time between corn and soybean maturity and planting the next cover crop (Dabney et al., 2011; Kaspar and Singer, 2011). However, information about their effectiveness in reducing nitrate loss in Iowa and the upper Mississippi River basin is limited (Dabney et al., 2011; Dinnes et al., 2002).

Tonitto et al. (2006) in a meta-analysis of 69 studies from across the United States showed that non-leguminous cover crops reduced nitrate leaching losses by an average of 70%, and the amount of reduction was directly related to cover crop growth. In the upper Mississippi River basin, however, the potential cover crop growing season between harvest and planting corn and soybean is short and cold, and only cold-tolerant species like winter rye (Secale cereale L.) reliably produce substantial growth (Snapp et al., 2005). The research summary indicated an average 31% reduction in nitrate-N concentration with use of a rye cover crop and nearly that reduction for an oat cover crop. However, the oat cover crop data comes from only one study with three years of corn and two years of soybeans. Research suggests that when using a cereal rye cover before corn, the cover should be terminated 14 days before planting to limit negative impact on corn growth and yield. However, the research summary indicated an average 6% reduction in corn yield following a rye cover crop. There is no effect on soybean yield, so rye growth can continue longer in the spring and potentially provide more benefit in reducing nitrate-N loss. A slight corn yield reduction has been measured even when implementing oat as a cover crop. However, early planting in the fall is needed to realize any nitrate-N reduction, which is about half those compared to winter rye (due to oat kill by freezing temperatures).

### Living Mulches

A living mulch is a permanent land cover within a primary row crop, in this case corn. While some studies have had success growing row crops in a living mulch system, proper management involves a steep learning curve and has very specific requirements. In addition, there can be a year or two of living mulch establishment before a row crop can be planted. Average corn yield reduction for the area surrounding Iowa is only 9% based on the literature survey, but more localized research has shown 58% to 86% yield reductions. One of the main problems is the direct competition between the living mulch and the row crops, which includes row crop stand establishment and competition for water and nutrients. Nitrate reduction, however, can be large, with the research summary indicating an average 41% reduction in nitrate-N concentration. A benefit in addition to water quality is reduced soil erosion and enhanced soil physical structure.

## Land Use

### Perennial Crops (Energy Crops)

Energy crops are grown with the intention of using the biomass as a fuel feedstock. There are several methods for conversion of biomass into fuels, and there are multiple crops, which may be suitable as feedstock for specific processes. However, currently there are few markets for these products and those that exist are localized. With the current infrastructure and economic environment, there is likely to be limited implementation of perennial energy crops. There is substantial nitrate-N reduction potential, with the research summary indicating 72% nitrate-N reduction with conversion from row-crop production. Additional benefits include increased wildlife habitat, reduced soil erosion, and enhanced soil physical properties.

### Perennial Cover (CRP)

The Conservation Reserve Program (CRP) is a long-term (10-15 year) program intended to limit erosion and protect resources. Additionally, these systems are not fertilized and will, over time, substantially limit the amount of nitrogen leaving the area enrolled in the program. The research summary indicated an average 85% reduction in nitrate-N concentration with conversion to CRP from row-crop production.

### Extended Rotations

An extended rotation is a farming practice that includes a primary row crop of corn, and at least two years of a different crop that typically is a forage legume such as alfalfa. In practice, the specific rotation and crop combinations are extensive and may not be consistent on a given field. In this study, an extended rotation is defined as a corn-soybean-alfalfa-alfalfa rotation. Due to growing nitrogen fixing legumes three years in a row, very little, if any, nitrogen needs to be applied in the subsequent corn year. There is very little concurrent water quality and corn yield data for specific extended rotations. However, the research summary indicated an average 42% reduction in nitrate-N concentration in tile drainage water, with corn yields approximately 10% higher.

### Grazed Pastures

There are substantial areas of Iowa, especially southern Iowa, with pastureland. However, there was no pertinent data for nitrogen leaching from these systems in Iowa. Additionally, pastures can be grouped into several management schemes including intensively grazed, rotationally grazed, and grazed with cattle fenced off from the stream. As no relevant data was available, these systems were assumed to perform similar to the perennial crop (CRP) practice and have limited leaching and erosion. Based on the CRP practice, an average 85% reduction in nitrate-N concentration with conversion to grazed pasture from row crop production can be expected.

## Edge-of-Field

### Drainage Water Management

This practice consists of actively managing tile control structures that raise or lower the water table in a field. These systems have little, if any, impact on nitrate-N concentrations, but do reduce the amount of tile drainage water by an average of 33% (based on the literature survey for studies in and around Iowa) and therefore reduce nitrate load in tile drainage. They also have little or no effect on corn yield. Generally, water is released before planting and before harvest to allow for in-field traffic.

### Shallow Drainage

With this practice, subsurface tile drains are installed more closely together, but shallower than conventional tile drainage installation in Iowa, 0.75 m (2.5 ft) compared to 1.2 m (4 ft). As with drainage water management, corn yields and nitrate-N concentrations are not significantly affected, but tile drainage volume is reduced by an average of 32%, therefore reducing nitrate load. This practice would only apply to new tile drainage systems. One benefit of shallow drainage over drainage water management is that there is no need for annual or biannual management.

### Wetlands (Targeted for Water Quality)

Performance of installed wetlands is dependent on the wetland-to-watershed ratio, meaning how large is the wetland compared to the watershed area above the wetland. The larger the wetland, the greater the percentage of nitrate-N removal. From reported values from multiple wetlands in Iowa, the nitrate concentration reduction averages 52%. Many factors are involved with implementation of wetlands, including how much land is available and the nitrate-N influent concentration. To achieve the greatest nitrate reduction benefits, the wetlands need to be targeted to receive nitrate. The primary nitrate-N reduction wetland program in Iowa is the Conservation Reserve Enhancement Program (CREP), which has a limited, although growing, dataset. Wetlands restored specifically for habitat benefit are not being considered in this effort as they may or may not receive nitrate-N, and as a result, the primary water quality benefit is from land being taken out of production.

**Bioreactors**

Denitrification woodchip bioreactors are excavated pits filled with woodchips, with tile drainage water flowing through the woodchips. The intent is to pass water from the tile line into the bioreactor with denitrifying bacteria converting nitrate contained in the tile water into di-nitrogen gas. Bioreactors are intended to be implemented on a farm scale treating up to 100 acres of tile-drained land. Since bioreactors are relatively new, little research information from in and around Iowa is available. However, one study looking at four bioreactors in Iowa showed an average nitrate-N reduction of 43% for water going through the bioreactor. These systems can be designed with higher removal rates, up to maybe 50% of the nitrate-N load coming from a tile drainage system by maximizing retention time and minimizing by-pass flow. Like wetlands, the larger a bioreactor is, the more potential for nitrate-N reduction. However, there are concerns with over-designed systems as the denitrifying bacteria can produce methylmercury, which is highly toxic and can bioaccumulate in fish.

**Buffers**

Buffers along streams come in many sizes and shapes and can host a diverse plant population. Buffers additionally have habitat benefits, provide animal corridors, reduce sediment transport from fields, and stabilize stream banks. Only nitrate in water passing through the root zone of a buffer will be impacted by denitrification, therefore, the effect of buffers in tile-drained landscapes may be limited because only a small proportion of the total water yield passes through the root zone and tile flow is shunted through the buffer via the drainage pipe. However, the literature survey indicated an average nitrate-N concentration reduction of 91% for water actually passing through a buffer root zone. Many factors influence buffer performance including buffer width, vegetation type/age, and depth to the water table, yet nitrate-N removals are high in all situations.

## Nitrogen Reduction Practice Performance

The practices listed in **Table 1**, and associated nitrate reduction and corn yield change, were developed using several literature resources. For consistency, individual years of data (site years) were extracted from the reviewed documents to allow for direct comparisons. Large variations in nitrate reduction and yield effects were found for most practices, with the extreme minimum and maximum values also listed in **Table 1**. Average values in the table are not simply an average of the maximum and minimum, but are average values based on multiple observations. Specific methods for calculating the values are described below. **Great care was taken to insure correct comparisons were being made from each study.**

**Table 1. Practices with the largest potential impact on nitrate-N concentration reduction (except where noted). Corn yield impacts associated with each practice also are shown as some practices may be detrimental to corn production. See text on calculations for minimum, maximum, average, and standard deviation values for nitrate reduction and corn yield change.**

| | Practice | Comments | % Nitrate-N Reduction+ | | | % Corn Yield Change++ | | |
|---|---|---|---|---|---|---|---|---|
| | | | Min | Average (SD*) | Max | Min | Average (SD*) | Max |
| Nitrogen Management | Timing | Moving from Fall to Spring Pre-plant Application | -80 | 6 (25) | 43 | -16 | 4 (16) | 71 |
| | | Spring pre-plant/sidedress 40-60 split Compared to Fall Applied | -60 | 5 (28) | 33 | 2 | 10 (7) | 25 |
| | | Sidedress - Compared to Pre-plant Application | -95 | 7 (37) | 45 | -3 | 0 (3) | 5 |
| | | Sidedress - Soil Test Based Compared to Pre-plant | -29 | 4 (20) | 45 | -12 | 13 (22)** | 70 |
| | Source | Liquid Swine Manure Compared to Spring-Applied Fertilizer | -9 | 4 (11) | 25 | -17 | 0 (13) | 35 |
| | | Poultry Manure Compared to Spring Applied Fertilizer | -32 | -3 (20) | 21 | -33 | -2 (14) | 73 |
| | Nitrogen Application Rate | Reduce to Maximum Return to Nitrogen value 149 kg N/ha (133 lb N/ac) for CS and 213 kg N/ha (190 lb N/ac) for CC | 0 | 10‡ | 27 | 0 | -1‡‡ | -1 |
| | Nitrification Inhibitor | Nitrapyrin in Fall - Compared to Fall-Applied without Nitrapyrin | -33 | 9 (19) | 33 | -4 | 6 (22) | 104 |
| | Cover Crops | Rye | -10 | 31 (29) | 94 | -28 | -6 (7) | 5 |
| | | Oat | 26 | 28(2)*** | 30 | -6 | -5 (1) | -4 |
| | Living Mulches | e.g. Kura clover - Nitrate-N reduction from one site | 12 | 41 (16) | 53 | -86 | -9 (32) | 71 |
| Land Use | Perennial | Energy Crops - Compared to Spring-Applied Fertilizer | 26 | 72 (23) | 98 | | -100Ɣ | |
| | | Land Retirement (CRP) -Compared to Spring- Applied Fertilizer | 67 | 85 (9) | 98 | | -100Ɣ | |
| | Extended Rotations | At least 2 years of alfalfa in a 4 or 5 year rotation | 24 | 42 (12) | 62 | -27 | 7 (7) | 15 |
| | Grazed Pastures | No pertinent information from Iowa - assume similar to CRP | | 85**** | | | -100Ɣ | |
| Edge-of-Field | Drainage Water Mgmt. | No impact on concentration | -11 | 33 (32)^ | 98 | | | |
| | Shallow Drainage | No impact on concentration | 5 | 32 (15)^ | 54 | | | |
| | Wetlands | Targeted Water Quality | 11 | 52† | 92 | | | |
| | Bioreactors | | 12 | 43 (21) | 75 | | | |
| | Buffers | Only for water than interacts with the active zone below the buffer. This would only be a small fraction of all water that makes it to a stream | 33 | 91 (20) | 99 | | | |

+ A positive number is nitrate concentration or load reduction and a negative number is an increase.
++ A positive corn yield change is increased yield and a negative number is decreased yield. Soybean yield is not included as the practices are not expected to affect soybean yield.
* SD = standard deviation.

** This increase in crop yield should be viewed with caution as the sidedress treatment from one of the main studies had 110 kg-N/ha (95 lb-N/acre) for the preplant treatment but 123 kg-N/ha (110 lb-N/acre) to 225 kg-N/ha (200 lb-N/acre) for the sidedress with soil test treatment so the corn yield impact may be due to nitrogen application rate differences.
*** Based on 1 study with 3 years of corn and 2 years of soybean.
**** This number is based on the Land Retirement number – there are no observations to develop a SD.
‡ Reduction calculated based on initial application rate for each Major Land Resource Area (MLRA). Mean value is the statewide result while min and max values are based on individual MLRAs. Background application rates can be found in Table 12.
‡‡ Calculated based on the Maximum Return to Nitrogen (MRTN) relative yield at the given rates.
ɣ The number is -100, indicating a complete cropping change and therefore a corn yield of zero.
^ These numbers are based on load reduction since there is no impact on concentration with these practices.
† Based on one report looking at multiple wetlands in Iowa (Helmers et al., 2008a). The minimum and maximum are estimates from that report based on observations from CREP wetlands.

## Calculations for Practice Performance

The following methods were used to determine the minimum, mean, and maximum reduction in nitrate concentrations and the impacts on corn yield for each practice. These values were calculated using the same approach for most practices. However, for some practices the method was different, with those differences explained below. Nitrate-N concentrations were used rather than loads because tile, subsurface, and overland flow can vary across the state, which would have an impact on calculated load reductions. See "Appendix A – Literature Reviewed" for more details on specific research studies used for each practice.

Although only nitrate-N reductions are used here, some of the practices may have other benefits such as phosphorus and sediment reduction (cover crops), or aesthetic and wildlife benefits (wetlands and buffers). Any additional benefits were not included in the economic analysis.

### Nitrate-N Reduction Minimum and Maximum

Minimum and maximum values for the timing, source, nitrification inhibitor, energy crop, land retirement (CRP), cover crop, living mulch, extended rotation, bioreactors, and buffer practices were calculated based on individual site-years from each research study. For example, if there were 10 years of data for a potential reduction practice and the highest resulting nitrate-N concentration for one of the years was 5% higher than the corresponding controlled comparison (control) practice, the nitrate-N removal of that practice in that year would be -5% (or a 5% nitrate-N concentration increase). If the lowest concentration for one of the years was a nitrate-N concentration of 25% lower than the corresponding comparison practice, the nitrate-N removal of the potential reduction practice would be 25% (or 25% decrease in nitrate-N concentration). The standard deviations for each practice were also determined based on the site-year data.

### Nitrate-N Reduction Mean

The mean nitrate-N concentration reduction values were based on a corn-soybean rotation rather than individual crop years. In other words, the rotation concentrations resulting from the reduction practice were averaged, the result of which was divided by the average concentrations of the control practice and subtracted from 1. For example, assume there are 4 years of data for nitrogen application rate reduction in a corn-soybean rotation having a rotation average tile nitrate-N concentration of 2 for the first round of corn-soybean and 4 for the second round of corn-soybean. The comparison has 4 years of data at the "normal" nitrogen application rate with a nitrate-N concentration of 6 for the first round and 8 for the second round. The resulting mean tile flow nitrate-N reduction of the rotation due to reducing nitrogen application rate would be computed as in Equation 1.

**Equation 1**

$$mean = 1 - \frac{\left(\frac{2+4}{2}\right)}{\left(\frac{6+8}{2}\right)} = 0.57\ or\ 57\%$$

**Yield Calculations**

Corn yields for the practices are calculated the same way for minimum and maximum values, however, the comparison is change in yield. Here a negative change is reduced yield, and a positive change is increased yield. Mean yield change for a potential reduction practice from the comparison practice is calculated by averaging all observed yields in the potential reduction practice, subtracting average observed yield of the comparison practice, then dividing by the average observed yield of the comparison practice.

**Calculations Differing from Those Outlined Above**

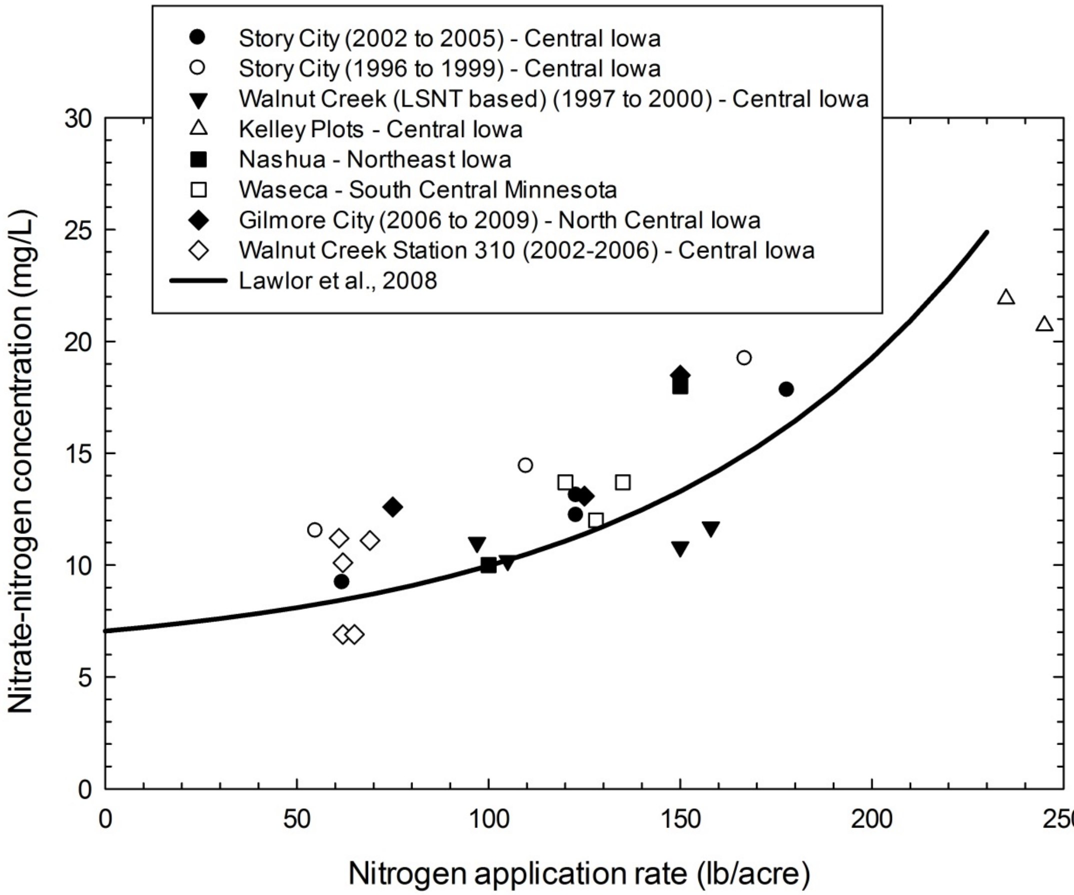
Reductions for other potential reduction practices required different approaches.

***Nitrogen Application Rate***

The nitrate-N concentration in tile flow water at a given fertilizer application rate was determined with an equation developed by Lawlor et al. (2008). Tile flow nitrate results from Lawlor et al. (2008) have been compared to other data from studies in Iowa and south-central Minnesota, and the data are in-line with the information from Lawlor et al. (2008) (Figure 1)

This data set was not adjusted for differences in rainfall, and, as mentioned earlier, long term increases or decreases in precipitation may influence this trend.

**Figure 1. Nitrogen application rate effect from various studies on tile drainage nitrate-N concentration for a corn-soybean rotation compared to the tile-flow response curve developed by Lawlor et al. (2008).**




***Pastures***

There was little pertinent data about nitrate-N concentrations coming from pastures in Iowa. The assumption was made that nitrate-N concentrations in water leaving the root zone are the same as for perennial energy crops.

***Drainage Water Management***

Drainage water management (controlled drainage) and shallow drainage have little, if any, impact on nitrate-N concentration. They do, however, reduce the amount of water leaving the system thus reducing the total nitrate-N load. In addition, there was little evidence that corn yield was significantly impacted by the practice. Minimum, maximum, and average load reductions are used instead of nitrate-N concentrations. The values used are site averages, and do not include analysis across site-years.

***Wetlands***

Wetlands are dynamic systems and nitrate-N concentration reduction is dependent on design. A nitrate-N removal of 52% was assigned to this practice based on an annual project report by Helmers et al. (2008a) where the average wetland is 0.785% of the contributing watershed. Ultimately, practice performance will depend on the size of the wetland.

### *Bioreactors*

Bioreactors also are heavily dependent on design, and could be sized to remove up to 50% or more of the nitrate load from a tile line. However, preliminary research in Iowa shows an average nitrate reduction of 43% from one study using the mean calculation procedure outlined above. These practices should have no impact on yield, as they are not installed in areas that would typically be farmed.

## Estimates of Potential Nitrate-N Load Reduction with Nitrogen Reduction Practices

There are three main sets of practices that can be considered for load reduction. One is the nitrogen input for corn production, with focus on nitrogen fertilization practices. A second is soil water management, with focus on retaining water in fields or removal of nitrate from water leaving fields. A third is changing land use, with focus on cropping systems that have less row crops and more crops or rotations with increased perenniality. In all practice options, the goal is to maintain nitrogen in soil with less conversion to nitrate and less movement with water from fields to surface water systems, especially during times of the year with greatest chance of loss. **No one practice alone will reduce nitrate-N levels in surface water systems to levels desired, such as a 45% reduction in waters leaving Iowa and moving to the Gulf of Mexico. It will take a suite of practices, and likely different practices in different areas of Iowa.**

This section describes the potential for reducing the loading of nitrate-N to Iowa surface waters using various standalone practices and a few combined practice scenarios. Included are economic assessments; potential for nitrate-N load reductions; practice limitations, concerns, or considerations; and other ecosystem services of a range of practices that have the potential for load reduction. The practices are grouped into nitrogen management practices, edge-of-field and land use practices. **For the combined practice scenarios, it must be noted these are not recommendations, but rather example scenarios.**

To estimate the baseline nitrate-N load, estimates of existing land use, literature estimates of nitrate-N concentrations in tile and subsurface water, and estimates of water yield to streams were used to compute a baseline load amount. For each standalone practice/scenario, the baseline nitrate-N concentrations were adjusted based on literature estimates for each practice and then used to compute a scenario load of nitrate-N, which was compared to the baseline load. From this comparison, the estimate of potential nitrate-N load reduction for each standalone practice or combination of practices was computed. **It is important to note the computed reductions for standalone practices are not additive, that is, it is not possible to add together reductions from multiple practices.**

Economic costs for each practice include estimates for implementing the practice at the field level and any potential impact on crop yield, specifically corn grain yield. An equal annualized cost (EAC) was computed so those practices with annualized costs and those with large initial capital costs could be appropriately compared. For the capital costs, a design life of 50 years and a discount rate of 4% were used. The price of corn was assumed to be $5/bushel and the cost of nitrogen was assumed to be $0.50/lb N. The price of corn and nitrogen is variable and higher or lower prices than used in this document would impact the cost estimates that are reported. **This document primarily includes farm level costs associated with the practices. It should be noted there could be additional costs and benefits for some of the practices or scenarios if implemented at a broad scale. These types of considerations are included in Section 2.4.**

Practice/scenario costs for implementation and potential for nitrate-N load reduction were calculated by Major Land Resource Area (MLRA), and then accumulated for a statewide cost and reduction amount. It is important to note that for any of the load estimates, there would be substantial uncertainty in the estimated load just based on uncertainty in performance in the nitrogen reduction practice. In addition, for nitrogen reduction practice, there would be a lag time from the time of practice implementation to the time water quality benefits are achieved. This analysis has not addressed the lag time associated with the practices, or the considerable time that might be needed to actually implement the practice or scenario.

## Background on Nitrate-N Load Estimation

### Agricultural Background Information for Iowa

The nitrogen science team also developed a spreadsheet-based nitrogen load model to estimate nitrate-N delivery to surface waters on a Major Land Resource Area (MLRA) basis. As part of this modeling effort, the current land use and nitrogen application rates were required so any water quality benefits from the addition of nitrate-N reduction strategies could be estimated.

Iowa is part of 10 MLRAs (Figure 2 and Table 2). Each has different characteristics of soils, landscape, precipitation, and temperature. The state was divided into these areas to distinguish between agricultural systems and reduction practices that may differ in benefit across the state.

**Figure 2. The 10 Major Land Resource Areas (MLRAs) in Iowa. Descriptions can be found in Table 2.**

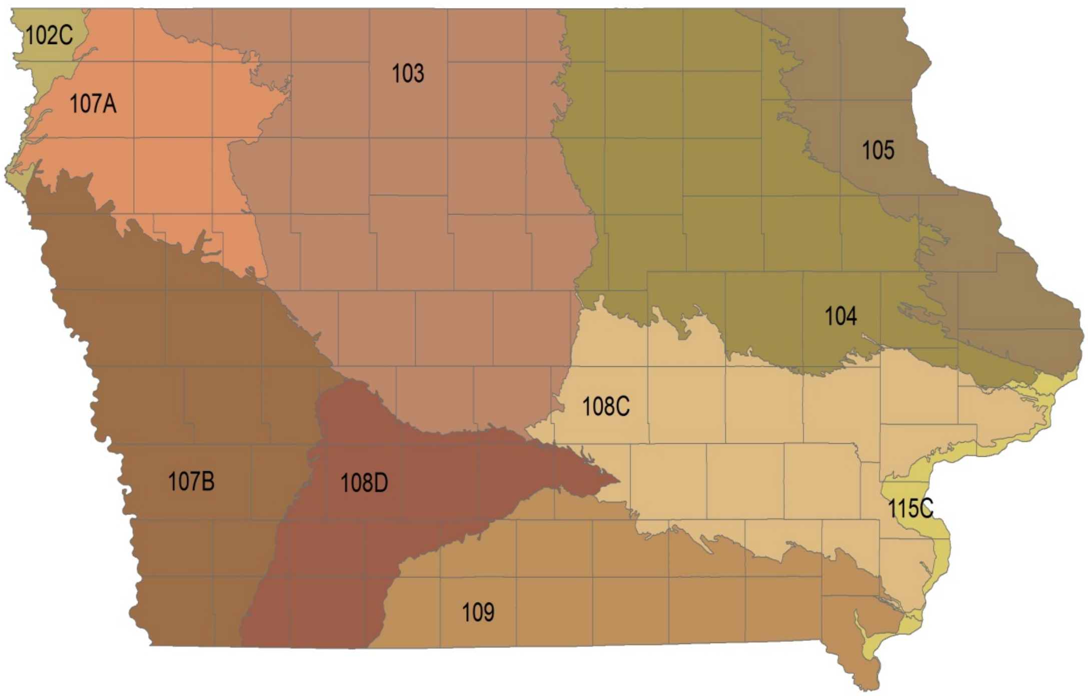


**Table 2. Brief description of the Major Land Resource Areas (MLRAs) in Iowa.**

| | | Landscape | | Climate | | |
|---|---|---|---|---|---|---|
| **MLRA** | **Description** | **Elevation m (ft)** | **Local Relief m (ft)** | **Total Precipitation mm (in)** | **Average Annual Temperature °C (°F)** | **Freeze Free days** |
| 102C | Loess Uplands | 335-610 (1,099-2,001) | 2-9 (7-30) | 585-760 (23-30) | 6-11 (43-52) | 170 |
| 103 | Central Iowa and Minnesota Till Prairies (aka. Des Moines Lobe) | 300-400 (984-1,312) | 3-6 (10-20) | 585-890 (23-35) | 6-10 (43-50) | 175 |
| 104 | Eastern Iowa and Minnesota Till Prairies | 300-400 (984-1,312) | 3-6 (10-20) | 735-940 (29-37) | 7-10 (45-50) | 180 |
| 105 | Northern Mississippi Valley Loess Hills | 200-400 (656-1,312) | 3-6 (10-20) | 760-965 (30-38) | 6-10 (43-50) | 175 |
| 107A | Iowa and Minnesota Loess Hills | 340-520 (1,115-1,706) | 3-30 (10-98) | 660-790 (26-31) | 7-9 (45-48) | 165 |
| 107B | Iowa and Missouri Deep Loess Hills | 185-475 (607-1,558) | 3-30 (10-98) | 660-1,040 (26-41) | 8-13 (46-55) | 190 |
| 108C | Illinois and Iowa Deep Loess and Drift – West-Central | 155-340 (509-1,115) | 3-6 (10-20) | 840-965 (33-38) | 8-11 (46-52) | 185 |
| 108D | Illinois and Iowa Deep Loess and Drift – Western | 210-460 (689-1,509) | 3-6 (10-20) | 840-940 (33-37) | 9-11 (48-52) | 185 |
| 109 | Iowa and Missouri Heavy Till Plain | 200-300 (656-984) | 3-6 (10-20) | 865-1,040 (34-41) | 9-12 (48-54) | 190 |
| 115C | Central Mississippi Valley Wooded Slopes - Northern | Similar to 108C | | | | |

As presented in the following discussion, a range of data was used to develop background information needed for reduction practices and reduction strategy comparisons. Although the years the data were drawn from may not be the same, an effort was made to represent the state as accurately as possible given the available data.

### *Crop Yield*

Total grain harvest (bushels) for both corn and soybean, and total harvested land (acres) for both corn and soybean for each MLRA, were determined by summing county estimates determined from the 2007 Agriculture Census (United States. National Agricultural Statistics Service, 2009). Data from counties that are split between MLRAs were partitioned based on the percent of the county in each MLRA (Equation 2). For example, 96% of Audubon County is in MLRA 107B, while the other 4% is in MLRA 108D. Corn grain harvested in 2007 in Audubon County was 18,088,508 bushels (459,477,045 kg). Splitting the grain between

MLRAs results in 17,364,968 bushels (441,097,963 kg) in MLRA 107B and 723,540 bushels (18,379,082 kg) in MLRA 108D.

**Equation 2**

$$Value_{MLRA} = \sum_{All\ Counties\ in\ MLRA} Value_{County} * \frac{\%County_{MLRA}}{100}$$

The number of harvested acres for each MLRA also was calculated with this equation. Once harvested grain and harvested area were summed for each MLRA, yield values were calculated (harvested grain/harvested area). Resulting yields are shown in Table 3.

**Table 3. Corn and soybean grain yields for each MLRA compiled from the 2007 Ag. Census.**

| MLRA | Corn Yield | | Soybean Yield | |
|---|---|---|---|---|
| | Mg/ha | bu/ac | Mg/ha | bu/ac |
| 102C | 10.0 | 159 | 3.6 | 53 |
| 103 | 10.7 | 170 | 3.4 | 50 |
| 104 | 10.7 | 171 | 3.4 | 51 |
| 105 | 10.7 | 170 | 3.4 | 50 |
| 107A | 9.9 | 158 | 3.4 | 51 |
| 107B | 9.6 | 153 | 3.3 | 49 |
| 108C | 10.9 | 173 | 3.4 | 51 |
| 108D | 9.4 | 150 | 3.3 | 49 |
| 109 | 9.6 | 153 | 3.2 | 47 |
| 115C | 11.0 | 176 | 3.3 | 49 |

Yield for corn in a continuous corn system was adjusted down while corn yield in a corn-soybean system was adjusted up to account for an approximate 8% yield reduction (Erickson, 2008) in a continuous corn system compared to corn in rotation with soybean (Table 4).

**Table 4. Corn yields in corn-soybean and a continuous corn for each MLRA compiled from the 2007 Ag. Census with rotation yield adjustments based on Erickson (2008).**

| MLRA | Corn Yield in Corn-Soybean | | Corn Yield in Continuous Corn | |
|---|---|---|---|---|
| | Mg/ha | bu/ac | Mg/ha | bu/ac |
| 102C | 10.2 | 163 | 9.4 | 150 |
| 103 | 11.0 | 175 | 10.1 | 161 |
| 104 | 11.0 | 176 | 10.2 | 162 |
| 105 | 11.2 | 179 | 10.4 | 165 |
| 107A | 10.1 | 161 | 9.3 | 148 |
| 107B | 9.8 | 156 | 9.0 | 143 |
| 108C | 11.1 | 177 | 10.2 | 163 |
| 108D | 9.5 | 151 | 8.7 | 139 |
| 109 | 9.7 | 155 | 9.0 | 143 |
| 115C | 11.4 | 181 | 10.5 | 167 |

### *Crop Areas*

Crop areas were determined from NASS crop layer data for 2006 – 2010 using GIS methods. A summary can be found in Table 5. A corn-soybean rotation is the dominant practice in the state as well as in each MLRA with the exception of MLRA 105 and 108D, where pasture and hay crop (PH) was the dominant practice.

**Table 5. MLRA crop areas for a corn-soybean rotation (CS), a continuous corn system (CC), various extended rotations (EXT), and a pasture and hay crop (PH).**

| MLRA | CS | CC | EXT | PH |
|---|---|---|---|---|
| | ha (ac) | ha (ac) | ha (ac) | ha (ac) |
| 102C | 68,860 (170,151) | 20,266 (50,077) | 7,357 (18,179) | 15,729 (38,866) |
| 103 | 1,917,134 (4,737,173) | 506,918 (1,252,577) | 77,125 (190,573) | 142,196 (351,362) |
| 104 | 1,293,724 (3,196,748) | 417,324 (1,031,193) | 111,299 (275,016) | 162,700 (402,026) |
| 105 | 154,347 (381,386) | 137,565 (339,918) | 81,381 (201,090) | 285,371 (705,142) |
| 107A | 742,064 (1,833,615) | 84,358 (208,446) | 38,529 (95,204) | 48,123 (118,910) |
| 107B | 1,189,034 (2,938,063) | 165,281 (408,404) | 113,560 (280,603) | 206,634 (510,586) |
| 108C | 865,024 (2,137,445) | 193,934 (479,204) | 125,678 (310,546) | 346,020 (855,004) |
| 108D | 388,642 (960,321) | 26,307 (65,004) | 80,779 (199,602) | 404,699 (999,998) |
| 109 | 235,615 (582,197) | 25,849 (63,872) | 81,675 (201,816) | 633,259 (1,564,762) |
| 115C | 51,711 (127,776) | 18,210 (44,996) | 8,168 (20,183) | 12,762 (31,534) |
| Iowa Total | 6,906,154 (17,064,873) | 1,596,013 (3,943,694) | 725,551 (1,792,812) | 2,257,495 (5,578,194) |

### *Hydrologic Characteristics*

Tile drained areas per MLRA were determined based on soil series identified as requiring drainage in the Iowa Drainage Guide and limited to slopes less than or equal to 2%. Drained land as % of row cropped land is shown in Table 6.

**Table 6. Estimated land with subsurface tile drainage as % of row cropped land for each MLRA in Iowa**

| MLRA | Drained Land (% Row Crop) |
|---|---|
| 102C | 20.9 |
| 103 | 66.8 |
| 104 | 32.2 |
| 105 | 16.6 |
| 107A | 38.7 |
| 107B | 24.9 |
| 108C | 42.1 |
| 108D | 36.1 |
| 109 | 69.8 |
| 115C | 71.7 |

The amount of tile drainage, along with land slope, soil type, and land use, impact the relationship between rainfall and water yield, meaning water leaving the landscape and flowing down streams and rivers. Total stream water yield used in this study was developed based on observed flow events in several watersheds and long-term precipitation.

**Table 7. Estimated total water yield from the MLRAs in Iowa. Based on discharge data from 38 gages in Iowa.**

| MLRA | Water Yield | |
|---|---|---|
| | mm/yr | in/yr |
| 102C | 139 | 5.5 |
| 103 | 263 | 10.4 |
| 104 | 302 | 11.9 |
| 105 | 286 | 11.3 |
| 107A | 187 | 7.4 |
| 107B | 208 | 8.2 |
| 108C | 284 | 11.2 |
| 108D | 250 | 9.8 |
| 109 | 305 | 12.0 |
| 115C | 285 | 11.2 |

***Nitrogen Application***

Nitrogen application rates for each MLRA were determined using Equation 2, which is the sum of the application per county in the MLRA. Rates for fertilizer and manure at the county scale were taken from David et al. (2010). Since that study was designed to look at a total nitrogen balance for regions in the state, manure numbers included all cattle (both grain-fed and pastured). Since manure from pastured cattle is not applied to production crops, these cattle were removed from this analysis, leaving only grain-fed cattle. Replacement cattle numbers came from the 2002 Census of Agriculture (United States. National Agricultural Statistics Service, 2007). Adjustments also were made to manure nitrogen amounts by adjusting for nitrogen availability as described below. The methods for fertilizer nitrogen application rates developed by David et al. (2010) used county level data from the 1997 and 2002 Census of Agriculture. The methods employed included distributing statewide fertilizer sales reported by the Association of American Plant Food Control Officials in 2008 to counties based on county-level fertilizer, lime, and soil conditioner expenditure for 1997 and 2002 reported by the Census of Agriculture.

Fertilizer application to turfgrass was estimated based on a method described by the Iowa Department of Natural Resources nutrient budget report Libra et al. (2004) and an EPA report suggesting approximately 9% of fertilizer nitrogen sold goes to turfgrass (Doering et al., 2011). Here, 9% of the statewide fertilizer nitrogen sales were proportioned to MLRAs based on the statewide percentage of urban area contained in each MLRA (Table 8). For example, MLRA 103, which contains Des Moines, makes up 24% of the urban area in the state meaning it would receive 24% of the turfgrass fertilizer.

**Table 8. Fertilizer nitrogen application to turfgrass based on % of urban area in each MLRA.**

| MLRA | Fertilizer to Turf grass | | Urban Area |
|---|---|---|---|
| | tonne | short ton | % of State Total |
| 102C | 756 | 833 | 1 |
| 103 | 19,445 | 21,434 | 24 |
| 104 | 14,743 | 16,251 | 18 |
| 105 | 4,623 | 5,096 | 6 |
| 107A | 5,933 | 6,540 | 7 |
| 107B | 11,025 | 12,153 | 14 |
| 108C | 11,476 | 12,650 | 14 |
| 108D | 5,304 | 5,847 | 7 |
| 109 | 5,409 | 5,962 | 7 |
| 115C | 1,654 | 1,823 | 2 |

The manure total nitrogen values from David et al. (2010) were adjusted for first-year crop availability based on the upper bounds reported in Sawyer and Mallarino (2008a) (Table 9). This adjustment was done so manure nitrogen could be combined with fertilizer nitrogen to establish total plant-available nitrogen application rates.

**Table 9. Manure total nitrogen available to the crop (as applied) in the year of application for MLRA total N partitioning.**

| Manure Source | Availability (%) |
|---|---|
| Cattle | 40 |
| Broilers | 60 |
| Layers | 60 |
| Turkey | 60 |
| Hog | 100 |

To more accurately account for commercial nitrogen fertilizer applied to corn, adjustment was made for estimates of nitrogen application to pasture and alfalfa hay, based on phosphorus use. This process involved using the total amount of nitrogen fertilizer after accounting for turfgrass application and allocating fertilizer to pasture at the Iowa State University recommendation rate on Bluegrass pasture, 90 kg/ha for single application to most of the state (Barnhart et al., 1997). Nitrogen application to pasture for each MLRA was calculated using Equation 3.

**Equation 3**

$$N_{MLRA_{Pasture}} = MLRA_{Pasture\ Area}\left(\frac{90\ kg\ N}{ha}\right)$$

Fertilizer nitrogen application to alfalfa was based on crop use of phosphorus, so nitrogen from monoammonium phosphate (MAP) and diammonium phosphate (DAP) was allocated to alfalfa based on phosphate removal of the crop, which was assumed to be 6.3 kg $P_2O_5$/tonne of alfalfa (12.5 lb $P_2O_5$/short ton) (Sawyer et al., 2011c) (Equation 4). It also was assumed the ratio of MAP sales to DAP sales was the same ratio as the MAP and DAP applied to alfalfa (based on $P_2O_5$ needs) (Equation 5). Statewide sales for MAP and DAP are from 1997 and 2002 as reported by the Iowa Department of Agriculture and Land Stewardship (IDALS, 2011) (Table 10). Total $P_2O_5$ was calculated based on $P_2O_5$ being 52% of MAP and 46% of DAP. Total nitrogen was calculated based on nitrogen being 11% of MAP and 18% of DAP (Equation 7 and Equation 8). A yield estimate of 9 tonnes/ha/yr (4 ton/acre/yr) was used for all alfalfa area in the state (Duffy, 2011).

Total $P_2O_5$ applied for each MLRA is effectively Equation 4.

**Equation 4**

$$P_2O_{5_{MLRA}} = MLRA_{Alfalfa\ Area}\left(\frac{9\ tonne_{Alfalfa}}{ha}\right)\left(\frac{6.3\ kg_{P_2O_5}}{tonne_{Alfalfa}}\right)$$

This total was used to estimate the contribution of both MAP and DAP to the $P_2O_5$ application in Equation 5 and Equation 6.

**Table 10. Monoammonium phosphate and diammonium phosphate sold in Iowa in 1997 and 2002 (Reported by IDALS Fertilizer Consumption).**

| Year | Product | Amount Sold | | Total Nitrogen | | Total $P_2O_5$ | |
|---|---|---|---|---|---|---|---|
| | | tonne | short ton | tonne | short ton | tonne | short ton |
| 1997 | MAP | 137,310 | 151,356 | 15,104 | 16,649 | 71,401 | 78,705 |
| | DAP | 353,800 | 389,991 | 63,684 | 70,198 | 162,748 | 179,396 |
| 2002 | MAP | 159,314 | 175,611 | 17,525 | 19,318 | 82,843 | 91,317 |
| | DAP | 336,045 | 370,420 | 60,488 | 66,675 | 154,581 | 170,394 |
| Average | MAP | 148,312 | 163,483 | 16,314 | 17,983 | 77,122 | 85,011 |
| | DAP | 344,922 | 380,205 | 62,086 | 68,437 | 158,664 | 174,894 |

**Equation 5**

$$P_2O_{5_{MAP\ MLRA}} = P_2O_{5_{MLRA}}\left(\frac{P_2O_{5_{MAP\ Sales}}}{P_2O_{5_{Total\ Sales}}}\right)$$

**Equation 6**

$$P_2O_{5_{DAP\ MLRA}} = P_2O_{5_{MLRA}} - P_2O_{5_{MAP}}$$

Using the percentage analysis of N and $P_2O_5$ in the MAP and DAP products, and the amount of $P_2O_5$ applied, the N application for each MLRA was calculated (Equation 7, Equation 8, and Equation 9)

**Equation 7**

$$N_{MAP} = \left(\frac{11\%\ N\ in\ MAP}{52\%\ P_2O_5\ in\ MAP}\right) P_2O_{5_{MLRA}}$$

**Equation 8**

$$N_{DAP} = \left(\frac{18\%\ N\ in\ DAP}{46\%\ P_2O_5\ in\ DAP}\right) P_2O_{5_{MLRA}}$$

**Equation 9**

$$N_{MLRA_{Alfalfa}} = N_{MAP} + N_{DAP}$$

Nitrogen (fertilizer nitrogen plus available manure nitrogen) application rate to corn for each MLRA was then calculated using Equation 10.

**Equation 10**

$$N_{MLRA_{Corn}} = \left(N_{MLRA_{Fertilizer}} - N_{MLRA_{Pasture}} - N_{MLRA_{Alfalfa}}\right) + N_{MLRA_{Manure}}$$

The purpose of the above calculations was to more accurately determine the fertilizer nitrogen application rate to corn since assuming all fertilizer nitrogen consumed was applied to corn would result in an overestimation of corn nitrogen application rates. Any overestimation of nitrogen application rates to corn would result in higher nitrate-N concentration estimates and would overestimate the impact of a nitrogen application rate reduction. Fertilizer, manure and total nitrogen calculated for each MLRA are shown in Table 11.

**Table 11. Nitrogen application rates to corn for each MLRA modified from David et al. (2010).**

| MLRA | Commercial Fertilizer | | Manure | | Total | |
|---|---|---|---|---|---|---|
| | kg N/ha | lb N/ac | kg N/ha | lb N/ac | kg N/ha | lb N/ac |
| 102C | 131 | 117 | 94 | 84 | 225 | 201 |
| 103 | 153 | 136 | 40 | 35 | 192 | 171 |
| 104 | 151 | 134 | 33 | 29 | 183 | 163 |
| 105 | 146 | 130 | 37 | 33 | 183 | 163 |
| 107A | 145 | 129 | 72 | 64 | 217 | 193 |
| 107B | 143 | 128 | 24 | 22 | 167 | 149 |
| 108C | 166 | 148 | 34 | 30 | 200 | 178 |
| 108D | 121 | 108 | 20 | 18 | 141 | 126 |
| 109 | 138 | 123 | 31 | 28 | 169 | 151 |
| 115C | 162 | 144 | 25 | 22 | 187 | 166 |
| Iowa Total | 149 | 133 | 37 | 33 | 186 | 166 |

These nitrogen application rates, although based on possibly outdated data, were used in conjunction with current crop area data (Table 5) to determine the total amount of nitrogen applied to corn (i.e. assume the application rates have not changed significantly since the data were collected). These nitrogen rates also were used to partition application to continuous corn and corn in a corn-soybean rotation by assuming continuous corn received 56 kg/ha (50 lbs/ac) (Blackmer et al., 1997; Sawyer et al., 2011c) more N than corn in a corn-soybean rotation. This assumption was made in the absence of actual application rate data for the corn-soybean rotation and continuous corn. Application rates for corn in a corn-soybean rotation were adjusted down to account for the increased rates on continuous corn, keeping total nitrogen applied constant. Table 12 provides the nitrogen application rates for each rotation. For comparison, nitrogen fertilizer (or crop available manure nitrogen equivalent) recommendations for corn in Iowa (Blackmer et al., 1997) range from 112 to 168 kg N/ha (100-150 lb N/acre) for corn in a corn-soybean rotation and from 168 to 224 kg N/ha (150-200 lb N/acre) for continuous corn; and from the Corn Nitrogen Rate Calculator (Sawyer et al., 2011b) at a nitrogen price of $0.50/lb N and a corn price of $5.00/bu, the range for corn-soybean is 136-164 kg N/ha (121-146 lb N/acre) and for continuous corn is 198-226 kg N/ha (177-202 lb N/acre). The calculated nitrogen application rates given in Table 12 show the state as a whole has nitrogen applied very close to the upper end of the profitable range as calculated by the Corn Nitrogen Rate Calculator.

**Table 12. Calculated nitrogen application rates to continuous corn and corn in a corn-soybean rotation.**

| | Total Nitrogen Applied | | Rate on CB | | Rate on CC | |
|---|---|---|---|---|---|---|
| **MLRA** | **tonne** | **short ton** | **kg N/ha** | **lb N/ac** | **kg N/ha** | **lb N/ac** |
| 102C | 12,300 | 13,558 | 204 | 182 | 260 | 232 |
| 103 | 281,502 | 310,298 | 173 | 154 | 229 | 204 |
| 104 | 194,785 | 214,710 | 161 | 144 | 217 | 194 |
| 105 | 39,195 | 43,204 | 147 | 131 | 203 | 181 |
| 107A | 98,606 | 108,693 | 206 | 184 | 262 | 234 |
| 107B | 127,240 | 140,256 | 155 | 139 | 211 | 189 |
| 108C | 124,996 | 137,782 | 182 | 163 | 238 | 213 |
| 108D | 31,058 | 34,235 | 134 | 120 | 190 | 170 |
| 109 | 24,319 | 26,806 | 159 | 142 | 215 | 192 |
| 115C | 8,223 | 9,064 | 163 | 146 | 220 | 196 |
| Iowa Total | 942,225 | 1,038,607 | 169 | 151 | 225 | 201 |

**Calculation of Baseline Nitrate-N Load**

Nitrate-N contribution was estimated as a function of land use and nitrogen application rates across Iowa on the basis of universal nitrogen curves (e.g. the Lawlor et al. curve in Figure 1) for continuous corn and corn-bean rotations that relate subsurface flow nitrate-N concentration to nitrogen application rate. Nitrate yield is the product of nitrate concentration and water yield. Water yield was generated on the basis of stream flow versus precipitation regressions developed for watersheds across Iowa. Daily precipitation data was downloaded from the National Climatic Data Center for the period 1980 through 2010. Data were obtained for 231 weather stations within Iowa and 127 stations in states surrounding Iowa within approximately 40 miles of Iowa. The data from these stations was approximately 30% incomplete. To complete the record for each station, missing daily values were estimated as the inverse distance weighted average of the 5 nearest stations having data on that day. These data were summed by year to obtain the total annual precipitation for each of the 358 weather stations. Discharge data were downloaded from the USGS Water Watch web pages for 38 gauge stations distributed across Iowa and annual water yields were calculated for each station for the period 1980 through 2010. The watershed boundary corresponding to each gauge station was determined and annual precipitation data for all weather stations within (and sometimes near) each watershed were averaged and used to represent the annual precipitation for each watershed. Examination of the relationship between annual water yield and precipitation suggested that most of the annual variation in water yield could be explained by precipitation in the current and preceding year (equation 11):

**Equation 11**

$$WY_{t,i} = \beta_{1,i}P_{t,i} + \beta_{2,i}P_{t,i}^2 + \beta_{3,i}P_{t-1,i} + \varepsilon_{t,i}$$

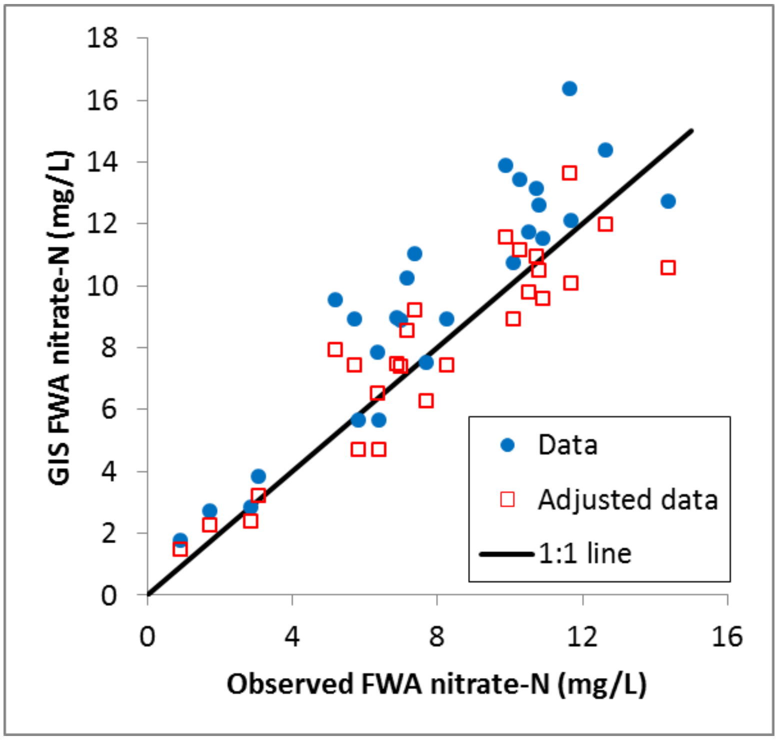
where $\beta_{1,i}$, $\beta_{2,i}$, and $\beta_{3,i}$ are regression coefficients for watershed *i*, (*i* = 1,…,38 watersheds), $P_{t,i}$ and $WY_{t,i}$ are the precipitation and water yield, respectively, for year *t* and watershed *i*, and $\varepsilon_{t,i}$ is the prediction error for year *t* and watershed *i*. Including the preceding year, year *t*-1, provides a surrogate for changes in groundwater storage whereby a wet prior year would likely result in a higher water table while a dry prior year would result in a lower water table in year *t*. Due to including the prior precipitation year, the 1980-2010 annual precipitation data can only predict the 1981-2010 annual water yields. The regression model $R^2$ for fitting these 30 years of water yield ranged from 0.617 to 0.934 across the 38 watersheds (average 0.845). All cases, including those with low $R^2$, had long term average accuracy within a few percent of the observed average. In most cases, all three regression coefficients were statistically significant at the 0.05 level of significance. In one case $\beta_1$ was not significant and in seven cases $\beta_3$ was not significant at the 0.05 level of significance but these were retained to maintain the same functional form across all watersheds. For a few combinations of very low precipitation in two consecutive years, equation 11 returned a negative value in which case the water yield was set to zero.

Equation 11 was applied to the annual precipitation data to generate an annual water yield estimate at each weather station location. To accomplish this, regression coefficients for each weather station were estimated as the inverse distance weighted values from the three nearest USGS watersheds using the distance from the approximate center of the watershed to the precipitation station. The regression coefficients for each weather station in conjunction with the station precipitation data were used to generate an annual water yield at each of the 358 weather stations for 1981 to 2010. These water yields corresponding to the 358 weather stations were used to generate a 300 m water yield grid for the state of Iowa for each year from 1981 to 2010 using the kriging procedure in ArcGIS. Because the work here is focused on long term performance, the 1981-2010 average water yield was used. This water yield map was utilized on an MLRA basis to estimate water yield for an individual MLRA.

Iowa STORET and USGS stream gauge data were assembled for 26 sample stations on Iowa rivers having at least 5 years of at least monthly nitrate concentration data and a flow-weighted-average (FWA) nitrate-N concentration was calculated for each site. For each of these watersheds, the GIS generated nitrate-N concentration based on land use and nitrogen application rate was compared with the observed FWA concentration. Based on these analyses, land use and nitrogen management explained most of the variability in nitrate concentration at larger watershed scales (Figure 3). Nitrate-N concentrations estimated based on land use and N application rates overestimated the observed nitrate concentrations by about 17% on the basis of a least-squares statistical analysis. Some overestimation was anticipated because the concentration based on N application is for subsurface water. Accordingly, this 17% difference could be largely explained by in-stream loss of nitrate and by dilution due to surface runoff and is consistent with both published and unpublished work. Nitrate concentration in stream flow is a function of contributions from subsurface flow (water that infiltrates the soil and then is either intercepted by a drainage tile or is returned to the surface drainage through other subsurface flow pathways) and surface runoff (overland flow that does not infiltrate into the soil, including rain water that is intercepted by a surface tile intake and delivered to the surface drainage by the tile system). Surface runoff generally has low nitrate concentration in tile drained landscapes of the Des Moines Lobe and thus surface runoff during rain events will dilute the higher in-stream concentrations generally observed between rain events.

For nitrate-N load calculations the surface runoff component of the water yield was estimated to be 17% and the remaining 83% was estimated to be subsurface flow. Estimates of the water yield (surface and subsurface) were combined with nitrate-N concentration estimates based on land use and nitrogen application to compute nitrate-N load. The surface runoff nitrate-N concentration was assumed to be negligible (<1 mg/L). The analysis summarized in subsequent sections of this document estimated nitrate-N load at the MLRA scale. For the baseline load scenario, estimates of existing practices on the MLRA scale including land use and nitrogen management were used to compute a baseline nitrate-N load that was used for comparison to the implementation scenarios.

**Figure 3. Observed FWA concentration versus GIS average nitrate concentration (solid blue circles) and GIS concentrations adjusted down by 17% to account for dilution from surface runoff and in-stream losses (open red squares) for 26 watersheds within Iowa (prediction efficiency = 82.5%).**




## Nitrogen Management Practices

### Move Fall Applied Nitrogen to Spring Preplant

This practice involves moving all of the current fall anhydrous ammonia and/or fall liquid swine manure application to the spring before planting.

#### *Practice limitations, concerns, or considerations*

- Infrastructure to support increased anhydrous ammonia use in the spring.
- Risk associated with applying fertilizer and manure in the spring due to limited number of days available for field work and possible yield reduction due to delayed fertilization/planting.
- With all liquid swine manure being applied in the spring, environmental concerns due to soil compaction, increase risk of runoff shortly following manure application, and increased risk of rapid movement to tile lines due to frequent wet soil conditions in the spring.

#### *Costs/benefits*

This practice is dynamic between MLRAs because the yield impact by moving from fall to spring varies by the different baseline corn yield in each MLRA. Although there may be a risk of not having enough suitable days to apply all nitrogen in the spring, this was not factored into the cost as the "value" of risk was not a component of this practice evaluation. This value could be included in future practice evaluations, with as an example by Hanna and Edwards (2007). The EAC values used for each MLRA (using baseline N application rates) are shown in Table 13.

**Table 13. Cost of moving all anhydrous ammonia and liquid swine manure from fall to spring, using baseline nitrogen application rates in each MLRA. Crop cost is only associated with any corn yield impact. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Timing Cost for Corn-Soybean (EAC) | Timing Cost for Continuous Corn (EAC) |
|---|---|---|
| | $/acre | $/acre |
| 102C | -16 | -33 |
| 103 | -18 | -35 |
| 104 | -18 | -35 |
| 105 | -18 | -35 |
| 107A | -16 | -33 |
| 107B | -16 | -32 |
| 108C | -18 | -36 |
| 108D | -16 | -31 |
| 109 | -16 | -32 |
| 115C | -18 | -36 |

### *Practice potential relative to nitrate-N load reduction*

#### *Scenario FNa: Move all fall anhydrous ammonia application to the spring*

All of the anhydrous applied in the fall is moved to spring application – MAP and DAP are not considered in this scenario and it is assumed no urea or urea-ammonium nitrate solution is fall applied as a primary nitrogen source for corn. It is estimated that currently approximately 25% of the total fertilizer nitrogen consumed in Iowa is applied in the fall as anhydrous ammonia. Any liquid swine manure application is left unchanged. Nitrogen application rates are not changed and a 4% yield increase occurs when applying nitrogen in the spring versus the fall, which was determined based on the literature (and included a range of nitrogen application rates). Any difference in cost of anhydrous ammonia purchased for application in the fall versus spring is assumed to be minor compared to current market fluctuations, therefore the price of nitrogen is not changed for fall vs. spring application. Although there could be substantial infrastructure costs with moving all anhydrous ammonia application to the spring, these costs are not considered. Moving all fall anhydrous ammonia to the spring is estimated to have the potential to reduce nitrate-N loading by 200 tons/year, which is about a 0.1% overall nitrate-N load reduction at an annual cost of approximately $-113,308,000/year (net economic benefit) (Table 14).

#### *Scenario FNb: Move all liquid swine manure and anhydrous ammonia applications to the spring*

With this scenario, the assumption is made that costs are the same as simply moving fall applied anhydrous ammonia fertilizer to the spring. Changes in infrastructure costs are not considered. It is estimated that nearly all the liquid swine manure is currently fall applied. Moving all fall applied liquid swine manure and fall anhydrous ammonia to the spring is estimated to have the potential to reduce nitrate-N loading by 1,000 tons/year which is about a 0.3% overall nitrate load reduction at an annual cost of approximately $-148,716,000/year (net economic benefit) (Table 14).

**Table 14. Example Statewide Results for Individual Practices at Estimated Maximum Potential Acres, Nitrate-N Reduction and Farm-Level Costs**

**Notes: Research indicates large variation in reductions not reflected in this table. Some practices interact such that the reductions are not additive.**

**Additional costs could be incurred for some of these scenarios due to industry costs or market impacts.**

**A positive $/lb N reduction, total cost or EAC is a cost. A negative $/lb N reduction, total cost or EAC is a benefit.**

| | Name | Practice/Scenario | Nitrate-N Reduction % (from baseline) | Potential Area Impacted for practice * (million acres) | Total Load (1,000 short ton) | Cost of N Reduction $/lb (from baseline) | Total Equal Annualized Cost (million $/year) | State Average EAC ** ($/acre) |
|---|---|---|---|---|---|---|---|---|
| | BS | Baseline *** | | | 307 | | | |
| Nitrogen Management | CCb | Cover crops (rye) on ALL CS and CC acres | 28 | 21.0 | 221 | 5.96 | 1,025 | 49 |
| Nitrogen Management | RR | Reducing nitrogen application rate from background to the MRTN 133 lb N/ac on CB and to 190 lb N/ac on CC (in MLRAs where rates are higher than this) | 9 | 18.9 | 279 | -0.58 | -32 | -2 |
| Nitrogen Management | CCa | Cover crops (rye) on all no-till acres | 6 | 5.1 | 288 | 5.97 | 227 | 45 |
| Nitrogen Management | SN | Sidedress all spring applied N | 4 | 13.5 | 295 | 0.00 | 0 | 0 |
| Nitrogen Management | NI | Using a nitrification inhibitor with all fall applied fertilizer | 1 | 2.2 | 305 | -1.53 | -6 | -3 |
| Nitrogen Management | FNb | Move all liquid swine manure and anhydrous to spring preplant | 0.3 | 7.3 | 306 | -74.36 | -149 | -20 |
| Nitrogen Management | FNa | Moving fall anhydrous fertilizer application to spring preplant | 0.1 | 5.7 | 307 | -283.27 | -113 | -20 |
| Edge-of-Field ***** | W | Installing wetlands to treat 45% of the ag acres | 22 | 12.8 | 238 | 1.38 | 191 | 15 |
| Edge-of-Field ***** | BR | Installing denitrification bioreactors on all tile drained acres | 18 | 9.9 | 252 | 0.92 | 101 | 10 |
| Edge-of-Field ***** | BF | Installing Buffers on all applicable lands **** | 7 | 0.4 | 284 | 1.91 | 88 | 231 |
| Edge-of-Field ***** | CD | Installing Controlled Drainage on all applicable acres | 2 | 1.8 | 300 | 1.29 | 18 | 10 |
| Land Use Changes | EC | Perennial crops (Energy crops) equal to pasture/hay acreage from 1987. Take acres proportionally from all row crop. This is in addition to current pasture. | 18 | 5.9 | 253 | 21.46 | 2,318 | 390 |
| Land Use Changes | P/LR | Pasture and Land Retirement to equal acreage from 1987 (in MLRAs where 1987 was higher than now). Take acres from row crops proportionally. | 7 | 1.9 | 287 | 9.12 | 365 | 192 |
| Land Use Changes | EXT | Doubling the amount of extended rotation acreage (removing from CS and CC proportionally). | 3 | 1.8 | 297 | 2.70 | 54 | 30 |

*Acres impacted include soybean acres in corn-soybean rotation as the practice has a benefit to water quality from the rotation.

**EAC stands for Equal Annualized Cost (50 year life and 4% discount rate) and factors in the cost of any corn yield impact as well as the cost of physically implementing the practice. Average cost based on 21.009 million acres, costs differ by region, farm, field.

***Baseline load includes both point and nonpoint source.

****Acres impacted for buffers are acres of buffers implemented and EAC are per acre of buffer.

*****These practices include substantial initial investment costs.

## Reducing Nitrogen Application Rate

This practice involves reducing the MLRA average nitrogen rate applied to corn to the Maximum Return to Nitrogen (MRTN) recommendation, the rate currently recommended in Iowa for continuous corn and corn following soybean.

### *Practice limitations, concerns, or considerations*

- Potentially negative impact on soil total nitrogen and soil organic matter if nitrogen application rates are too low and soil nitrogen is mined (Christianson et al., 2012), lowering soil quality over the long term.
- Risk of inadequate nitrogen for corn in high nitrogen responsive seasons.
- Not recognizing the uncertainty in nitrogen application requirements and impact on corn yield if nitrogen rate is too low.

### *Costs/benefits*

This practice utilizes the on-line Corn Nitrogen Rate Calculator (MTRN based recommendation system) (Sawyer et al., 2011b) to determine nitrogen rate impacts on fertilizer cost and yield return. Application rate is highly dynamic as any nitrogen application rate may be selected and each MLRA has different baseline application rates.

### *Other services – ecosystem or environmental*

- Since soil organic matter has a fairly constant ratio of carbon to nitrogen, the nitrogen input and removal balance associated with crop production can positively or negatively affect several soil properties associated with soil organic matter.

### *Practice potential relative to nitrate-N load reduction (Scenario RR)*

The maximum return to nitrogen (MRTN) application rate (based on assumed $5/bu corn and $0.50/lb nitrogen) for a corn-soybean rotation is 133 lb N/ac and 190 lb N/ac for continuous corn. **Of note, these MRTN values will vary based on corn and nitrogen prices, which is particularly important due to the variability in corn prices. As such, increases or decreases in corn prices without change in nitrogen price would increase or decrease the MRTN application rate, but rates will stay constant to those used within if the ratio of nitrogen-price-to-corn-price stays at 0.10.** No change was made for those MLRAs that have a lower nitrogen application rate than the MRTN (the rate was not increased to the MRTN level). Relative changes in yield with rate reduction were determined from the Corn Nitrogen Rate Calculator. Since the average application rate statewide is above the MRTN rate, there is not a direct cost associated with reducing the average application rate. However, there would be potential for increased risk of having inadequate nitrogen. Implementing the nitrogen rate reduction to the MRTN on all corn-soybean and continuous corn acres is estimated to have the potential to reduce nitrate-N loading by 28,000 tons/year, which is about a 9% overall load reduction at an annual cost of approximately $-32,308,000 (a net economic benefit) (Table 14). The Corn Nitrogen Rate Calculator (Sawyer et al., 2011b) has a profitable range ($1/acre net return) around the MRTN. This range for corn-soybean is 136-164 kg N/ha (121-146 lb N/acre) and for continuous corn is 198-226 kg N/ha (177-202 lb N/acre). When using the low end of the profitable range, the overall estimated nitrate-N load reduction is 15%, and when using the high end of the profitable range, the estimated load reduction is 4%.

## Sidedress All Spring Applied Nitrogen

***Practice limitations, concerns, or considerations***

Although producers make several trips with implements during the growing season, sidedressing nitrogen may add an additional operation as sometimes multiple activities are combined into one operation with preplant applications. There may be a need for investing in new equipment to make sidedress application possible, which could increase cost.

***Costs/benefits***

Since the number of field trips due to various field activities in the spring and early summer can vary depending on the year, producer, and crop, simply adding the cost of an additional operation for sidedressing was not possible. As a result, there was no cost associated with switching to a sidedress application and from Table 1 there was no corn yield benefit.

***Practice potential relative to load reduction (Scenario SN)***

Since most corn is fertilized (assume low acreage of corn that would not receive full nitrogen application), the cropland in the state that this practice would impact is 15.4 million acres. An additional assumption is that no producers are currently implementing this practice. There is currently some implementation of sidedress N application, but no data or levels of current implementation are available. Implementing sidedress nitrogen application on all corn-soybean and continuous corn acres receiving spring-applied nitrogen is estimated to have the potential to reduce nitrate-N loading by 12,000 tons/year which is about a 4% overall nitrate-N load reduction at an annual cost of approximately $0/year (Table 14).

## Using a Nitrification Inhibitor (Nitrapyrin) with All Fall Applied Anhydrous Ammonia

***Practice limitations, concerns, or considerations***

Use of nitrapyrin with all fall-applied anhydrous ammonia could have an impact on demand for the product, which could increase cost, but for this analysis it is assumed the cost of nitrapyrin would not change with increased use. Currently it is estimated that 2 million acres are receiving nitrapyrin in Iowa (Dow AgroSciences, 2012).

***Costs/benefits***

Research shows a corn yield increase and nitrate-N loss decrease when using nitrapyrin with fall applied anhydrous ammonia when compared to anhydrous ammonia applied at the same nitrogen rate without nitrapyrin. Because yield is impacted, the EAC for nitrapyrin application is different for each MLRA. Additionally, there is a product cost of approximately $11.50/acre (Sawyer, 2011). The following table gives the EAC when changes in corn yield are included in Table 14.

**Table 15. Cost of using nitrapyrin with fall anhydrous ammonia application, using baseline nitrogen application rates and current nitrapyrin use for each MLRA. Crop cost is only associated with any corn yield impact. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Nitrapyrin Cost for Corn-Soybean (EAC) | Nitrapyrin Cost for Continuous Corn (EAC) |
|---|---|---|
| | $/acre | $/acre |
| 102C | -20 | -39 |
| 103 | -21 | -43 |
| 104 | -22 | -43 |
| 105 | -21 | -43 |
| 107A | -20 | -39 |
| 107B | -19 | -37 |
| 108C | -22 | -44 |
| 108D | -18 | -36 |
| 109 | -19 | -37 |
| 115C | -22 | -45 |

***Practice potential relative to nitrate-N load reduction (Scenario NI)***

The primary assumption with this scenario is that nitrogen application rates and crop acres do not change from the baseline. Also assumed is that the nitrification inhibitor is applied with fall anhydrous at the appropriate rate and application is late fall with soil temperatures at 50°F and cooling. The only cost associated with this practice is the material, which is $11.50/acre. There is a corn yield increase of just over 6%. This scenario assumes there are currently 2 million acres receiving nitrapyrin in Iowa (Dow AgroSciences, 2012). Also, relative to the overall applicability of this practice, it is estimated that currently approximately 25% of the total fertilizer nitrogen consumed in Iowa is applied in the fall as anhydrous ammonia. The corn acres currently receiving nitrapyrin are proportionally split between the MLRAs based on how many corn acres are in the MLRA. Additionally, the acres for nitrapyrin use are partitioned to corn rotated with soybean and continuous corn based on the number of acres in each crop rotation. Table 16 shows the land area currently impacted by nitrapyrin application to corn. Nitrapyrin applied to corn rotated with soybean takes into account the impact of nitrapyrin across the two-year rotation, therefore the total number of acres exceed 2 million. Implementing use of a nitrification inhibitor with all fall applied anhydrous ammonia is estimated to have the potential to reduce nitrate-N loading by 2,000 tons/year, which is about a 1% overall nitrate load reduction, at an annual cost of approximately $-6,105,000 (net economic benefit) (Table 14).

**Table 16. Area estimated to currently receive nitrapyrin with fall applied anhydrous ammonia in Iowa. The total area is greater than the 2 million acre estimate because of the acres for soybean in the two-year corn-soybean rotation.**

| MLRA | Inhibitor applied to CS (acres) | Inhibitor applied to CC (acres) |
|---|---|---|
| 102C | 30578 | 6377 |
| 103 | 854007 | 153491 |
| 104 | 571117 | 135977 |
| 105 | 18497 | 73142 |
| 107A | 319757 | 20506 |
| 107B | 518258 | 41835 |
| 108C | 385020 | 55632 |
| 108D | 162955 | 5916 |
| 109 | 101322 | 6243 |
| 115C | 22616 | 6147 |

## Cover Crops

The cover crop in this practice/scenario is late summer or early fall seeded winter cereal rye. Winter rye offers benefits of easy establishment, seeding aerially or with drilling, growth in cool conditions and initial growth when planted in the fall, and continued growth in the spring.

### *Practice limitations, concerns, or considerations*

- Impact on seed industry due to increased demand for rye seed.
- Row crop out of production to meet rye seed demand.
- New markets for cover crop seed production.
- Economic opportunities for seeding a cover crop.
- Livestock grazing.
- Corn and soybean planting equipment designed to manage cover crops in no-till.
- Negative impact on corn grain yield.

### *Costs/benefits*

The winter rye cover crop practice is an annual cost with little to no capital investment. Items included in the annual cost are seed and seeding, and cover crop termination (chemically killed and/or plowed down). Seeding at a rate of 60 lb/acre and at a cost of $0.125/lb seed the total seed cost would be $7.50/acre per year (Singer, 2011). There were several cost sources for seeding using a no-till drill, which range from $8.40/acre (Duffy, 2011) to $15/acre (Singer, 2011), with Edwards et al. (2011) estimating $13.55/acre.

In order to grow the primary crop, the cover crop must be terminated (chemically killed and/or plowed down). Glyphosate is the primary herbicide used for this procedure, and Singer (2011) suggested use at 24 oz product/acre with a cost of $0.083/oz, or $2.00/acre. Additionally, there is a cost associated with hiring spray equipment between $6 to $8/acre (Edwards et al., 2011).

The base cost of this practice (before any corn yield impact) ranges from $29/acre to $32.50/acre per year (value of $32.5/acre used for cost analysis). Any cost associated with a corn yield reduction due to the preceding rye cover crop depends on the baseline corn yields in each MLRA. The cost of implementing a rye cover crop, including corn yield impact, is shown in Table 17. From the review of literature, the estimated yield impact for corn following rye is -6%. No yield impact occurs with soybean following a preceding rye cover crop, therefore, no soybean yield impact is included in the implementation cost.

**Table 17. Cost of using a rye cover crop. This cost is for operations, materials, and corn yield impact. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Cost of Implementing a Rye Cover Crop on Corn-Soybean Ground (EAC) | Cost of Implementing a Rye Cover Crop on Continuous Corn Ground (EAC) |
|---|---|---|
| | $/acre | $/acre |
| 102C | 40.5 | 83.5 |
| 103 | 42.5 | 86.5 |
| 104 | 42.5 | 87.5 |
| 105 | 42.5 | 86.5 |
| 107A | 40.5 | 83.5 |
| 107B | 39.5 | 81.5 |
| 108C | 43.5 | 87.5 |
| 108D | 39.5 | 80.5 |
| 109 | 40.5 | 81.5 |
| 115C | 43.5 | 88.5 |

***Other services – ecosystem or environmental***

- Wildlife habitat.
- Decreased erosion and loss of surface runoff contaminants (e.g. reduced phosphorus loss).
- Benefits to soil health and soil organic matter.

***Practice potential relative to nitrate-N load reduction***

*Scenario CCa: Plant a rye cover crop on all no-till acres*

The rationale for using this scenario is that farmers currently practicing no-till are more likely to implement cover crops and the lack of fall tillage is conducive to timely establishment of fall-planted cover crops. As no-till soybean is more common following corn, continuous corn is considered separately (Table 18). There is no assumption made about potential change in rye seed price or other establishment practices as rye cover crops are adopted. Also, there is no distinction made between fall and spring applied N. Implementing rye cover crops on the no-till acres is estimated to have the potential to reduce nitrate-N loading by 18,000 tons/year, which is about a 6% overall nitrate-N load reduction, with an annual cost of approximately $227 million/year (Table 14).

**Table 18. Distribution of tillage in each MLRA. Base data is from a Conservation Technology Information Center (CTIC) database.**

| | No-Till | Mulch Till | No-Till | Mulch Till |
|---|---|---|---|---|
| MLRA | % of CC | % of CC | % of CS | % of CS |
| 102C | 4 | 16 | 11 | 25 |
| 103 | 4 | 34 | 9 | 49 |
| 104 | 11 | 37 | 24 | 38 |
| 105 | 11 | 30 | 31 | 37 |
| 107A | 8 | 21 | 14 | 40 |
| 107B | 39 | 24 | 53 | 21 |
| 108C | 15 | 31 | 36 | 28 |
| 108D | 28 | 28 | 45 | 24 |
| 109 | 11 | 21 | 34 | 24 |
| 115C | 9 | 37 | 33 | 29 |

*Scenario CCb: Plant a rye cover crop on all corn-soybean and continuous corn acres*

The same assumptions apply to this cover crop scenario as for the no-till only scenario. Any economic difference between the scenarios is due to increased acres, differences in corn yields, and corn acres in each MLRA. Incorporation of cover crops would force major changes in the agronomic practices where fall tillage is used. Implementing rye cover crops on all corn following soybean and continuous corn acres is estimated to have the potential to reduce nitrate loading by 79,000 tons/year which is about a 26% overall nitrate-N load reduction, with an annual cost of approximately $1,025 million/year (Table 14).

## Edge-of-Field Practices

### Wetlands (Targeted for Water Quality)

***Practice limitations, concerns, or considerations***

- Contractor availability could limit rapid development of wetlands.
- Land availability – willing landowners to install wetlands.
- Limited landscape sites ideal for wetland installation.
- Increased costs for installation on non-ideal sites.

***Costs/benefits***

Wetland installation and maintenance cost estimates (from Christianson et al., In Preparation) include design cost, construction, seeding (buffer area around wetland), outflow structure, land acquisition, management (mowing), and control structure replacement. The example used in (Christianson et al., In Preparation) was based on a 10-acre wetland, with 35-acre buffer, treating 1,000 acres. The resulting EAC was $14.94/treated acre per year (net present value cost of $321/treated acre). They used a 4% discount rate and 50-year design life. (See Section 2.4 – Other Considerations Beyond Farm-Level Costs of Nutrient Reduction Practices.)

***Other services – ecosystem or environmental***

- Increased aesthetic landscape.
- Increased habitat for Iowa game and waterfowl.
- Depending on design, could provide hydrologic services through water flow attenuation.

***Practice potential relative to nitrate-N load reduction (Scenario W)***

*Installing wetlands to treat 45% of the ag land*

This scenario assumed 45% of the ag areas can be treated with wetlands. To achieve this large implementation, and on landscapes not easily suitable for wetlands, it would require complex and detailed design and enhanced installation for proper wetland performance. These wetlands, designed for water quality improvement, are assumed to receive water from all upland areas including tile drainage, percolation, and surface runoff. Impact on corn yield is assumed to be zero. For load reduction calculations, the area of the wetland is not subtracted from row crop land. However, land taken out of production is factored into the cost of the practice. Installing wetlands to treat 45% of the ag acres is estimated to have the potential to reduce nitrate-N loading by 69,000 tons/year, which is about a 22% overall nitrate-N load reduction at an annual cost of approximately $190,795,000 (Table 14). With wetlands, it may be possible to target the highest nitrate yielding areas of the landscapes and areas of the state in order to maximize overall nitrate-N reduction.

## Bioreactors

### *Practice limitations, concerns, or considerations*

- Limited to tile drained landscapes.
- Woodchip availability for the bioreactors.
- Increased cost of woodchips with installation of many bioreactors in a short period of time (100% implementation in a few years), or if all woodchips needed to be replaced at the same time.
- Additional industry (timber/woodchips) development due to demand.
- Contractor availability could limit rapid installations.

### *Costs/benefits*

Bioreactor installation and maintenance cost estimates (from Christianson et al., In Preparation) include control structures, woodchips, design, construction, seeding, additional tile, management, and maintenance. The example used in (Christianson et al., In Preperation) was based on a 0.25 acre bioreactor with a 50-acre treatment area. The resulting EAC was $10.23/ treated acre per year (net present value cost of $220/treated acre). (See Section 2.4 – Other Considerations Beyond Farm-Level Costs of Nutrient Reduction Practices.)

### *Practice potential relative to nitrate-N load reduction (Scenario BR)*

*Installing denitrification bioreactors on all tile drained cropland*

This scenario assumes denitrification woodchip bioreactors would be installed on 100% of the tile drained cropland. Estimates for tile drained cropland were developed from the USDA-ARS-NLAE and are shown in Table 19. The practice is assumed to have no impact on crop yield. The scenario does not account for land taken out of production for bioreactor installation as bioreactors can generally be installed in a non-cropland area. Additionally, there are no assumed costs associated with increased demand for woodchips or land use shifting to wood production because of the practice. Installing bioreactors to treat all tile drained cropland is estimated to have the potential to reduce nitrate-N loading by 55,000 tons/year, which is an 18% overall nitrate-N load reduction at an annual cost of approximately $101,481,000 (Table 14). In reality, it may not be feasible to treat all tile drainage water. It is important to recognize that the nitrate-N reductions from wetlands and bioreactors are not additive since they both may treat the same water. This would need to be considered in a statewide strategy that incorporates multiple practices.

**Table 19. Rowcrop land assumed tile drained based on soil type and slope class.**

| | **Drained Land** |
|---|---|
| **MLRA** | **% rowcrop** |
| 102C | 21 |
| 103 | 67 |
| 104 | 32 |
| 105 | 17 |
| 107A | 39 |
| 107B | 25 |
| 108C | 42 |
| 108D | 36 |
| 109 | 70 |
| 115C | 72 |

## Buffers

### *Practice limitations, concerns, or considerations*

Buffers have the potential to be implemented adjacent to streams to intercept shallow groundwater and reduce nitrate-N concentrations. While there could be broad implementation of this practice, the nitrate-N load reduction will be limited by the amount of shallow groundwater intercepted by the buffer.

### *Costs/benefits*

Costs of buffers can vary greatly depending on width, type of vegetation, and if substantial earthwork is required. For the analysis, a cost of establishment and implementation was assumed to be $300/acre with an EAC of $13.96/acre/year. In addition, there would be a cost of land out of production which was assumed to be equal to the average cash rent for corn and soybean land for each MLRA (Edwards and Johanns, 2011a; Edwards and Johanns, 2011b). From this, the EAC for buffer implementation by MLRA are as shown in Table 20.

**Table 20. Cost of implementing buffers (cash rent for corn and soybean cropland plus establishment EAC). (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Buffer Cost (EAC) |
|---|---|
| | $/acre |
| 102C | 234 |
| 103 | 237 |
| 104 | 241 |
| 105 | 228 |
| 107A | 246 |
| 107B | 238 |
| 108C | 228 |
| 108D | 217 |
| 109 | 188 |
| 115C | 222 |

### *Other services – ecosystem or environmental*

- Buffers would be expected to reduce sediment export and phosphorus export with surface runoff.
- Buffers would provide wildlife habitat benefits

### *Practice potential relative to nitrate-N load reduction (Scenario BF)*

*Installing buffers on all applicable acres*

Using a 35 ft wide buffer on each side of agricultural streams that are not currently buffered would add buffers on 44,768 miles of agricultural streams for a total buffer area of 380,000 acres. Installing buffers on all applicable cropland is estimated to have the potential to reduce nitrate-N loading by 23,000 tons/year, which is about a 7% overall nitrate-N load reduction at an annual cost of approximately $87,679,000/year (Table 14).

## Controlled Drainage

### *Practice limitations, concerns, or considerations*

Controlled drainage, also known as drainage water management (DWM), has limited applicability in Iowa due to the requirement of low slopes. This scenario considers controlled drainage, but drainage water management could also be achieved through shallower drain placement. However, shallower drain placement would have significant costs due to replacement of existing tile systems.

- Increased demand for control structures if short-term installation on all suitable area.
- Increased contractor costs associated with increased design and installation demand.

***Costs/benefits***

Controlled drainage and drainage water management installation and maintenance cost estimates (from Christianson et al., In Preparation) include structure cost (assumption of 20 acres per structure), system design, contractor installation, farmer management time (raise and lower control gate devices), structure replacement, and control device replacement. Resulting equal annualized cost was $9.86/acre per year.

***Other services – ecosystem or environmental***

- Managing the water table at a shallower depth could result in increased surface runoff, which would have implications for soil erosion and transport of other surface runoff contaminants (e.g. phosphorus).

***Practice potential relative to nitrate-N load reduction (Scenario CD)***

*Installing controlled drainage and drainage water management on all applicable acres*

The applicable cropland area was developed from the USDA-ARS-NLAE and is shown in Table 21 . Controlled drainage is limited to areas with land slopes less than 1% (Frankenberger et al., 2006). It is possible the land area considered suitable for controlled drainage is conservative since these estimates are based on soil maps; for example when the slope class is 0-2% it is assumed that an equivalent percentage of cropland has a slope from 0-1% slope and from 1-2% slope. Controlled drainage has little, if any, impact on nitrate-N concentration in tile flow; however, research suggests that water outflow is reduced by 33%. Also, little to no impact on crop yield is expected. Installing controlled drainage on all applicable cropland is estimated to have the potential to reduce nitrate-N loading by 7,000 tons/year, which is about a 2% overall nitrate-N load reduction at an annual cost of approximately $18,016,000 (Table 14).

**Table 21. Area suitable for controlled drainage and drainage water management.**

| | **Land Suitable for DWM** | |
|---|---|---|
| **MLRA** | **% rowcrop** | **% Drained Land** |
| 102C | 4 | 17 |
| 103 | 14 | 21 |
| 104 | 6 | 17 |
| 105 | 2 | 14 |
| 107A | 7 | 18 |
| 107B | 4 | 18 |
| 108C | 7 | 17 |
| 108D | 5 | 13 |
| 109 | 9 | 14 |
| 115C | 12 | 17 |

## Land Use Change Practices

### Grazed Pasture and Land Retirement Replacing Row Crops

#### *Practice limitations, concerns, or considerations*

- Market and price shifts due to reduced row crop production.
- New markets for grass-fed and organic beef.

#### *Costs/benefits*

The cost of switching land use from corn and soybean to pasture was calculated by subtracting the average cash rent received for pasture in each MLRA from the average cash rent for corn and soybean land (Edwards and Johanns, 2011a; Edwards and Johanns, 2011b). As there is limited data for both improved and unimproved pasture, the average cash rent of those two pasture categories was used for each MLRA. The resulting EACs for the practice implementation are shown in Table 22.

**Table 22. Cost of implementing pasture (cash rent for corn and soybean cropland minus cash rent for pasture land). (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Pasture Cost (EAC) |
|---|---|
| | $/acre |
| 102C | $150 |
| 103 | $169 |
| 104 | $171 |
| 105 | $159 |
| 107A | $173 |
| 107B | $159 |
| 108C | $159 |
| 108D | $148 |
| 109 | $122 |
| 115C | $145 |

Cost estimates for land retirement were based on income lost by taking land out of corn and soybean production (cash rent for corn and soybean) plus an annual maintenance cost. The maintenance was assumed to be mowing twice per year at a cost of $13.85/acre/mowing event ($27.70/acre/year) (Edwards et al., 2011). The EAC for each MLRA are shown in Table 23.

**Table 23. Cost of retiring corn and soybean row crop land. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Cost of Retiring Land (EAC) |
|---|---|
| | **$/acre** |
| 102C | 248 |
| 103 | 251 |
| 104 | 254 |
| 105 | 242 |
| 107A | 260 |
| 107B | 251 |
| 108C | 241 |
| 108D | 231 |
| 109 | 202 |
| 115C | 236 |

***Other services – ecosystem or environmental***

- Increase wildlife habitat.
- Decrease soil erosion, surface runoff, and surface runoff transported pollutant export (e.g. P).
- Provide hydrologic services, that is, reduction of water runoff amount and rate.
- Increase carbon sequestration.
- Reduce greenhouse gas emissions.

***Practice potential relative to nitrate-N load reduction***

*Scenario P/LR: Pasture and Land Retirement to equal pasture/hay and CRP acreage from 1987 (in MLRAs where 1987 acreage was higher than current). Row crop acres were reduced proportionally for corn-soybean rotation and continuous corn.*

This scenario increases the acreage of pasture and CRP to equal the pasture/hay and CRP acreage in 1987, which was the first time land was enrolled in CRP. Also, this scenario might be potentially obtainable as a viable alternative to row crop production. Some of the MLRAs have more land in pasture/hay and CRP land now than in 1987, but the current amount was not adjusted down to the 1987 level. Research suggests that pasture/hay and CRP reduces nitrate-N loss by at least 85% when compared to any land in corn or soybean. Statewide, this scenario impacts 1.9 million acres. Converting that amount of land from row crops to pasture/hay and CRP (approximate 9% reduction in row crops) is estimated to have the potential to reduce nitrate-N loading by 20,000 tons/year which is a 7% overall nitrate-N load reduction at an annual cost of approximately $364,631,000 (Table 14).

## Perennial Crops (Energy Crops) Replacing Row Crops

***Practice limitations, concerns, or considerations***

- Immediate limited market for perennials as energy crops.
- Market shifts in crop prices and demand.

***Costs/benefits***

Although there is not a current large market for perennial biomass crops as a source for energy or transportation fuel production, there are local and regional markets for those crops with current prices (example $50/ton). A publication from 2008 in the Ag Decision Maker series (Duffy, 2008) had estimates on the cost of production, transportation, and storage of switchgrass. At an assumed 4 ton/acre production level, the resulting revenue is $200/acre. The $50/ton does not cover the cost to harvest, store, and transport, thus, land retirement is more profitable. The Ag Decision Maker costs factor in a land charge, and land rent for corn and soybean was used to represent the cost of switching from row crops to perennials. Since land rent is different in each MLRA, the resulting cost of producing energy crops varies by MLRA (Table 24).

**Table 24. Cost of producing a perennial energy crop, assuming 4 ton/acre production level and a sales price of $50/ton. (Note: A positive EAC is a cost. A negative EAC is a benefit. Costs include cost of production, transportation, storage, land rent, and estimated returns)**

| MLRA | Cost of Producing Energy Crops (EAC) |
|---|---|
| | $/acre |
| 102C | 399 |
| 103 | 402 |
| 104 | 405 |
| 105 | 392 |
| 107A | 411 |
| 107B | 402 |
| 108C | 392 |
| 108D | 382 |
| 109 | 353 |
| 115C | 387 |

***Other services – ecosystem or environmental***

- Increase wildlife habitat.
- Decrease erosion, surface runoff, and surface runoff transported pollutant export (e.g. phosphorus).
- Provide hydrologic services, that is, reduction of water runoff amount and rate.
- Increased agricultural/economic diversity.

***Practice potential relative to nitrate-N load reduction***

*Scenario EC: Perennial crops (energy crops) to equal pasture/hay acreage in 1987.*

This scenario switches corn and soybean row crop land to energy crops at the amount equivalent to reach the total number of acres in pasture/hay in 1987 for each MLRA (Table 25). Row crop acres were reduced proportionally for the corn-soybean rotation and continuous corn. This scenario is estimated to have the potential to reduce nitrate-N loading by 54,000 tons/year, which is a 18% overall nitrate-N load reduction at an annual cost of approximately $2,317,734,000 (Table 14).

**Table 25. Land area converted from corn and soybean to energy crops to reach the 1987 acres in pasture/hay for each MRLA.**

| MLRA | % of MLRA converted to energy crops | Acres converted to energy crops |
|---|---|---|
| 102C | 12 | 41,537 |
| 103 | 6 | 502,181 |
| 104 | 14 | 818,917 |
| 105 | 35 | 907,608 |
| 107A | 11 | 285,877 |
| 107B | 14 | 714,923 |
| 108C | 18 | 894,591 |
| 108D | 31 | 871,829 |
| 109 | 38 | 1,363,425 |
| 115C | 13 | 60,695 |

## Extended Rotation (corn-soybean-alfalfa-alfalfa-alfalfa)

For this analysis the extended rotation was assumed to be corn followed by soybean followed by three years of alfalfa.

### *Practice limitations, concerns, or considerations*

- Reduced the amount of corn and soybean produced in Iowa.
- Market shift in product production (more alfalfa) and associated price for crops produced.
- Increased livestock production to feed alfalfa.
- Market shift as little fertilizer nitrogen is needed for the corn following alfalfa.

### *Costs/benefits*

As done with other practice costs related to perennial crops, the cost of the extended rotation is based on applicable cash rent values for each crop (Ag Decision Maker series, Duffy, 2008). The calculation shown is used in Equation 12.

**Equation 12**

$$\frac{3\ alfalfa\ years * \left(Cash\ Rent_{corn-soybean} - Cash\ Rent_{Alfalfa\ \ Hay}\right)}{5\ year\ total\ rotation}$$

This gives a range of \$0/ac to \$65/acre cost across the MLRAs and a state average of \$35/acre before accounting for a corn yield improvement of 7% for the extended rotation. The resulting costs, after the corn yield improvement, are shown in Table 26.

**Table 26. The EAC cost of the extended rotation in each MLRA. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Extended Rotation Cost (EAC) | Extended Rotation Cost Including Increased Corn Yield (EAC) |
|---|---|---|
| | $/acre | $/acre |
| 102C | $0 | -$12 |
| 103 | $42 | $30 |
| 104 | $33 | $21 |
| 105 | $19 | $6 |
| 107A | $17 | $5 |
| 107B | $53 | $42 |
| 108C | $47 | $34 |
| 108D | $65 | $54 |
| 109 | $50 | $38 |
| 115C | $29 | $16 |

***Other services – ecosystem or environmental***

- Increased wildlife habitat.
- Decrease erosion, surface runoff, and surface runoff transported pollutant export (e.g. phosphorus).
- Provide hydrologic services, that is, reduction of water runoff amount and rate when land is in alfalfa.
- Benefits to soil health and soil organic matter.

***Practice potential relative to nitrate-N load reduction***

*Scenario EXT: Doubling the current amount of extended rotation acreage.*

Increasing the acreage of extended rotations by doubling the current amount of extended rotations (and reducing proportionally the corn-soybean rotation and continuous corn) in each MLRA (Table 27) is estimated to have the potential to reduce nitrate-N loading by 10,000 tons/year which is a 3% overall nitrate-N load reduction at an annual cost of approximately $54,081,000 (Table 14).

**Table 27. Current extended rotation amount in each MLRA and the percent of land diverted from corn-soybean rotation and continuous corn for doubling the amount of extended rotation (EXT).**

| MLRA | % of Rowcrop (current) | % of Rowcrop diverted to EXT from CS | % of Rowcrop diverted to EXT from CC |
|---|---|---|---|
| 102C | 8 | 6 | 2 |
| 103 | 3 | 2 | 1 |
| 104 | 6 | 5 | 1 |
| 105 | 22 | 12 | 10 |
| 107A | 4 | 4 | 0 |
| 107B | 8 | 7 | 1 |
| 108C | 11 | 9 | 2 |
| 108D | 16 | 15 | 1 |
| 109 | 24 | 21 | 2 |
| 115C | 10 | 8 | 3 |

## Combined Scenarios for Nitrate-N Load Reduction

As evident by results presented in Table 14, no one practice will achieve the needed reductions without major land use changes. As a result, a combination of practices will be needed. The combinations could be endless but a few combined scenarios are highlighted below. Based on Iowa DNR estimates, nonpoint source load reductions would need to achieve 41% of the overall 45% load reduction in nitrate-N with the remaining 4% load reduction coming from point sources. The potential phosphorus reduction associated with these combined scenarios also was calculated (additional discussion of procedures used for calculating phosphorus load reduction is provided in the phosphorus strategies document). Based on Iowa DNR estimates, nonpoint source load reductions would need to achieve 29% of the overall 45% load reduction in phosphorus with the remaining 16% load reduction coming from point sources. **These combined scenarios should not be viewed as recommendations, but rather example combinations of practices that have the potential to reduce nitrate-N load reduction. Actual implementation is likely to include combinations beyond those presented here.**

### *Scenario NCS1*

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, 60% of corn-soybean and continuous corn acres have cover crops in all MLRAs, 27% of all ag land is treated with a wetland, and 60% of the tile drained acres are treated with a bioreactor. This scenario is estimated to have the potential to reduce nitrate-N loading by 125,000 tons/year which is approximately a 42% overall nitrate-N load reduction at an annual cost of approximately $755,518,000 (Table 28).

### *Scenario NCS2*

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, 100% of corn-soybean and continuous corn acres have cover crops in all MLRAs except 103 and 104, 43% of all ag land in MLRAs 103 and 104 are treated with a wetland, and 95% of the tile drained acres in MLRAs 103 and 104 are treated with a bioreactor. Since MLRAs 103 and 104 have a fairly low level of no-till adoption, which makes cover crops more conducive, we assumed there might be greater difficulty getting high levels of cover crop adoption in these areas. This scenario is estimated to have the potential to reduce nitrate-N loading by 121,000 tons/year which is approximately a 39% overall nitrate-N load reduction at an annual cost of approximately $631,475,000 (Table 28).

### *Scenario NCS3*

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, 95% of corn-soybean and continuous corn acres have cover crops, 34% of all ag land in MLRAs 103 and 104 are treated with a wetland, and 5% of all corn-soybean and continuous corn acres are converted to hay, pasture, or CRP. This scenario is estimated to have the potential to reduce nitrate-N loading by 129,000 tons/year which is approximately a 42% overall nitrate-N load reduction at an annual cost of approximately $1,213,617,000 (Table 28).

### *Scenario NCS4*

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, nitrification inhibitor used with all commercial fall applied nitrogen, sidedress all spring applied nitrogen, 38.25% of all ag land is treated with a wetland, 85% of the tile drained acres are treated with a bioreactor, and 85% of all applicable acres have controlled drainage. This scenario is estimated to have the potential to reduce nitrate-N loading by 128,000 tons/year which is approximately a 42% overall nitrate-N load reduction at an annual cost of approximately $225,469,000 (Table 28).

***Scenario NCS5***

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, nitrification inhibitor used with all commercial fall applied nitrogen, sidedress all spring applied nitrogen, 29.25% of all ag land is treated with a wetland, 65% of the tile drained acres are treated with a bioreactor, 65% of all applicable acres have controlled drainage, and 15% of corn-soybean and continuous corn acres are converted to energy crop (perennial based) production. This scenario is estimated to have the potential to reduce nitrate-N loading by 127,000 tons/year which is approximately a 41% overall nitrate-N load reduction at an annual cost of approximately $1,417,782,000 (Table 28).

***Scenario NCS6***

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, 25% of corn-soybean and continuous corn acres have cover crops in all MLRAs, 25% of corn-soybean and continuous corn acres are converted to extended rotations in all MLRAs, 27% of all ag land is treated with a wetland, and 60% of the tile drained acres are treated with a bioreactor. This scenario is estimated to have the potential to reduce nitrate-N loading by 126,000 tons/year which is approximately a 41% overall nitrate-N load reduction at an annual cost of approximately $541,718,000 (Table 28).

***Scenario NCS7***

This scenario assumes that all corn acres use the Maximum Return to Nitrogen Rate, nitrification inhibitor used with all commercial fall applied nitrogen, sidedress all spring applied nitrogen, 31.5% of all ag land is treated with a wetland, 70% of the tile drained acres are treated with a bioreactor, 70% of all applicable acres have controlled drainage, and 70% of all agricultural streams have a buffer. This scenario is estimated to have the potential to reduce nitrate-N loading by 127,000 tons/year which is approximately a 41% overall nitrate-N load reduction at an annual cost of approximately $240,300,000 (Table 28).

***Scenario NCS8***

This scenario is the same as NCS7 except that phosphorus reduction practices are added to achieve the necessary phosphorus reduction goal. For this scenario the cost for the nitrate-N reduction is $240,300,000 but the cost for the phosphorus reduction is $-163,377,000 (benefit). As a result, the total cost for this scenario where there is approximately a 41% overall nitrate-N load reduction and 29% overall phosphorus load reduction is $76,923,000. (Table 28)

**Table 28. Example Statewide Combination Scenarios that Achieve the Targeted Nitrate-N Reductions, Associated Phosphorous Reductions and Estimated Equal Annualized Costs based on 21.009 Million Acres of Corn-Corn and Corn-Soybean Rotation.**

**Notes: Research indicates large variation in reductions from practices that is not reflected in this table. Additional costs could be incurred for some of these scenarios due to industry costs or market impacts.**

| | | Nitrate-N | Phosphorus | Cost of N Reduction from baseline ($/lb) | Initial Investment (million $) | Total EAC* Cost (million $/year) | Statewide Average EAC Costs ($/acre) |
|---|---|---|---|---|---|---|---|
| Name | Practice/Scenario** | % Reduction from baseline | | | | | |
| NCS1 | Combined Scenario (MRTN Rate, 60% Acreage with Cover Crop, 27% of ag land treated with wetland and 60% of drained land has bioreactor) | 42 | 30 | 2.95 | 3,218 | 756 | 36 |
| NCS2 | Combined Scenario (MRTN Rate, 100% Acreage with Cover Crop in all MLRAs but 103 and 104, 45% of ag land in MLRA 103 and 104 treated with wetland, and 100% of tile drained land in MLRA 103 and 104 treated with bioreactor) | 39 | 40 | 2.61 | 2,357 | 631 | 30 |
| NCS3 | Combined Scenario (MRTN Rate, 95% of acreage in all MLRAs with Cover Crops, 34% of ag land in MLRA 103 and 104 treated with wetland, and 5% land retirement in all MLRAs) | 42 | 50 | 4.67 | 1,222 | 1,214 | 58 |
| NCS4 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 85% of all tile drained acres treated with bioreactor, 85% of all applicable land has controlled drainage, 38.25% of ag land treated with a wetland) | 42 | 0 | 0.88 | 4,810 | 225 | 11 |
| NCS5 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 65% of all tile drained acres treated with bioreactor, 65% of all applicable land has controlled drainage, 29.25% of ag land treated with a wetland, and 15% of corn-soybean and continuous corn acres converted to perennial-based energy crop production) | 41 | 11 | 5.58 | 3,678 | 1,418 | 67 |
| NCS6 | Combined Scenario (MRTN Rate, 25% Acreage with Cover Crop, 25% of acreage with Extended Rotations, 27% of ag land treated with wetland, and 60% of drained land has bioreactor) | 41 | 19 | 2.13 | 3,218 | 542 | 26 |
| NCS7 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 70% of all tile drained acres treated with bioreactor, 70% of all applicable land has controlled drainage, 31.5% of ag land treated with wetland, and 70% of all agricultural streams have a buffer) | 42 | 20 | 0.95 | 4,041 | 240 | 11 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| NCS8 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 70% of all tile drained acres treated with bioreactor, 70% of all applicable land has controlled drainage, 31.5% of ag land treated with a wetland, and 70% of all agricultural streams have a buffer) - Phosphorus reduction practices (phosphorus rate reduction on all ag land, Convert 90% of Conventional Tillage CS & CC acres to Conservation till and Convert 10% of Non-No-till CS & CC ground to No-Till) | 42 | 29 | *** | 4,041 | 77 | 4 |

* EAC stands for Equal Annualized Cost (50 year life and 4% discount rate) and factors in the cost of any corn yield impact as well as the cost of physically implementing the practice. Average cost based on 21.009 million acres, costs will differ by region, farm and field.

** Scenarios that include wetlands, bioreactors, controlled drainage and buffers have substantial initial investment costs.

*** The N practices and cost of N reduction are the same as NCS7. Reducing P application meets the P reduction goal and lowers the cost of the scenario.

## Additional Economic Considerations

The cost estimates reported were equal annualized costs (EAC). However, edge of field practices have a high initial investment (Table 29) while the other practices primarily have an annual cost. The EAC includes the amortized cost of the initial investment over the life of the investment (50 year life and 4% discount rate).

**It is important to consider the initial investment of practices as a possible hurdle as this up-front cost may limit adoption.** For example, wetlands have a large initial investment but very low annual operating cost. Cover crops have low initial cost but an operating expense to plant and burn down, plus annual yield drag. Practices to be implemented must be both feasible to adopt and affordable to operate. Individual farmer preference and local landscape constraints also will influence the decision.

**Table 29. Edge-of-Field Practices with Significant Initial Investment to Install, Potential Area, Estimated Initial Investment and Equal Annualized Costs.**
**Note: A positive $/lb N reduction, total cost or EAC is a cost. A negative $/lb N reduction, total cost or EAC is a benefit.**

| | | | Investment and Re-investment (Million $) | | Equal Annualized Cost (Million $/year) | | | |
|---|---|---|---|---|---|---|---|---|
| **Name** | **Practice/Scenario** | **Total Area Impacted for practice (Million acre) *** | **Initial Investment** | **Present Value of Replacement Cost** | **Annualized Initial Investment** | **Annualized Maintenance Cost** | **Annual Operating Cost (including impact on Crop Yield)** | **Total Equal Annualized Cost** |
| W | Installing wetlands to treat 45% of the ag acres | 12.8 | 4,044 | 27 | 188 | 1 | 1 | 191 |
| BR | Installing denitrification bioreactors on all tile drained acres | 9.9 | 1,320 | 650 | 61 | 30 | 10 | 101 |
| BF | Installing Buffers on all applicable lands ** | 0.4 | 114 | 0 | 5 | 0 | 82 | 88 |
| CD | Installing Controlled Drainage on all applicable acres | 1.8 | 295 | 68 | 14 | 3 | 1 | 18 |

* Acres impacted include soybean acres in corn-soybean rotation as the practice has a benefit to water quality from the rotation.
** Acres impacted for buffers are acres of buffers implemented and EAC are per acre of buffer.

Similar tradeoffs occur when selected combination scenarios explained in the N-report are considered (Table 30). NCS1, NCS3, and NCS8 meet the N and P reduction targets of 41 and 29 percent, respectively. Compared to NCS3, NCS1 has a $2 billion higher initial investment, but $474 million lower annual operating cost. While the EAC for NCS8 is $77 million per year the initial investment is approximately $4 billion. NCS4 and NCS7 have low annual costs and high initial costs, but most importantly, do not meet the target for P reduction.

**A caution when reviewing average investment and average cost values - these are based on 21.009 million acres in continuous corn and corn-soybean rotation. In reality, the practices and costs will differ due to site-specific characteristics. However, the average investment and cost helps put the state number in perspective relative to other costs and returns.**

**Table 30. Initial Investment and Equal Annualized Cost of Examples of Combination Scenarios. Notes: NCS1, NCS3 and NCS8 Achieve Both Nitrogen and Phosphorous Target Reductions; Remaining Scenarios Meet Only the Nitrogen Target.**

| | | Investment and Re-investment (Million $) | | Equal Annualized Cost** (Million $/year) | | | |
|---|---|---|---|---|---|---|---|
| **Name** | **Practice/Scenario** | **Initial Investment** | **Present Value of Replacement Cost*** | **Annualized Initial Investment** | **Annualized Maintenance Cost** | **Annual Operating Cost (including impact on Crop Yield)** | **Total Equal Annualized Cost** |
| NCS1 | Combined Scenario (MRTN Rate, 60% Acreage with Cover Crop, 27% of ag land treated with wetland, 60% of drained land has bioreactor) | 3,218 | 406 | 150 | 19 | 587 | 756 |
| NCS2 | Combined Scenario (MRTN Rate, 100% Acreage with Cover Crop in all MLRAs but 103 and 104, 43% of ag land in MLRA 103 and 104 treated with wetland, 95% of tile drained land in MLRA 103 and 104 treated with bioreactor) | 2,357 | 355 | 110 | 17 | 505 | 631 |
| NCS3 | Combined Scenario (MRTN Rate, 95% of acreage in all MLRAs with Cover Crops, 34% of ag land in MLRA 103 and 104 treated with wetland, 5% land retirement in all MLRAs) | 1,222 | 8 | 57 | 0 | 1,156 | 1,214 |
| NCS4 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 85% of all tile drained acres treated with bioreactor, 85% of all applicable land has controlled drainage, 38.25% of ag land treated with a wetland) | 4,810 | 632 | 224 | 29 | -28 | 225 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| NCS5 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 65% of all tile drained acres treated with bioreactor, 65% of all applicable land has controlled drainage, 29.25% of ag land treated with a wetland, and 15% of corn-soybean and continuous corn acres converted to perennial-based energy crop production) | 3,678 | 483 | 171 | 23 | 1,224 | 1,418 |
| NCS6 | Combined Scenario (MRTN Rate, 25% Acreage with Cover Crop, 25% of acreage with Extended Rotations, 27% of ag land treated with wetland, 60% of drained land has bioreactor) | 3,218 | 406 | 150 | 19 | 373 | 542 |
| NCS7 | Combined Scenario (MRTN Rate, Inhibitor with all Fall Commercial N, Sidedress All Spring N, 70% of all tile drained acres treated with bioreactor, 70% of all applicable land has controlled drainage, 31.5% of ag land treated with a wetland, 70% of all agricultural streams have a buffer) | 4,041 | 521 | 188 | 24 | 28 | 240 |
| NCS8 | This scenario is the same as NCS7 except phosphorus reduction practices are added to achieve the necessary phosphorus reduction goal. For this scenario the cost for the nitrate-N reduction is $240.3 million but the cost for the P reduction is $-163.4 (benefit). Total cost for this scenario with approximately 41% nitrate-N load reduction and 29% P load reduction is $77 million. | 4,041 | 521 | 188 | 24 | -135 | 77 |

* Present value of replacement structures to match 50-year time horizon.
** Annualized cost

## Future Research Needs

A number of potential practices have been discussed and would be good to investigate further. However, and of importance, little research is available that documents concurrent crop production and water quality (nitrate-N loss) effects. Future research in Iowa focused on nutrient reduction strategies should include:

- Variable nitrogen rate application
- In-season sensor-based nitrogen application
- Nitrogen and manure additives, inhibitors, and slow release products
- Better estimates of actual nitrogen application rates (including fertilizer and manure), and on a geographic-specific basis.
- While MLRA scale estimates for nitrogen application rates were used in this assessment, county-based estimates from David et al. (2010) show some counties with estimated average application rates much higher than the statewide or MLRA average rate. This in part could be due to manure application rate in these counties. As a result, there needs to be increased focus on the role of manure in supplying crop nitrogen needs.
- Information on the sustainability of nitrogen in soil organic matter with decreased nitrogen application rates
- Two-stage ditch designs
- Oxbow restoration and stream meanders
- Directing tile drainage water through riparian buffers
- Impact of denitrification practices on greenhouse gas emissions
- Overall nitrate reduction with combinations of practices
- Large scale monitoring of nitrate transport as impacted by single and combination of nitrate reduction practices
- Large scale modeling to estimate nitrate-N transport with models like the Root Zone Water Quality Model (RZWQM)
- Integration and comparison to USGS SPARROW modeling
- Developing cover crop systems that do not reduce yields for the following corn crop
- Need for water quality and yield impacts of living mulches, specifically bluegrass
- There is a need for monetizing economic benefits that might be derived from improved water quality or other ecosystems services. These could be compared to the cost of nutrient reduction practice implementation.

While significant research has been conducted on the potential performance of various nutrient reduction practices, there still is a need for development of additional practices, testing of new practices, evaluating potential unintended consequences of practices, and verifying practice performance at implementation scales. Many of the studies used in this evaluation and practice choice were conducted at the plot scale, and while they provide critical information, and studies of this kind should continue, there also is a need for studies that scale up the area of practice implementation to better assess water quality impacts across landscapes and with multiple practices.

In addition, to assess potential landscape-scale changes, there is a need for better tracking of practices currently in place, including but not limited to land use, crop rotations, nutrient applications, tillage, and conservation practices. In the analysis conducted here, the practices and existing conditions were aggregated on a MLRA scale, but actual implementation would be at a much finer-scale. This highlights the need for actual practice information at the field level to make better future assessments on potential gains or actual gains in achieving nitrogen and phosphorus nutrient reductions to surface waters.

## Appendix A – Literature Reviewed

Not all literature listed here was used in determining practice impacts on nitrate reduction. However, all research work listed was reviewed for applicability to this nitrogen reduction strategy effort. From the research literature, nitrate concentration, load, and yield data were added to a spreadsheet table for compilation and comparison. Comments in the following text similar to "data was added to the table" indicate that the water quality or agronomic data from the research were used in the spreadsheet and mean, min, and max calculations.

### Timing of Nitrogen Application

Data from a total of six studies went into determining the impact on nitrate and corn yield. Current thoughts of the nitrogen science team are that the price variability in nitrogen in recent years has limited the cost difference between fall and spring application, therefore, the same fertilizer nitrogen cost is used for all timing comparisons. There will be a possible economic gain due to increased yields with a change in application timing.

(Randall and Sawyer, 2008)
Interpretation section – "Spring application of N is superior to fall application in most cases." The advantages are limited, however, to warm and wet conditions. Authors suggest losses of fall applied N may be as much as 50% under perfect denitrification conditions. Reductions of N loss due to leaching are estimated to be around 15% with as little as no reduction and as much as 25%, depending on application timing and weather conditions. Applying in spring could cost between $5 and $10 per acre more. However, this could be a wash if more is applied in the fall to offset expected losses. Authors suggest an estimated 12.9 million acres out of 50.6 million acres in the Corn Belt could benefit. This paper was not used in the practice table but was used to guide estimates of fall nitrogen application.

(Randall and Mulla, 2001)
This paper reports an average of 20% load reduction at Waseca, Minnesota (1987-1993) when comparing fall vs. spring nitrogen application over a 4-year period. The addition of nitrapyrin reduced nitrate-N concentrations by 15%. The split application (pre-plant along with sidedress in a 40%-60% split) also reduced annual nitrate-N concentrations from tile lines by 20% over the same 4-year period. This study also included information about nitrate-N concentrations from different cropping systems, which was the same as information in (Randall et al., 1997). Data from this paper was not included in the practice table.

(Randall, 2008)
This paper has nitrate concentration numbers for both fall and spring applications, however, all fall applications used N-Serve, meaning there is no real control treatment to compare against. A point of interest is the fall 135 kg N/ha (120 lb N/acre) treatment with N-Serve and the spring 135 kg N/ha (120 lb N/acre) treatment have weighted nitrate-N concentrations of 13.2 and 13.7 mg/L, respectively. Corn yields for the fall 120 lb N/acre treatment with N-Serve were 0.9 Mg/ha (14 bu/acre) higher than the corresponding spring application. Data for yield and nitrate was added to the table for timing, inhibitor, and sidedress.

(Vetsch and Randall, 2004)
This paper has limited data for use in this project. Fall corn yields for grain and silage were 10.9 and 16.8 Mg/ha, respectively, while spring yields for corn were 11.7 and 17.6 Mg/ha for grain and silage, respectively. Anhydrous ammonia at 123 kg N/ha was applied to both spring and fall treatments. Data was not included in the practice table.

(Randall and Vetsch, 2005c)
This 6-year study from Waseca, Minn., has information about nitrogen application timing as well as the use of a nitrification inhibitor with a 134 kg N/ha application rate. All data has been added to the table as site years. The main effects are:

- 6-year 11% average increase in yield when moving from fall to spring application with 1 year having a 71% increase. The average over the other 5 years is actually slightly negative.
- 6-year average of 8% increase in yield with the addition of N-Serve. One year had a 41% increase with a 1.6% increase excluding that year.

Data was included in the practice table.

(Randall et al., 2003a)
This was a 7-year study at Waseca, Minn., (1987-1993) with 150 kg N/ha application rate. This study looked at timing, nitrapyrin, and sidedress. Site years have been added to the table. Main effects are:

- 7-year 5.4% average increase in corn yield when moving from fall to spring.
- 7-year 10.2% average increase in corn yield when moving from fall to pre-plant + sidedress (40-60 split).
- 7-year 5.9% average increase in corn yield when using nitrapyrin in the fall.

Data was included in the practice table.

(Randall et al., 2003b)
This was the drainage component of the research at Waseca, Minn., from 1987 to 1994. Nitrogen application rate was 150 kg N/ha. Site years have been added to table and include both corn and soybean. One note is that there was no drainage in the soybean plots in 1988 or 1989 and no drainage in the corn plots in 1989. Main effects are:

- 7-year 6.8% average nitrate-N decrease when considering the entire rotation and moving from fall to spring nitrogen application over the study years. The range was an increase of 80% in the soybean year of 1992 and a reduction of 22.9% in the corn year of 1990.
- 7-year 4.8% average nitrate-N decrease when considering the entire rotation and moving from fall application to a pre-plant/sidedress split (40-60). The range was an increase of 60% in the soybean year of 1992 and a reduction of 26.3% in the corn year of 1991.

Data was included in the practice table.

(Randall and Vetsch, 2005a)
This research was carried out at a site in Waseca, Minn., between 1994 and 2000. The study investigated nitrogen loss from plots with anhydrous applied at 135 kg N/ha in the corn year of a corn-soybean rotation. Information on a full rotation was collected between 1995 and 1999 with 1994 having a corn crop only and 2000 having a soybean crop only. Results show nitrate-N concentrations for spring-applied nitrogen are lower than the corresponding fall-applied treatments in the corn year. However, the soybean plots have nearly the same nitrate-N concentrations for both treatments. All site year data has been added to the practice table. This paper also had information on nitrification inhibitors, which was added to the practice table.

(Clover, 2003)
This thesis explored nitrate-N concentrations from three years of a corn-soybean production in central Illinois. The treatments involved a fall and spring application as well as using a nitrification inhibitor. In addition to the spring application the study investigated a sidedress application. Both fall and spring treatments included a 76 kg N/ha, 156 kg N/ha, and a 234 kg N/ha rate. The inhibitor and sidedress treatments were applied at the 156 kg N/ha rate. Nitrate-N concentrations were lower coming out of the spring-applied corn plots (~25%), while the corresponding soybean plots were about the same for both

spring-applied and fall-application (depending on the year). The timing, sidedress, and inhibitor numbers have all been added to the practice table.

### Rate of Nitrogen Application

The tile flow nitrate-N data related to application rate will be compared to the currently used rate equation from Lawlor et al. (2008). Preliminary investigation of research on nitrate-N concentration from tile drainage at various nitrogen application rates shows a similar trend to the Lawlor study even when considering data from surrounding states. Modifications to the Lawlor study have not been made to this point. This approach assumes changing nitrogen application rates will not have an impact on water yield from tile drainage. Again, this study is primarily limited to nitrate-N concentrations as water yield is addressed in a separate effort.

Rate has a significant impact on resulting tile flow nitrate-N concentration. Rate is also an important factor in most other practices as each farmer chooses the rate of nitrogen to apply. Because of this, rate serves as a starting point for the in-field practices.

(Lawlor et al., 2008)
This research was conducted near Gilmore City, Iowa, between 1990 and 2004. Information gathered included nitrogen application rate and annual flow-weighted nitrate-N concentration. This study only looked at the corn-soybean rotation. All data has been added as site years to the practice table. The equation developed in this publication will be compared to an equation developed with all available data from Iowa and southern Minnesota.

(Bakhsh et al., 2005)
This paper summarizes work conducted at Nashua, Iowa, from 1993 to 1998. Although the focus of the paper was liquid swine manure, no directly comparable application rates were available for incorporation into the source section of the practice table. The commercial fertilizer rates will be used as part of a nitrogen application rate vs. nitrate-N concentration response curve. The data has been added to the table as site years, but is not being used.

(Randall et al., 2003b)
This paper was summarized under the Timing of Nitrogen Application practice section. Only treatments with applications in the spring were added to the Rate practice in order to stay consistent with the Lawlor et al. (2008) research. However, data is only being used for comparison.

(Kanwar et al., 1995)
This paper is summarized in the Sidedress practice section, but data for rate has been added as site years to the table.

(Jaynes et al., 2001)
This study was conducted in central Iowa on a 22 ha field with an existing tile system in a corn-soybean rotation. Results show an increase in nitrate-N concentration with an increase in fertilizer rate as well as a general increase in corn yield with an increase in fertilizer rate. Fertilizer rates were 202, 135, and 67 kg/ha. Results have been added to the practice table.

### Sidedress

Not all sources listed here were used in the nitrogen reduction practice table. Suitability was determined based on proximity to Iowa and information collected and provided in the paper. A total of 9 studies were used in the three sidedress categories (sidedress compared to fall applied, sidedress compared to spring pre-plant, and sidedress test based compared to spring pre-plant) in the practice table.

(Clover, 2003)
See information under the Timing of Nitrogen Application practice section.

(Jaynes, 2009)
This poster, presented at the 2009 ASA annual meeting, suggested there was no statistically significant impact on nitrate-N concentrations when sidedressing nitrogen at early to mid-season (V6 or V10) when comparing to nitrogen application just after planting. Data has been added to the practice table.

(Bakhsh et al., 2002)
This research from Nashua, Iowa, highlights 6 years of data (1993-1998) comparing pre-plant applied N (110 kg N/ha) and sidedress applied N (with 30 kg N/ha applied with planting) based on late-spring nitrate tests (LSNT) results (total N application ranged from 123 kg N/ha to 225 kg N/ha). Results are mixed, however, the range of nitrate concentration reductions is -28.6 to 45.2%. Corn yield increases ranged from 1.7 to 69.8%. This data has been added to the practice table as site years.

(Ruiz Diaz et al., 2008)
This paper reports corn yields for various treatments for 30 sites in Iowa over 3 years. The treatments considered here are 134 kg N/ha pre-plant (also included early season sidedress and post emergence); 269 kg N/ha pre-plant (also included early season sidedress and post emergence); 67+ kg N/ha which included pre-plant or early season with additional N added mid-late season based on sensor readings (average total application over the 30 sites was 135 kg N/ha); and 134+ kg N/ha which included pre-plant or early season with additional N added mid-late season based on sensor readings (average total application over the 30 sites was 146 kg N/ha). The 67+ treatment is compared to the 134 treatment and the 134+ is compared to the 269 treatment in terms of corn yield. There is a large range of responses (-11.9 to 7.3 Mg/ha) with an average of -2.8 Mg/ha. No information on nitrate was measured. This dataset was not added to the practice table because, as of now, we are not including mid-season crop sensing-based sidedressing.

(Jaynes and Colvin, 2006)
This research from a site in central Iowa reports nitrate-N concentrations as well as corn yields. There were 4 treatments represented as H (high application rate corresponding to farmer application rate of 199 kg N/ha), M (medium application rate corresponding to the economic optimum of 138 kg N/ha), L (a purposely low rate of 69 kg N/ha), and R (a treatment receiving two rounds of 69 kg N/ha – one early and one midseason). Data from the two treatments with 138 kg N/ha total application was assessed. Data was added to the practice table as site-year under sidedress.

(Jaynes et al., 2004)
This paper highlights a watershed study in Iowa looking at changing fertilizer application practice to a rate based on a late spring nitrate test (LSNT). In this study, two conventional practice watersheds were compared to one where farmers applied nitrogen based on the LSNT for years 1992 to 2000. There was a noticeable reduction in nitrate concentration after the first year of the 5-year study where historically there was no statistical difference in the three watersheds. A summary is shown here and data was added to the practice table.

**Table 3. Flow-weighted average annual $NO_3$ concentration in the discharge from the control (CN1 and CN2) and treated (TR1) subbasins.**

| Subbasin | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 |
|---|---|---|---|---|---|---|---|---|---|
| | mg N $L^{-1}$ | | | | | | | | |
| CN1 | 9.9 | 8.2 | 9.2 | 13.1 | 14.0 | 8.4 | 11.1 | 15.8 | 16.5 |
| TR1 | 12.5 | 9.2 | 8.9 | 16.0 | 15.6 | 10.8 | 10.2 | 11.7 | 11.0 |
| CN2 | 13.7 | 9.7 | 10.2 | 16.7 | 15.4 | 13.1 | 14.0 | 16.5 | 15.1 |

(Randall et al., 2003a; Randall et al., 2003b)
These papers were summarized under the Timing of Application practice section.

(Kanwar et al., 1995)
This paper had 2 years of data (1993 and 1994) on nitrate-N response from LSNT recommended N application rates. The data was different than that presented in Bakhsh et al. (2005). Data from this paper has also been added as site years to the Rate and Source sections (to possibly be compared to the rate curve in the future). Over all, the treatments averaged a 9% reduction in nitrate-N concentration when compared to the spring pre-plant treatment. Data has been added to the practice table.

(Baker and Melvin, 1999)
This report has results from a sidedress treatment from 1994 to 1999. Application rates were partially based on LSNT results, and ranged from 45 to 157 kg N/ha. Nitrate concentrations were not significantly different and yields were generally lower with sidedressing compared to pre-plant N application. Data from this paper has been added to the practice table.

## Application Source

Not all data from literature listed here was included in the practice table. Four studies were used for the liquid swine manure section and three studies were used for the poultry manure section.

(Lawlor et al., 2011)
This research at Gilmore City, Iowa, shows the differences between commercial fertilizer and liquid swine manure. The timing component was also used from this work. The first-year nitrogen availability rate of liquid swine manure was assumed to be 100%, which is the top end of the current recommended first-year crop availability values (Sawyer and Mallarino, 2008b). All data has been added to the practice table as site years, although a linear interpolation was done to make direct N application rate comparisons.

(Chinkuyu et al., 2002)
This research conducted at Ames, Iowa, was a 3-year study (1998 to 2000) looking at the application of laying hen manure. The treatments are spring-applied UAN at 168 kg N/ha, spring-applied laying hen manure at 168 kg N/ha (actual total N application rates of 115, 219, and 117 kg N/ha for 1998 to 2000), and spring-applied laying hen manure at 336 kg N/ha (actual application rates of 254, 324, and 324 kg N/ha for 1998 to 2000). There was also an associated lysimeter study with the same treatments. The 168 kg N/ha manure treatment had actual rates of 167, 169, and 162 kg N/ha, while the 336 kg N/ha manure treatment had 337, 338, and 325 kg N/ha applied. The paper assumed a nitrogen availability of 75% for the manure applications, which was accepted practice at the time, but the data has been re-estimated here to assume 55% availability, which is the current recommendation (Sawyer and Mallarino, 2008b). Data has been added as site years into the table with a linear interpolation between commercial fertilizer applications to make a better comparison.

(Bakhsh et al., 2005)
This paper was summarized in the Nitrogen Application Rate section as there were no directly comparable rates of liquid swine manure and commercial fertilizer. The rates and nitrate results have been added into the practice table as site years, for possible comparison to any rate equation that is developed.

(Ruiz Diaz and Sawyer, 2008; Ruiz Diaz et al., 2011)
These papers were used for yield numbers from poultry manure applications. Results show little yield impact (positive or negative) of using manure. Data was added to the practice table.

(Rakshit, 2002)
This thesis had two years of data from multiple farms with multiple liquid swine application rates. Although there were no exact rate comparisons between manure and fertilizer nitrogen in the study, the multiple manure nitrogen rates and multiple nitrogen fertilizer rates applied in addition to the manure nitrogen allowed for linear interpolation between rates for comparison. All data was added to the practice table, but there tended to be a slight yield decrease in the comparison.

## Nitrification Inhibitors (Nitrapyrin)

Not all literature here was included in the Nitrification Inhibitor section of the practice table. A total of 8 studies were included.

(Randall and Sawyer, 2008)
The interpretation section indicated mixed results on nitrate loss, yet some positive results are shown with the addition of nitrapyrin and anhydrous ammonia in late October (14% reduction). Authors suggest an approximate 15% of corn acres might benefit from use of nitrapyrin with late-applied anhydrous ammonia. At an estimated cost of $7.50/acre with 3.5 lb/acre nitrate-N reduction, the technology will cost around $2.15/lb nitrate-N reduced. This paper was only used as a guide.

(Randall, 2008)
See timing section for a brief overview of this paper.

(Nelson and Huber, 1980)
This article addresses the use of N-Serve from Dow Chemical Company. This paper states the chemical is registered with the EPA "…for use with ammonical fertilizers applied to corn, sorghum, wheat, and cotton," with application rates between 0.27 to 0.56 kg a.i./ha. Also, N-Serve should be band-applied a minimum of 10 cm below the surface. This study also reports corn yield response to the nitrification inhibitor nitrapyrin at 0.55 kg a.i./ha added to fall-applied anhydrous ammonia. The range of yield increase for nitrapyrin was 104, 32, 13, and 8% for 1973, 1974, 1975, and 1978, respectively. The authors also discuss yield increases from using the inhibitor in the spring, but that will not be addressed here. Also, the authors provide an opinion on the probability of seeing a yield increase on different types of soils due to the use of nitrification inhibitors (does not distinguish between chemical compounds). Results are represented below where "Poor, <20% chance of increase at any location any year; Fair, 20-60% chance of increase; Good, >60% chance of increase." Specific data was not added to the practice table.

| Soil Texture | Fall Applied |
|---|---|
| Sands | Poor |
| Loamy sands, sandy loams, and loams | Fair-Good |
| Silt loams | Good |
| Clay loams and clays | Good |

(Wolt, 2004)
This meta-analysis used several studies, but only those conducted in the Midwest and with nitrapyrin application in the fall for corn will be used here. There were no applicable studies with nitrate leaching except one by Yadav (1997), which reports a residual nitrate-N reduction in the soil sink (below the root zone) of 24.5% and 25.4% at two sites, but did not distinguish between inhibitor application time. There were no studies used in the meta-analysis from Iowa where nitrapyrin was applied in the fall with anhydrous before corn so results were not directly applicable to Iowa. However, the following table highlights work done in the Midwest which indicated an average of 18% yield increase with a standard deviation of 41.8%. Data was not used in the practice table, however, results for Iowa are similar.

| State | Yield Change | Study |
|---|---|---|
| OH | 3 | Johnson 1995 |
| | 10.7 | |
| | 3.1 | |
| IN | 60 | McCormick et al. 1984 |
| | 1.7 | |
| | 27.9 | |
| | 1.4 | |
| OH | 2 | Stehouwer and Johnson 1990 |
| | 16 | |
| | 22.2 | |
| | 5.4 | |
| | -0.8 | |
| | 0 | |
| | 8.2 | |
| IN | 5.1 | Sutton et al. 1985,1986 |
| | 5.4 | |
| IL | 0 | Touchton et al. 1979a |
| IL | 14.6 | Touchton et al. 1979b |
| | -12.1 | |
| IN | 206.9 | Warren et al. 1975 |
| | 1.3 | |
| | 30.7 | |
| IN | 8.7 | Warren et al. 1980 |
| | 18.8 | |
| | 9.8 | |

(Owens, 1987)
This paper presents results from lysimeters in Ohio. A nitrate leaching reduction was found, but the timing of nitrapyrin treated urea application was not clearly described. Over 6 years the two treated lysimeters had a 23.7 and 26.9% reduction in nitrate-N concentration. All site years have been added to the practice table.

(Ellsworth et al., 1999)
This brief conference proceedings article about research on N-Serve in Iowa shows a 6.5% increase in yield when comparing plots with 125 lb N/acre anhydrous ammonia treated with N-Serve and applied in the fall to plots at 125 lb N/acre without N-Serve applied in the fall. Data has been added to the practice table.

(Nelson et al., 1977)
This paper summarizes results from a study in Indiana at the Pinney-Purdue Agricultural Center in 1975. The study looked at continuous corn at 0, 85, and 179 kg N/ha application rates with and without nitrapyrin. The study had no leaching data. The crop yields were added to the practice table.

(Clover, 2003; Randall and Vetsch, 2005b; Randall and Vetsch, 2005c; Randall et al., 2003a; Randall et al., 2003b)
See information discussed in the Timing of Nitrogen Application section.

## Drainage Water Management and Shallow Drainage

A number of studies were used in this section. All but one was included in the Agricultural Drainage Management Coalition (ADMC) report.

(Helmers et al., 2010)
This paper addressed water table response at a site with conventional, controlled, and shallow drainage at Crawfordsville, Iowa. Yield data was available for split plots with both corn and soybean which showed no statistically significant differences in either corn or soybean yields. Drainage volume was significantly reduced in both the controlled drainage and shallow drainage with three-year averages for the conventional, controlled, and shallow drainage at 31.5, 22.0, and 18.5 cm, respectively. The site year yield data was added to the practice table.

(Helmers, Unpublished)
This is research with drainage water management at Crawfordsville, Iowa. Controlled drainage showed a slight reduction in nitrate-N concentration (5.6%) when compared to conventional drainage. However, there was an increase in nitrate-N concentration of 29.4% in the shallow drainage treatment. Loads were also estimated from data reported in this study. That information was not added to the practice table as the (ADMC, 2011) study includes that data.

(Sands et al., 2008)
The same data was shown in a 2006 proceedings paper and a 2008 international paper.
In this 5-year study in Minnesota, little difference was seen in outflow concentration from shallow drainage vs. deep drainage. In addition, little difference was seen in differing levels of drainage intensity. The primary result of the study is a statistically significant reduction in drainage volume with shallow drainage as well as a significant reduction in nitrate load. In addition, there is a statistically significant reduction in drainage volume when drainage intensity is reduced, as well as a significant reduction in nitrate load. Reporting is a bit difficult here as results for both drainage depths include both drainage intensities and results for both drainage intensities include both drainage depths. The drainage intensity will not be used, only the drainage depths. Also, only reductions in load will be used. There was no yield data with this research. Data was not added to the practice table.

(ADMC, 2011)
This report lists several controlled and shallow drainage sites in Minnesota, Iowa, and Illinois. Data from locations not in or near the Iowa border were not used due to possible differences in flow patterns. Concentrations reported were generally similar between conventional, shallow, and controlled drainage. However, there was a significant volume reduction in the controlled and shallow drainage. Results from the sites were summarized and added to the practice table.

(Cooke et al., 2002)
This study was used due to the location of the research – Douglas County, Ill. Authors found significant nitrate-N load reduction (22 to 51%) in the shallow (3-foot and 2-foot deep drains) drainage plots when compared to conventional drainage. Data was added to the practice table.

## Extended Rotations – Ideally 2 or more years of alfalfa

Although two or more years of alfalfa in the rotation was the goal for inclusion of research, very little data from around Iowa was available. This section does include other extended rotations with a total of four studies contributing.

(Liebman et al., 2008)
This 4-year study from Iowa investigates a number of cropping rotations including a 2-year (corn-soybean), a 3-year (corn-soybean-small grain + red clover green manure), and a 4-year (corn-soybean-small grain + alfalfa-alfalfa hay). Although there are no nitrate tile flow concentrations, there was a yield and an economic analysis of the different rotations. Fertilizer was managed based on soil testing and included composted manure, urea applied at planting, and sidedressed UAN as needed. Phosphorus and potassium were also applied as needed. Since this wasn't a nitrate loss paper, fertilizer application will not be considered in relation to crop yields, although fertilizer costs were factored into the economic analysis. Crop yields were added to the practice table, but not the economic values.

Gross revenues, production costs, labor requirements, and returns to land and management for contrasting rotation systems, 2003 to 2006.

| | | | | | |
|---|---|---|---|---|---|
| | | | | Return to land | Return to land |
| | Gross | Production | Labor | and management, | and management, |
| Rotation | revenue† | cost‡ | requirement | no subsidies§ | with subsidies¶ |
| | $/ha/yr | $/ha/yr | hours/ha/yr | $/ha/yr | $/ha/yr |
| 2-yr | | | | | |
| corn | 1202.05 | 582.48 | 1.61 | 603.52 | 793.96 |
| soybean | 757.18 | 331.99 | 2.03 | 405.01 | 489.83 |
| average | 979.62 | 457.24 | 1.82 | 504.27 | 641.90 |
| | | | | | |
| 3-yr | | | | | |
| corn | 1238.63 | 500.42 | 4.25 | 695.68 | 895.57 |
| soybean | 816.34 | 291.61 | 2.52 | 499.61 | 585.71 |
| small grain/clover | 499.29 | 251.99 | 1.9 | 228.28 | 303.29 |
| average | 851.42 | 348.01 | 2.89 | 474.52 | 594.85 |
| | | | | | |
| 4-yr | | | | | |
| corn | 1250.41 | 483.97 | 4.27 | 723.73 | 924.15 |
| soybean | 824.12 | 292.63 | 2.52 | 506.35 | 592.65 |

| | | | | | |
|---|---|---|---|---|---|
| small grain/alfalfa | 613.8 | 350.44 | 2.67 | 236.65 | 311.64 |
| alfalfa | 929.04 | 194.27 | 4.17 | 693.1 | 768.1 |
| average | 904.34 | 330.33 | 3.41 | 539.96 | 649.14 |
| | | | | | |
| | | | | | |
| † Crop prices used in the calculations were \$95.70 $Mg^{-1}$ for corn; \$227.85 $Mg^{-1}$ for soybean; \$82.45 $Mg^{-1}$ for triticale grain; | | | | | |
| \$110.25 $Mg^{-1}$ for oat grain; \$54.45 $Mg^{-1}$ for triticale and oat straw; and \$77.10 $Mg^{-1}$ for alfalfa hay. | | | | | |
| ‡ Costs included field operations, handling, and hauling, and for corn, drying as well. Land and labor costs were not included. | | | | | |
| § Labor charge was set at \$10 $h^{-1}$. | | | | | |
| ¶ Crop subsidies comprised loan deficiency, counter cyclical, and direct payments. | | | | | |

(Tomer, 2011)
This personal communication between Mark Tomer and Dan Jaynes represented 7-years of data – see Liebman et al. (2008) for a description of the study, and compared a corn-soybean rotation to a corn-soybean-small grain-alfalfa rotation. Results showed an 8 mg $NO_3$-N/L average tile flow nitrate concentration from the extended rotation and 11.5 mg $NO_3$-N/L from the 2-year rotation. Data were added to the practice table.

(Huggins et al., 2001)
This 3-year study from Minnesota investigated what happens with conversion from a continuous alfalfa or a CRP cropping system to a corn-corn-soybean rotation. This rotation does not exactly fit the intended rotation for this project, but it has been added to the practice table and will contribute to information about continuous corn and corn-soybean rotations.

(Kanwar et al., 2005)
This 6-year study had several plots with strip intercropping (corn/soybean/oat interseeded in berseem clover), an extended rotation (alfalfa/alfalfa/alfalfa/corn/soybean/oat), and a conventional rotation (corn/soybean). All fertilization was done in the spring with a sidedress application based on the late spring nitrate test (LSNT). Nitrate-N concentrations from all treatments were added to the practice table.

## Cover Crops

Seven studies were used for the cover crop section. Not all studies listed here were used due to lack of proximity to Iowa.

(Kaspar et al., 2008)
An interpretive summary for cover crops indicates that colder climates generally realize smaller benefits from cover crops due to limited growth and frozen soils limiting water movement. “Reductions in nitrate load observed with a cover crop range from 13% in Minnesota to 94% in Kentucky.” Establishment (seed for rye) will cost around \$25/acre giving a cost of \$0.57 to \$1.42 per pound of N reduced. Cover crops could likely be implemented on 70-80% of corn-soybean ground. Data were not added to the practice table.

(Kaspar et al., 2003)
This report summarizes work conducted west of Ames, Iowa. The study involved multiple treatments, however, only the cover crop (rye) and control treatments are considered here. All plots were fertilized with 224 kg N/ha (200 lb N/acre) as UAN, which was surface-applied in the spring before corn. Each treatment had four replicates. In the first year of monitoring, the cover crop nitrate-N concentrations in tile-flow were just greater than the control plots (27 compared to 25 mg $NO_3$-N/L), however, in the second year cover crop nitrate-N concentrations were much lower (6 compared to 19 mg $NO_3$-N/L). Corn yields from 2000 and 2002 were 10.3 and 12.4 Mg/ha (164 and 198 bu/acre) for the control plots while 10.3 and 11.0 Mg/ha (164 and 176 bu/acre) for the cover crop plots. Soybean yields in 2001 were 3.1 Mg/ha (46 bu/acre) for the control plots and 3.0 Mg/ha (44 bu/acre) for the cover crop plots. This data has been summarized in Kaspar et al. (2007), therefore, data from this report were not added to the practice table but were added from the 2007 paper.

(Kaspar et al., 2007)
A 4-year study in Iowa had an average 59.1% reduction in nitrate-N concentration in tile flow with a rye cover crop. This study had a corn yield response in year 1 of -9.7% with the cover crop, no difference in year 3, and no difference in soybean yield response in year 2 but a -6.7% response in year 4. Site year data were added to the practice table.

(Kaspar et al., 2012)
A 5-year study in Iowa had an average 44.4% reduction in nitrate-N concentration in tile flow with a rye cover crop and a 24.2% reduction in nitrate-N in tile flow with a oat cover crop. On average this study had a -0.2% yield response for corn after a rye cover crop and a -5.0% response after oat. Soybean after rye averaged a -6.5% yield response after rye and a -14.9% response after oat. Site year data were added to the practice table.

(Qi and Helmers, 2008)
This study conducted in northwest Iowa had a tile flow nitrate-N concentration reduction of 11% with a rye cover crop (this was not statistically significant), a reduction of 49.5% with kura clover (with no mention of corresponding corn yields), and a reduction of 60.4% when comparing a perennial grass system with a corn-soybean rotation. Data were not added to the practice table as it is reported in (Qi et al., 2011).

(Qi et al., 2011)
This paper, with research in Iowa, presents nitrate-N concentrations in tile flow from a rye cover crop (in both corn and soybean), a living mulch (kura clover) with corn, and a perennial forage. Over the 4 years of the study, there was no statistically significant reduction in nitrate-N concentration with a rye cover crop before the corn phase (12.8 mg $NO_3$-N/L) (with a yield of 8.1 Mg/ha) when compared to the control corn phase (13.8 mg $NO_3$-N/L) (with a yield of 8.4 Mg/ha, which is not statistically larger than with rye). With rye before soybean, however, there was a statistically significant reduction of 10.9% (11.4 mg $NO_3$-N/L) (with a yield of 2.5 Mg/ha) when compared to the soybean phase control (12.8 mg $NO_3$-N/L) (with a yield of 2.8 Mg/ha, which is not statistically larger than with rye). The kura clover living mulch was a continuous corn system which had 4-year average nitrate-N concentration of 6.8 mg $NO_3$-N/L (with a yield of 2.8 Mg/ha). The perennial forage treatment had a 4-year average nitrate-N concentration of 4.6 mg $NO_3$-N/L. Site year data were added to the practice table.

(Strock et al., 2004)
This paper reports research from southern Minnesota with three years of data. There was a 22.5% reduction in nitrate-N concentration in tile flow when comparing corn to corn after rye and a 47.7% reduction when comparing soybean to rye before soybean. There was no statistically significant change in observed crop yields for either corn or soybean with the rye cover crop and rye biomass averaging 1.4 Mg/ha for the three-year study period. Nitrate-N concentration for soybean in 1999 was statistically larger in 1999, and both of the rye treatments (before corn and before soybeans) were statistically smaller in 2000. The site years for both yield and nitrate-N concentration were added to the practice table.

(Sawyer et al., 2011a)
Results from four ISU outlying research farms in 2009-2011 (Ames, Crawfordsville, Lewis, and Nashua) showed an average 6% decrease in corn yield when following a rye cover crop. There was no effect of the rye cover crop on soybean yield. Data were added to the practice table.

(Pederson et al., 2010)
This report has information from 4 years (2007 to 2010), with a reduction in nitrate-N concentration in tile flow and a reduction in corn yield with the addition of a cover crop when comparing to spring UAN at 150 lb N/acre. The study was conducted at the NERF site near Nashua, Iowa Data were added to the practice table.

(PFI, 2011)
This report shows a significant reduction in corn yield at two locations in the study in 2009 and 2010 with seven total sites. There was one location where the cover crop treatment had a significantly increased corn yield. In general there was no significant difference in plots with cover crops compared to conventional agriculture. Data were added to the practice table.

### Living Mulches

Not all studies listed here were used to add data to the practice table.

(Kaspar et al., 2008)
Reduction in nitrate-N loss is assumed with the living mulch, but no information is available in the report. These systems can cost as much as $40.35 per acre per year, resulting in an assumed cost of $0.90 to $2.27 per pound of nitrate-N reduced. This data were not added to the practice table.

(Zemenchik et al., 2000)
This study looked at different methods of controlling kura clover for corn planting. Methods were a complete kill (with and without nitrogen added to the corn), band-killed, and suppressed (with and without nitrogen added to the corn). The results include corn yields but no nitrate leaching. Site-year data were listed in the practice table, but the main point is that the complete kura clover kill treatments generally have better yields, even when nitrogen is not added, than the band-killed or the suppressed treatments.

(Albrecht, 2009)
This report briefly outlines work that has been conducted with kura clover as a living mulch for corn. The author suggests yield loss of 0 to 10% in this type of system. In addition, the report suggests up to a 50% reduction in nitrate leaching (below the root zone). The data were not added to the practice table.

(Qi et al., 2011)
This paper from Iowa reports nitrate-N concentrations in tile flow from a rye cover crop (both corn and soybean crops), a living mulch (kura clover) with corn, and a perennial forage. This paper was summarized in the Cover Crops practice section.

(Sawyer et al., 2010)
This study was conducted on-farm in northeast Iowa in 2006 and 2007. There were 6 locations and 3 were with corn and the other 3 were soybean. Also, 6 nitrogen fertilizer application rates were used. Corn yield data were added to the practice table as site years.

## Energy Crops and Pasture

Not all studies listed here were used to add data to the practice table as some were not directly applicable. Two studies were used in the practice table for Energy Crops. The pasture section is assumed to be the same as energy crops, due to similarity in the systems and a lack of pertinent data for pastures.

(Owens et al., 1982)
This paper from Ohio reported subsurface water nitrate-N concentrations from a pasture system and found nitrate-N levels ranging from around 1 mg $NO_3$-N/L to just over 12 mg $NO_3$-N/L. The data set averages approximately 4 mg $NO_3$-N/L for the 5-year study. This study has no corn-soybean control. Nitrate-N concentrations from surface runoff are nearly always under 1 mg $NO_3$-N/L and will not be used in the practice table. Two notable trends: changing from continuous corn to pasture, it takes a number of years for subsurface nitrate-N concentrations to drop (watershed 104 in this study); and heavy winter animal feeding adds considerable nitrogen input into the pasture resulting in increasing nitrate concentrations each consecutive year because of buildup. Nitrate numbers were estimated from the reported figure and added as site years to the practice table, although not used.

(Owens et al., 1983b)
In a high-fertility study conducted in Ohio, where fertilization and grazing was described in Owens et al. (1983a), five watersheds were monitored for surface and subsurface discharge. Fertilizer was applied at 224 kg N/ha as ammonium nitrate (three separate doses). Two grazing programs were implemented – summer rotational grazing and winter grazing/feeding operation. The summer program had lower nitrate-N leaching concentrations with a range from around 2 mg $NO_3$-N/L to just under 10 mg $NO_3$-N/L, while the winter program ranged from just under 10 mg $NO_3$-N/L to around 18 mg $NO_3$-N/L. Data from the figure provided in the publication were estimated and added to the practice table as site-years for pasture although not used.

(Owens, 1990)
This study used percolate (leachate) from lysimeters to investigate cropping changes. Two scenarios were changing from continuous corn to a mix of alfalfa (70%) and orchard grass (30%). As expected, the cropping practice change took time to have an effect on nitrate-N leaching (approximately 1.5 years). From this research it appears it takes about the same amount of time for nitrate-N concentrations to increase to initial levels after changing back to continuous corn production. Nitrate-N concentrations in the publication were only displayed in figure format (below), but were generally around 1 or 2 mg $NO_3$-N /L. Data were not added to the practice table.

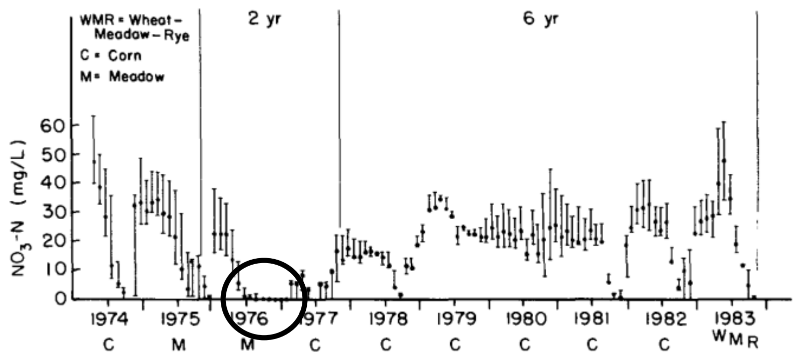



Fig. 2. Lysimeter Y102—Monthly ranges of flow-weighted $NO_3$–N concentration in percolate, 1974–1983. Dots represent average flow-weighted $NO_3$–N concentrations.

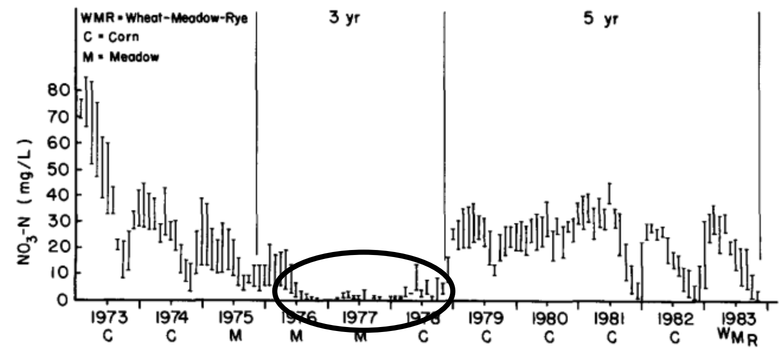



Fig. 3. Lysimeter Y103—Monthly ranges of flow-weighted $NO_3$–N concentration in percolate, 1973–1983. Dots represent average flow-weighted $NO_3$–N concentrations.

(Owens et al., 1992)
This follow-up study from the Owens et al. (1982) paper catalogues the same watersheds. The slow release nitrogen fertilizer treatments in that study will not be used here, although they don't appear to be different than the ammonium nitrate treatment. The site years for watershed 135 were estimated from the figure in the publication and added as site-years to the practice table for pasture. Fertilizer was added at 168 kg N/ha for this study. It is obvious the longer high fertilizer rates are added, the higher nitrate-N concentration in leachate becomes. Data were added to the practice table, but not used for average, max, or min computations as drainage patterns in Ohio tend to be different.

(Kaspar et al., 2008)
This paper summarizes research with perennial crops. Nitrogen leaching can be reduced by up to 90% with a perennial crop. Initial costs can be high, but reduced in years after establishment. Economic comparison was based on crop production. Possibly 20-30% of the current corn-soybean row crop acres could be converted to perennial crops "if infrastructure, processing facilities, and markets were encouraged and supported." This means the perennial crop practice is limited by demand for the product. A cost of $0.48 to $1.21 per pound of nitrogen reduced could be expected for a perennial alfalfa system. This paper was used as a reference, but data were not added to the practice table.

(Helmers, 2011b)
This data from a research site southwest of Ames, Iowa, compares switchgrass to conventional row crops. Only nitrate concentration in tile drainage from 2010 was available. Both fertilized and unfertilized switchgrass treatments were added as the nitrate concentrations were similar (0.16 mg $NO_3$-N/L and 0.55 mg $NO_3$-N/L, respectively). These data, although unpublished, were added to the practice table.

(Helmers, 2011a)
This data from the Bioenergy site west of Ames, Iowa, compares switchgrass (fertilized and unfertilized) to conventional row crops. The dataset from 2008 to 2010 includes results from both commercial fertilizer treatments and manure treatments. These data, although unpublished, were added to the practice table.

## Land Retirement (CRP)

Three studies were used for data entry into the practice table.

(Randall et al., 1997)
This paper, with research from southern Minnesota, reports yield, nitrate concentration, and subsurface drain flow for CRP and alfalfa. The two years (1992 and 1993) with adequate CRP yield data have CRP yields at 5250 and 5120 kg/ha, and alfalfa yields for 1990 through 1993 at 11610, 11900, 11480, 10270 kg/ha. Subsurface nitrate-N concentration in tile flow in 1991, 1992, and 1993 was reduced by 84%, 63%, and 34% for alfalfa, respectively, and 82%, 42%, and -5% for CRP, respectively, when compared to a corn-soybean rotation. Nitrate concentrations for 1991 through 1993 were reduced by 88%, 86%, and 90% for alfalfa, and 88%, 95%, and 98% for CRP, when compared to a corn-soybean rotation. Data were added to the practice table.

(Tomer et al., 2010)
This work in Walnut Creek, Iowa, compared a restored prairie watershed to an agricultural production watershed. Nitrate-N reductions were around 80% when compared to an agricultural watershed. Data from this study were added to the practice table.

(Qi et al., 2011)
This paper was summarized in the Cover Crops and Living Mulches practice sections. The research showed a 67 to 90% reduction in nitrate-N concentration in tile flow in a perennial vegetation system when compared to a corn-soybean rotation. The data were added to the practice table.

## Bioreactors

Only one study was reviewed as bioreactors are relatively new and effect on nitrate concentration reduction is heavily dependent on design considerations (sizing) (Schipper et al., 2010).

(Christianson, 2011)
This research evaluated four bioreactors in Iowa. Load reduction estimates were based on measured flow rates through the bioreactors and water samples before and after the bioreactor were analyzed for nitrate-N concentration. Nitrate reduction ranged from 12 to 75%. All available data were added to the practice table.

## Buffers

Buffers studies were reviewed differently from other practice studies as results depend on how much water moves through the root zone of the buffer system. In tile drained landscapes, little water may actually move through the buffer root zone as the tile shunts water through the buffer and outlets directly to the stream. Data from four studies were added to the practice table.

(Helmers et al., 2008b)
The interpretation section of this review paper indicated that costs for installation (as adopted from Qiu, 2003) amortized over a 10-year period resulted in a cost of $62.40 per acre per year. This paper was only used as a reference and data were not added to the practice table.

(Osborne and Kovacic, 1993)
This research was conducted in eastern Illinois in 1988 and 1989. The study setup included an entirely cropped area up to the stream, a cropped area with a forested buffer (16 m wide), and a cropped area with a grass buffer (39 m wide). Although drainage concentrations were not monitored, data from shallow and deep lysimeters, as well as piezometers, were reported in the paper and were added to the practice table. Results are averaged over two years (corn-soybean rotation), and were added double as site-years to maintain annual weighting. Data were estimated from the figure in the paper. Both buffer systems reduced nitrate-N concentrations from around 20 mg $NO_3$-N/L to less than 2 mg $NO_3$-N/L. Data were added to the practice table.

(Schoonover and Willard, 2003)
This paper reports research from southern Illinois conducted in 2000 and 2001. The research studied two riparian buffers (giant cane and forest), determining performance at distances away from a field of corn and soybean. Groundwater well data (wells between 3.5 and 4 m deep) were used to determine nitrate-N removal. Data was entered into the practice table as site-years, however, only the longest buffer lengths were used to determine removal rates (99.3% for the giant cane at 10 m and 81.7% for forest at 6.6 m). Data entered in the practice table were doubled for the corn-soybean rotation to maintain even annual weighting. Data were added to the practice table.

(Yamada et al., 2007)
This research was conducted near Treynor, Iowa, and compares groundwater and soil nitrate concentrations for a corn-soybean rotation, a switchgrass buffer, a smooth brome-alfalfa buffer, and a cottonwood-walnut buffer. This paper included groundwater nitrate concentrations for each location, however, only general information was obtainable from the figures in the paper and the tables provided were not helpful for more detailed data. Lysimeter data was available and was taken from a figure in the paper. These data were added to the practice table as site-years. Three years of monitoring was conducted. Although there were 4 treatments, the site layout was setup such that there was one buffer with a switchgrass, smooth brome-alfalfa, and tree segment. Estimated nitrate-N concentration reduction numbers were 86.3%, 92.0%, and 93.5% for 2003, 2004, and 2005, respectively, and are comparing the cropped land soil water to the soil water in the trees, after it has passed through switchgrass and brome-alfalfa. Data were added to the practice table.

(Spear, 2003)
This thesis reported results from three buffer field trials northeast of Ames, Iowa. One of the three sites (Risdal North), which was established prior to 1990, was a grass buffer 35 m in width. The other two (Risdal South and Strum) sites are both mixed buffers with grass, shrub, and tree components. Risdal South is 22 m wide and was established in 1990 while Strum is 17 m wide and was established in 1994. The thesis contains nitrate-N well concentrations from June 1996 to February 1999, but discussion in the thesis indicates removals are for July 1997 to December 1998. Each buffer was included as only 1 site year in the practice table. Nitrate-N concentration reductions for Risdal North, Risdal South, and Strum are 65.6%, 32.8%, and 48.6%, respectively.

This data was also reported in a proceedings abstract (Spear et al., 1998), however, it is not consistent with the above data, which is likely due to the fact the abstract reports data from August 1996 to August 1998. Risdal North is reported as having a nitrate-N concentration reduction of 75.8%. Risdal South is reported as having a nitrate-N concentration reduction of negligible (no numbers actually reported). Strum is reported as having a nitrate-N concentration reduction of 39.8%. Due to the preliminary nature of this data, the 2003 thesis data will be used instead and data were added to the practice table.

(Mayer et al., 2007)
This large literature review paper found that buffer width was a significant factor in performance, but also states:
"Overall, subsurface nitrogen removal is more efficient than removal through surface flow. Furthermore, subsurface nitrogen removal may be more directly influenced by soil type, watershed hydrology (e.g., soil saturation, groundwater flow paths, etc.), and subsurface biogeochemistry (organic carbon supply, high $NO_3^-$ inputs) through cumulative effects on microbial denitrification activity than on buffer width per se. Surface flows bypass zones of denitrification, and thus effectively remove nitrogen only when buffers are wide enough and have adequate vegetation cover to control erosion and filter movement of particulate forms of nitrogen. Herbaceous buffers, for example, may be better at intercepting particulate nitrogen in the sediments of surface runoff by reducing channelized flow. Based on a limited data set fitted to a log-linear model, Oberts and Plevan (2001) found that $NO_3^-$ retention in wetland buffers was positively related to buffer width ($R^2$ values ranged from 0.35–0.45). Nitrogen removal efficiencies of 65 to 75% and 80 to 90% were predicted for wetland buffers 15 and 30 m wide, respectively, depending on whether $NO_3^-$ was measured in surface or subsurface flow (Oberts and Plevan, 2001)."  Specific data were not added to the practice table.

## References

ADMC. 2011. Drainage Water Management for Midwestern Row Crop Agriculture. Agricultural Drainage Management Coalition, Owatonna, MN.

Albrecht, K. 2009. Midwest Cover Crop Council 2009 State/Province Report. Annual Report. University of Wisconsin-Madison, Madison, WI.

Baker, J., and S. Melvin. 1999. ADW Annual Report. Iowa State University, Ames, IA.

Bakhsh, A., R.S. Kanwar, and D.L. Karlen. 2005. Effects of liquid swine manure applications on NO3-N leaching losses to subsurface drainage water from loamy soils in Iowa. Agriculture, Ecosystems and Environment 109:118-128.

Bakhsh, A., R.S. Kanwar, T.B. Bailey, C.A. Cambardella, D.L. Karlen, and T.S. Colvin. 2002. Cropping system effects on NO3-N loss with subsurface drainage water. Transactions of the ASAE 45:1789-1797.

Barnhart, S.K., R.D. Voss, and J.R. Geroge. 1997. Fertilizing Pasture, pp. 1-6, *In* I. S. U. D. o. Agronomy, (ed.), Vol. Pm-869. Iowa State University: University Extension, Ames, IA.

Blackmer, A.M., R.D. Voss, and A.P. Mallarino. 1997. Nitrogen Fertilizer Recommendations for Corn in Iowa, *In* I. S. U. U. Extension, (ed.), Vol. Pm-1714. Iowa State University Extension, Ames, IA.

Chase, C., K. Delate, M. Liebman, and K. Leibold. 2008. Economic Analysis of Three Iowa Rotations, pp. 12. Iowa State University: University Extension, Ames, Iowa.

Chinkuyu, A.J., R.S. Kanwar, J.C. Lorimor, H. Xin, and T.B. Bailey. 2002. Effects of laying hen manure application rate on water quality. Transactions of the ASAE 45:299-308.

Christianson, L. 2011. Design and performance of denitrification bioreactors for agricultural drainage. PhD, Iowa State University, Ames, Iowa

Christianson, L., J. Tyndall, and M. Helmers. In Preperation. Financial Comparison of Seven Nitrate Reduction Strategies for Midwestern Agricultural Drainage.

Clover, M.W. 2003. Impact of nitrogen management on corn grain yield and nitrogen loss on a tile drained field. Master of Science, University of Illinois, Urbana-Champaign, IL.

Cooke, R., J. Nehmelman, and P. Kalita. 2002. Effect of Tile Depth on Nitrate Transport from Tile Drainage Systems. Proc. ASAE International Meeting, Chicago, IL2002. ASAE.

Dabney, S.M., J.A. Delgado, J.J. Meisinger, H.H. Schomberg, M.A. Liebig, T. Kaspar, J. Mitchell, and W. Reeves. 2011. Using Cover Crops and Cropping Systems for Nitrogen Management, p. 230-281, *In* J. A. Delgado and R. F. Follett, (eds.) Advances in Nitrogen Management for Water Quality. ed. Soil and Water Conservation Society, Ankeny, IA.

David, M., L. Drinkwater, and G. Mclsaac. 2010. Sources of Nitrate Yields in the Mississippi River Basin. (in English) Journal of Environmental Quality 39:1657-1667.

David, M.B., L.E. Gentry, D.A. Kovacic, and K.M. Smith. 1997. Nitrogen balance in and export from an agricultural watershed. Journal of Environmental Quality 26:1038-1048.

Dinnes, D.L., D.L. Karlen, D.B. Jaynes, T.C. Kaspar, J.L. Hatfield, T.S. Colvin, and C.A. Cambardella. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils. Agronomy Journal 94:153-171.

Doering, O.C.I., J.N. Galloway, T.L. Theis, and D.L. Swackhamer. 2011. Reactive Nitrogen in the United States: An Analysis of Inputs, Flows, Consequences, and Management Options. EPA-SAB-11-013. US Environmental Protection Agency, Washington D.C.

Dow AgroSciences. 2012. N-Serve Use in Iowa, *In* J. Sawyer, (ed.), Email Communication from Eric Scherder at Dow ed. Dow AgroSciences.

Duffy, M. 2008. Estimated Costs for Production, Storage and Transportation of Switchgrass, *In* I. S. University, (ed.), Vol. A1-22. Iowa State University University Extension, Ames, Iowa.

Duffy, M. 2011. Production Costs for Alfalfa or Alfalfa-grass Hay (A1-15-20haybudgetonly-2). Available at http://www.extension.iastate.edu/agdm/crops/xls/a1-15-20haybudgetonly.xls. Iowa State University Extension, Ames, IA.

Edwards, W., and A. Johanns. 2011b. Cash Rental Rates for Iowa 2011 Survey. Iowa State University: University Extension, Ames, IA.

Edwards, W., A. Johanns, and A. Chamra. 2011. 2011 Iowa Farm Custom Rate Survey, *In* I. S. University, (ed.), Vol. A3-10. Iowa State University, Ames, IA.

Ellsworth, J., K. Balkcom, and A.M. Blackmer. 1999. Fertilization to rescue corn crops following losses of fall nitrogen. p. 301-304 Proc. Integrated Crop Management, Ames, IA1999. Iowa State University.

Erickson, B. 2008. Corn/soybean rotation literature summary.

Frankenberger, J., Kladivko, E., Sands, G., Jaynes, D.B., Fausey, N.R., **Helmers, M.**, Cooke, R., Strock, J., Nelson, K., Brown, L. 2006. Drainage Water Management for the Midwest. Purdue Extension, Knowledge to Go. WQ-44. [Peer-reviewed by outside drainage experts]

Goolsby, E.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and F.J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya river basin: Topic 3. Report for the integrated assessment of hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program, Silver Spring, MD.

Hanna, M., and W. Edwards. 2007. Fieldwork Days in Iowa, *In* I. S. University, (ed.), Vol. PM-1874. Iowa State University University Extension, Ames, Iowa.

Helmers, M., R.D. Christianson, G. Brenneman, D. Lockett, and C. Pederson. 2010. Water table response to drainage water management in southeast Iowa. Proc. XVII World Congress of the International Commission of Agricultural Engineering (CIGR), Quebec City, Canada2010. Canadian Society for Bioengineering.

Helmers, M., W. Crumpton, P. Lawlor, C. Pederson, G. Stenback, R. Christianson, and D. Green. 2008a. Water and Nutrient Research: In-field and Offsite Strategies. Iowa State University, Ames, IA.

Helmers, M.J. 2011a. BE nitrate and flow data Unpublished.

Helmers, M.J. 2011b. COBS nitrate and flow data Unpublished Data.

Helmers, M.J., T.M. Isenhart, M.G. Dosskey, S.M. Dabney, and J.S. Strock. 2008b. Chapter 4 - Buffers and vegetative filter strips, p. 43-58 Final report : gulf hypoxia and local water quality concerns workshop. ed. American Society of Agricultural and Biological Engineers, Saint Joseph.

Huggins, D.R., G.W. Randall, and M.P. Russelle. 2001. Subsurface Drain losses of water and nitrate following conversion of perennials to row crops. Agronomy Journal 93:477-486.

IDALS. 2011. Fertilizer Distribution Report. Available at http://www.iowaagriculture.gov/feedAndFertilizer/fertilizerDistributionReport.asp (accessed August 19). Iowa Department of Agriculture and Land Stewardship.

Iowa State University: Agronomy Extension. 2011. N-Rate Calculator. Release 1.5. Iowa State University: Agronomy Extension, Ames, IA.

Iowa State University. 2011. Low-till drilled soybeans following corn. Iowa State University, Ames, Iowa.

Iowa State University Extension. 2011a. Ag. Decision Maker: Cash Rental Rate Estimation. Release 2011. Iowa State University Extension, Ames, IA.

Jaynes, D. 2009. Crop Yield and Nitrate Losses from Sidedressing N at Canopy Closure - Poster. Agricultural Science Association.

Jaynes, D. 2011. Oats as a cover crop, *In* R. Christianson, (ed.), Dan made a statement about using oats as a cover crop before corn and how there is no impact on crop yield but the water quality benefits relating to nitrate-N are only about half that of using rye as a cover crop. ed.

Jaynes, D., T. Colvin, D. Karlen, C. Cambardella, and D. Meek. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. (in English) Journal of Environmental Quality 30:1305-1314.

Jaynes, D.B., and T.S. Colvin. 2006. Corn yield and nitrate loss in subsurface drainage from midseason nitrogen fertilizer application. Agronomy Journal 98:1479-1487.

Jaynes, D.B., J.L. Hatfield, and D.W. Meek. 1999. Water quality in Walnut Creek watershed: Herbicides and nitrate in surface waters. Journal of Environmental Quality 28:45-59.

Jaynes, D.B., D.L. Dinnes, D.W. Meek, D.L. Karlen, C.A. Cambardella, and T.S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. Journal of Environmental Quality 33:669-677.

Kanwar, R.S., D.L. Karlen, and R.M. Cruse. 1995. Swine manure and N-management systems: Impact on groundwater quality. p. 91-94 Proc. Clean Water - Clean Environment - 21st Century: Team Agriculture - Working to protect water resources, Kansas City, MO1995. ASAE.

Kanwar, R.S., R.M. Cruse, M. Ghaffarzadeh, A. Bakhsh, D.L. Karlen, and T.B. Bailey. 2005. Corn-soybean and alternative cropping systems effects on NO3-N leaching losses in subsurface drainage water. Applied Engineering in Agriculture 21:181-188.

Kaspar, T., D. Jaynes, T. Moorman, and T. Parkin. 2003. Reducing nitrate levels in subsurface drain water with organic matter incorporation. Final Report to the American Farm Bureau Foundation for Agriculture. USDA-ARS National Soil Tilth Laboratory, Ames, IA.

Kaspar, T.C., and J.W. Singer. 2011. The Use of Cover Crops to Manage Soil, p. 321-337, *In* J. L. Hatfield and T. J. Sauer, (eds.) Soil Management: Building a Stable Base for Agriculture. ed. American Society of Agronomy and Soil Science Society of America, Madison, WI.

Kaspar, T.C., D.B. Jaynes, T.B. Parkin, and T.B. Moorman. 2007. Rye cover crop and gamagrass strip effects on NO3 concentration and load in tile drainage. Journal of Environmental Quality 36:1503-1511.

Kaspar, T.C., D.B. Jaynes, T.B. Parkin, T.B. Moorman, and J.W. Singer. 2012. Effectiveness of oat and rye cover crops in reducing nitrate losses in drainage water. Agricultural Water Management 110:25-33.

Kaspar, T.C., E.J. Kladivko, J.W. Singer, S. Morse, and D.R. Mutch. 2008. Chapter 10 - Potential and limitations of cover crops, living mulches, and perennials to reduce nutrient losses to water sources from agricultural fields in the uppper Mississippi River Basin, p. 129-148 Final Report: gulf hypoxia and local water quality concerns workshop. ed. American Society of Agricultural and Biological Engineers, Saint Joseph.

Lawlor, P.A., M.J. Helmers, J.L. Baker, S.W. Melvin, and D.W. Lemke. 2008. Nitrogen application rate effect on nitrate-N concentration and loss in subsurface drainage for a corn-soybean rotation. Transactions of the ASABE 51:83-94.

Lawlor, P.A., M.J. Helmers, J.L. Baker, S.W. Melvin, and D.W. Lemke. 2011. Comparison of liquid swine manure and ammonia nitrogen application timing on subsurface drainage water quality in Iowa.

Libra, R.D., C.F. Wolter, and R.J. Langel. 2004. Nitrogen and Phosphorus Budgets for Iowa and Iowa Watersheds. Iowa Department of Natural Resources, Iowa City, IA.

Liebman, M., L.R. Gibson, D.N. Sundberg, A.H. Heggenstaller, P.R. Westerman, C.A. Chase, R.G. Hartzler, F.D. Menalled, A.S. Davis, and P.M. Dixon. 2008. Agronomic and Economic Performance Characteristics of Conventional and Low-External-Input Cropping Systems in the Central Corn Belt. Agronomy Journal 100:600-610.

Lundvall, J. 2011. Cost of aerial seed application, *In* R. Christianson, (ed.), Conversation between John and Reid about the costs associated with flying on cover crop seed. ed.

Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. Journal of Environmental Quality 36:1172-1180.

Nelson, D.W., and D.M. Huber. 1980. Performance of nitrification inhibitors in the midwest (east), p. 75-88, *In* S. Matthias, et al., (eds.) Nitrification Inhibitors - Potentials and Limitations - ASA Special Publication Number 38. ed. American Society of Agronomy Soil Science Society of America, Madison, Wisconsin.

Nelson, D.W., L.E. Sommers, D.M. Huber, and H.L. Warren. 1977. Conserving energy with nitrification inhibitors, p. 361-376, *In* W. Lockeretz, (ed.) Agriculture and Energy. ed. Academic Press, New York.

Osborne, L.L., and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258.

Owens, L.B. 1987. Nitrate leaching losses from monolith lysimeters as influenced by nitrapyrin. Journal of Environmental Quality 16:34-38.

Owens, L.B. 1990. Nitrate-N concentrations in percolate from lysimeters planted to a legume-grass mixture. Journal of Environmental Quality 19:131-135.

Owens, L.B., R.W. Van Keuren, and W.M. Edwards. 1982. Environmental effects of a medium-fertility 12-month pasture program: II. Nitrogen. Journal of Environmental Quality 11:241-246.

Owens, L.B., R.W. Van Keuren, and W.M. Edwards. 1983a. Hydrology and soil loss from a high-fertility, rotational pasture program. Journal of Environmental Quality 12:341-346.

Owens, L.B., R.W. Van Keuren, and W.M. Edwards. 1983b. Nitrogen loss from a high-fertility, rotational pasture program. Journal of Environmental Quality 12:346-350.

Owens, L.B., W.M. Edwards, and R.W. Van Keuren. 1992. Nitrate levels in shallow groundwater under pastures receiving ammonium nitrate or slow-release nitrogen fertilizer. Journal of Environmental Quality 21:607-613.

Pederson, C., R. Kanwar, M. Helmers, and A. Mallarino. 2010. Impact of Liquid Swine Manure Application and Cover Crops on Groundwater Water Quality. Iowa State University, Ames, IA.

PFI. 2011. Cover Crop Effect on Cash Crop Yield: Year 2. Practical Farmers of Iowa, Ames, IA.

Qi, Z., and M.J. Helmers. 2008. Effect of cover crops in reducing nitrate-N leaching in Iowa. p. 283-294 Proc. Integrated Crop Management Conference, Ames, Iowa2008. Iowa State Univeristy.

Qi, Z., M.J. Helmers, R.D. Christianson, and C.H. Pederson. 2011. Nitrate-N Losses through Subsurface Drainage under Various Agricultural Land Covers. Journal of Environmental Quality 40:1578.

Rakshit, S. 2002. Liquid swine manure as a nitrogen source for corn and soybean production. Soil Science, Iowa State University, Ames, Iowa.

Randall, G. 2008. Managing nitrogen for optimum profit and minimum environmental loss Integrated Crop Management. ed. Iowa State University, Ames.

Randall, G., and J. Vetsch. 2005a. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by fall and spring application of nitrogen and nitrapyrin. (in English) Journal of Environmental Quality 34:590-597.

Randall, G.W., and D.J. Mulla. 2001. Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. Journal of Environmental Quality 30:337-344.

Randall, G.W., and J.A. Vetsch. 2005b. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by fall and spring application of nitrogen and nitrapyrin. (in English) Journal of Environmental Quality 34:590-597.

Randall, G.W., and J.A. Vetsch. 2005c. Corn Production on a subsurface-drained mollisol as affected by fall versus spring application of nitrogen and nitrapyrin. Agronomy Journal 97:472-478.

Randall, G.W., and J.E. Sawyer. 2008. Chapter 6 - Nitrogen application timing, forms, and additives, p. 73-85 Final report : gulf hypoxia and local water quality concerns workshop. ed. American Society of Agricultural and Biological Engineers, Saint Joseph.

Randall, G.W., J.A. Vetsch, and J.R. Huffman. 2003a. Corn production on a subsurface-drained mollisol as affected by time of nitrogen application and nitrapyrin. Agronomy Journal 95:1213-1219.

Randall, G.W., J.A. Vetsch, and J.R. Huffman. 2003b. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by time of nitrogen application and use of nitrapyrin. Journal of Environmental Quality 32:1764-1772.

Randall, G.W., D.R. Huggins, M.P. Russelle, D.J. Fuchs, W.W. Nelson, and J.L. Anderson. 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. Journal of Environmental Quality 26:1240-1247.

Ruiz Diaz, D., and J. Sawyer. 2008. Plant-available nitrogen from poultry manure as affected by time of application. (in English) Agronomy Journal 100:1318-1326.

Ruiz Diaz, D., J. Sawyer, and A. Mallarino. 2011. On-Farm Evaluation of Poultry Manure as a Nitrogen Source for Corn (vol 75, pg 729, 2011). (in English) Soil Science Society of America Journal 75:1176-1176.

Ruiz Diaz, D.A., J.A. Hawkins, J.E. Sawyer, and J.P. Lundvall. 2008. Evaluation of in-season nitrogen management stategies for corn production. Agronomy Journal 100:1711-1719.

Sands, G.R., I. Song, L.M. Busman, and B.J. Hansen. 2008. The effects of subsurface drainage depth and intensity on nitrate loads in the northern cornbelt. Transactions of the ASABE 51:937-946.

Sawyer, J. 2011. N-Serve Nitrification Inhibitor Price, *In* R. Christianson, (ed.), Email sent from John indicating a conversation with a chemical dealer. ed.

Sawyer, J., J. Pantoja, and D. Barker. 2011a. Nitrogen Fertilization of Corn Grown with Cover Crop. Iowa State University, Ames, IA.

Sawyer, J., E. Nafziger, J. Camberato, K. Steinke, J. Lamb, and C. Laboski. 2011b. N-Rate Calculator. Available at http://extension.agron.iastate.edu/soilfertility/nrate.aspx (accessed 11-4-2011). Iowa State University: Agronomy Extension, Ames, IA.

Sawyer, J.E., and A.P. Mallarino. 2008a. Using Manure Nutrients for Crop Production, *In* I. S. U. U. Extension, (ed.), Vol. PMR-1003. Iowa State University Extension, Ames, IA.

Sawyer, J.E., and A.P. Mallarino. 2008b. Using manure nutrients for crop production. Extension Publication. Iowa State University: University Extension, Ames, IA.

Sawyer, J.E., A.P. Mallarino, R. Killorn, and S.K. Barnhart. 2011c. A General Guide for Crop Nutrient and Limestone Recommendations in Iowa, *In* I. S. U. U. Extension, (ed.). Iowa State University Extension, Ames, IA.

Sawyer, J.E., P. Pedersen, D.W. Barker, D.A.R. Diaz, and K. Albrecht. 2010. Intercropping corn and kura clover: response to nitrogen fertilization. Agronomy Journal 102:568-574.

Schipper, L.A., W.D. Robertson, A.J. Gold, D.B. Jaynes, and S.C. Cameron. 2010. Denitrifying bioreactors-An approach for reducing nitrate loads to receiving waters. Ecological Engineering 36:1532-1543.

Schoonover, J.E., and K.W.J. Willard. 2003. Ground water nitrate reduction in giant cane and forested riparian buffer zones. Journal of the American Water Resources Association 39:347-354.

Singer, J. 2011. Cover Crops: Why, How, and How Much?, *In* R. Christianson, (ed.), A copy of a presentation given by Jeremy ed.

Snapp, S.S., S.M. Swinton, R. Labarta, D. Mutch, J.R. Black, R. Leep, J. Nyiraneza, and K. O'Neil. 2005. Evaluating cover crops for benefits, costs and performance within cropping system niches. Agronomy Journal 97:322-332.

Spear, B.A. 2003. Fate and transport of nitrate in groundwater through three multi-species riparian buffers along Bear Creek in central Iowa. MS Thesis. Iowa State University, Ames, IA.

Spear, B.A., W.W. Simpkins, T.B. Parkin, T.M. Isenhart, and R.C. Schultz. 1998. Fate and transport of nitrate in groundwater in a constructed riparian bufer in central Iowa. p. A 120 Proc. Geological Society of America - 1998 Annual Meeting and Exposition, Toronto, Ontario1998. GSA.

Strock, J.S., P.M. Porter, and M.P. Russelle. 2004. Cover cropping to reduce nitrate loss through subsurface drainage in the northern U.S. corn belt. Journal of Environmental Quality 33:1010-1016.

Tomer, M. 2011. Extended Rotation Nitrate-N Reduction, *In* D. Jaynes, (ed.), Verbal communication between Mark Tomer and Dan Jaynes ed.

Tomer, M., K. Schilling, C. Cambardella, P. Jacobson, and P. Drobney. 2010. Groundwater nutrient concentrations during prairie reconstruction on an Iowa landscape. (in English) Agriculture Ecosystems & Environment 139:206-213.

Tonitto, C., M.B. David, and L.E. Drinkwater. 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. Agriculture Ecosystems & Environment 112:58-72.

United States. National Agricultural Statistics Service. 2007. 2002 census of agriculture., pp. 1 online resource, Vol. 1 - 2. U.S. Dept. of Agriculture, National Agricultural Statistics Service, [Washington, D.C.].

United States. National Agricultural Statistics Service. 2009. 2007 census of agriculture, pp. 1 CD-ROM, Vol. 1-2. U.S. Dept. of Agriculture, National Agricultural Statistics Service, [Washington, D.C.].

Vetsch, J.A., and G.W. Randall. 2004. Corn Production as Affected by Nitrogen Application Timing and Tillage. Agronomy Journal:502-509.

Wolt, J.D. 2004. A meta-evaluation of nitrapyrin agronomic and environmental effectiveness with emphasis on corn production in the midwestern USA. Nutrient Cycling in Agroecosystems 69:23-41.

Yamada, T., S.D. Logsdon, M.D. Tomer, and M.R. Burkart. 2007. Groundwater nitrate following installation of a vegetated riparian buffer. Science of the Total Environment 385:297-309.

Zemenchik, R.A., K.A. Albrecht, C.M. Boerboom, and J.G. Lauer. 2000. Corn production with kura clover as a living mulch. Agronomy Journal 92:698-705.

## Section 2.3

## Iowa Science Assessment of Nonpoint Source Practices to Reduce Phosphorus Transport in the Mississippi River Basin

Prepared by the Phosphorus Science Team
May 2013

## Table of Contents

## Introduction

In late 2010, the Iowa Department of Agriculture and Land Stewardship and the College of Agriculture and Life Sciences at Iowa State University partnered to develop a statewide nutrient-loss reduction strategy for Iowa. A science team consisting of 23 individuals representing five agencies or organizations was formed to determine nitrogen (N) and phosphorus (P) reduction practices that have the greatest potential to reduce the Iowa contribution of N and P to the Mississippi River. Additionally, these practices should reduce nutrients delivered to local lakes and streams. Subgroup teams were formed to focus on N and P. This report summarizes the work of the P team.

Phosphorus is one of three primary nutrients for plant (crop) production along with nitrogen (N) and potassium (K), and therefore needs to be managed for agronomic production. Additionally, P is generally the limiting nutrient for algal production in fresh water systems (Schindler et al., 2008; Schindler, 1971), meaning the addition of P to fresh water can lead to eutrophication. Eutrophication has a negative impact on aquatic ecosystems by limiting oxygen available for aquatic species. Recently, the importance of P in the development of spring and summer hypoxia in the Gulf of Mexico has been realized (USEPA, 2007), with supporting work by Sylvan et al. (2006), hypothesizing when and why P can be the limiting nutrient in this system.

Much of the P being delivered to surface water resources is from nonpoint sources via agricultural runoff (Jacobson et al., 2011) and/or streambank erosion (Zaimes et al., 2008a; Zaimes et al., 2008b), although under some conditions loss through subsurface tile drains can be significant. Most P in runoff is sediment bound (Jacobson et al., 2011), 70% of the total P delivered to streams near agricultural fields (Mallarino and Wittry, 2005). However, dissolved P delivery to streams and lakes also is significant, especially in soils with high soil-test P (STP) levels or from soils with surface application of high rates of liquid swine manure or inorganic P fertilizers (Kleinman et al., 2002; Sharpley et al., 2002; Tabbara, 2003; Allen and Mallarino, 2008). Additionally, dissolved P is more readily available for biological uptake, and therefore has a potentially larger impact on eutrophication than sediment-attached forms of P. Phosphorus dissolved in stream water can be heavily influenced by the land immediately adjacent to the stream (Gburek and Heald, 1974; Gburek and Sharpley, 1998; Hongthanat et al., 2011). Although the sediment movement and delivery process is complex, sediment delivery is generally greatest from unprotected (bare) soils through erosion.

The P evaluation primarily focused on practices that limit or control P losses from agricultural land, and does not include known sources of P such as point sources, leaking rural septic systems, and streambank erosion. Although point sources (i.e., sewage treatment plants) may be substantial (30-40%) (USEPA, 2007), further research is needed on P reduction techniques for agricultural systems. Streambanks are known to be a potentially large source of stream sediment, with contributions ranging from approximately 40 to 80% of annual sediment loads in many Midwestern streams (Schilling et al., 2011; Sekely et al., 2002; Wilson et al., 2008). However, accurately accounting for streambank sources of P is extremely difficult and methods have not been developed to quantify streambank sediment contributions beyond a local scale. Therefore, evaluating strategies to reduce P losses from point sources and eroding streambanks (i.e., runoff volume reduction or bank stabilization) are beyond the scope of this effort.

Included in this document are results of the first step of evaluation from the P team. The initial work was done to determine practices expected to have the most potential for cost effective reduction of P export from sheet and rill erosion. The science team assembled a list of potential practices that offered the greatest P loss reductions, and the P subgroup team refined the list based on practices expected to have the greatest potential impact. The overall group then reviewed the list of practices and provided additional input.

The P team benefitted from previous work that resulted in the development of the Iowa P Index (Mallarino et al., 2002; NRCS, 2004). The assessment methodology adapted the Iowa P Index to estimate P-delivery from the major land resource areas (MLRAs) in the state. Although only portions of the Iowa P Index have been validated with water quality data, no other P transport model or risk assessment tool has been validated for Iowa or similar conditions. Literature was reviewed to ensure that P Index estimates were reasonable and to fill gaps in the model as needed. The Iowa P Index is a quantitative risk assessment tool that was developed to estimate P delivered from fields to the nearest stream by considering several factors in a multiplicative way within three P delivery pathways. These pathways are particulate, or sediment bound, P loss through erosion, dissolved P loss through surface runoff, and total P loss through subsurface drainage. The sum of the estimated P loss for each component provides an estimate of total P loss. The P team feels comfortable using the model in the manner described in this document to obtain acceptable estimates of P delivery from larger areas. Great care was taken to appropriately consider the implementation of P, soil, and conservation practices as they relate to a particular MLRA.

The P reduction practices considered have a range of implementation and treatment scales, and fall into three main groups: P management practices, erosion control and land use change, and edge-of-field practices.

- The P management practices considered focus on the most effective at reducing P loss and efficient use of P, including P application rate, P source (commercial fertilizer, liquid swine manure, and poultry manure), maintenance of optimum STP levels for crop production, and P placement.
- The intent of the land use options is primarily to reduce soil erosion. Examples include changing tillage practices; adding terraces, sediment control structures (basins or ponds); adding cover crops (i.e., rye) or a living mulch to the row crop system (i.e., growing kura clover with continuous corn); moving from a corn-soybean rotation to a 4- to 5-year rotation including alfalfa in the corn-soybean row cropping or to perennial crops used for energy production (i.e., switchgrass for ethanol); and land retirement [i.e., Conservation Reserve Program (CRP)], and converting row crop land to pasture.
- Edge-of-field technologies are designed primarily to remove sediments, or, in some cases, to capture dissolved P. They provide opportunities to remove P either in combination with the above practices or as stand-alone P reduction strategies. These practices include wetlands (targeted for water quality enhancement), and vegetated buffers along streams.

## Phosphorus Reduction Practices

Appropriate literature was reviewed (see "Appendix – Summary of Literature Reviewed") to determine the applicability of the listed practices and the likely benefit/detriment of implementation. Since this is an effort focused on the State of Iowa, most of the studies selected for evaluation were conducted in or around Iowa because most P delivery processes often are region specific due to predominant landforms, soils, hydrology, precipitation, and freeze/thaw patterns. Practices were compared to the most common management practices used in Iowa, which include a corn-soybean rotation with the P needed by the two crops surface-applied once after soybean harvest in the fall before soils freeze or snowfall occurs. Tillage includes chisel plowing cornstalks after harvest and disking/field cultivating in the spring before planting soybean. Before planting corn the normal practice is disking/field cultivating in the spring. Therefore, in this "normal practice" scenario, the P applied in the fall after soybean harvest is incorporated in spring when disking/field cultivating soil before planting corn.

The order of practices in the text below or in Table 1 does not represent a prioritized list, and is organized into P management, erosion control and land-use change, and edge-of-field practices. There are wide performance ranges for all practices with spatial, temporal, and climactic influences that are not directly considered here. Therefore, the minimum, maximum, and average (arithmetic mean) values, with the standard deviation, are presented in Table 1. Large standard deviations indicate large variation in the effectiveness of practices, with some practices being effective in reducing P loss for some situations, but ineffective in others. Much of the literature reviewed for this summary was from rainfall simulation studies, in which the effects of practices sometimes are over-estimated. See Appendix – Summary of Literature Reviewed for more information about specific literature reviewed.

## Phosphorus Management

### Phosphorus Application Rate and Timing

Research suggests that, in practice, P rate is less important than N rate as it affects water quality. The P rate affects the STP level, both in the short and long-term, with a small to moderate but long-term impact on annual P loss. Applied P quickly binds to soil particles in most Iowa soils and, unless there is significant soil erosion, only a small portion is available for runoff loss as dissolved P, except for runoff events occurring within a few days of surface P application (Allen and Mallarino, 2008; Tabbara, 2003). Key P management issues for crop production involve knowing the optimum STP level, applying P to avoid deficiencies, and achieving the optimum soil-test level over time by using various strategies that consider fertilization rates and the frequency of application. Therefore, in most fields, the fertilizer P application rates being used are those that maintain STP levels farmers want to maintain, largely based on estimated P removal. The soil-test levels being maintained often exceed those recommended by Iowa State University, however, which explains the high proportion of soils testing high and very high in the state as suggested by soil test summaries (Mallarino et al., 2011a). In practice, therefore, the historical P application rates and current STP level a farmer maintains is a most important and relevant issue for the economics of P management and impacts on water quality. The rate of P application becomes of great concern, however, when manure is applied for disposal purposes, when any manure type is applied at N-based rates to continuous corn, and when poultry manure (which often has a lower N/P ratio) is applied at N-based rates for corn after soybean or continuous corn. In these cases, there is the short-term direct effect of P rate on P runoff loss and also the long-term effect through excessive soil P increase.

### Soil-Test Phosphorus Level

Since a large portion of P loss is associated with erosion (sediment bound P or dissolved P in surface runoff), the amount of P applied to the soil and its effect on STP and total soil P has a significant impact on the total P loss from a field. Phosphorus loss can be reduced by decreasing the total soil P concentration, which means limiting or stopping P application to high-testing soils until STP is lowered to agronomically optimum concentrations. This practice does not reduce erosion, only the amount of sediment-bound and dissolved P lost.

### Site-Specific Phosphorus Management

Agricultural fields are becoming larger, and research shows large within-field variability concerning soil types, erosion risk, crop yield, P removal with harvest, and STP levels along with many other properties. Therefore, site-specific management that considers the P loss risk from different areas of a field could be a beneficial practice to reduce P loss, depending on the degree of variability present. The potential for site-specific management to reduce risk of P loss is not well studied, but on-farm research in Iowa has found variable-rate fertilizer and manure P application to be effective in reducing within field variability of STP levels (Bermudez and Mallarino, 2007; Mallarino and Wittry, 2010; Wittry and Mallarino, 2004). Therefore, variable-rate P application is expected to reduce P loss from fields compared with a uniform application based on the average STP level for a field.

**Source**

There is little evidence of P source (i.e., fertilizer compared to manure P) effects on short-term P delivery from fields if the P is incorporated into the soil. In the long term, however, manure compared with inorganic P forms can reduce runoff (Gilley and Risse, 2000; Gessel et al., 2004) by increasing soil organic carbon and improving soil structure. If runoff-producing rainfall events occur immediately after P application, significantly less P loss occurs with solid beef and poultry manure, compared with commercial fertilizer (Mallarino and Haq, 2007 and 2008).

**Placement**

Placing P in the plant root zone can increase P availability and allow for reduced application rates in some conditions, but extensive research has shown this is not the case in Iowa soils. Also, long term Iowa research shows that applying similar rates of broadcast or planter-band P results in similar STP levels. On the other hand, subsurface banding of P or incorporation of surface-applied P fertilizer or manure on sloping ground reduces P loss significantly compared with surface application when runoff-producing precipitation occurs within a few days or weeks of the application.

**Tillage**

Tillage practices affect soil erosion, which is the primary transport process of P delivery in Iowa. Increased tillage reduces ground cover by crop residues, exposing more soil to raindrop splash effects that contribute to sheet erosion. Some forms of tillage reduce soil aggregate stability, resulting in increased break-up of aggregates during rainfall events, increasing erodibility and reducing permeability of surface soil. Tillage effects on P loss are site specific, but less P loss generally occurs with minimum or no tillage than with conventional tillage, although no-till can increase the proportion of total P lost as dissolved P, especially in tile drained areas.

**Cover Crops**

Cover crops reduce soil erosion by improving soil structure, stability, and permeability in addition to providing ground cover as a physical barrier between raindrops and the soil surface. Cover crops can be seeded in the fall using a variety of methods including drilling after crop harvest, broadcasting after crop harvest, or aerially broadcasting before harvest. Because of the Iowa climate and mainly corn-soybean production systems, fall growth of cover crops is very limited. Although often there may be poor germination with aerial application, this seeding method and timing has potential for extending the growing season of the cover crop by seeding before row crop harvest. The effectiveness of cover crops in reducing erosion is related to the soil cover achieved, which is generally greater with early compared to late sowing for both fall and spring sowing. This cover is most important in the spring, however, when most runoff events occur. Termination of a winter rye cover crop two weeks before planting corn reduces the negative impact on corn growth and yield. However, the research summary indicates an average 6% reduction in corn yield following a rye cover crop. Soybean yield is not affected by winter rye cover crops, which can continue growing longer in the spring to provide more protection against erosion. Corn yield reduction has been small, if any, with oat as a cover crop.

## Land Use Change

**Sediment Control**

Numerous erosion and sediment delivery control practices can be appropriate at the field or sub-field scale to reduce sediment delivery. These include terraces (with multiple design criteria), grassed waterways to reduce gully erosion, water and sediment control basins to capture sediment in waterways, and ponds.

Ponds can be effective at removing sediment (and P), but generally are not built for this purpose in the agricultural setting. Some of these structures also may be located at field edges.

### Crop Choice (Extended Rotation)

For Iowa, an extended rotation can be defined as a rotation of corn, soybean, and at least three years of alfalfa or legume-grass mixtures managed for hay harvest. The P loss reduction with alfalfa or a legume-grass mixture in the rotation is associated with reduced soil erosion because of greater soil cover, and also higher P removal with hay than with corn grain or soybean seed. There is very little concurrent P loss and corn yield data for specific extended rotations compared to a corn-soybean rotation in Iowa, but much information is available for crop rotation effects on erosion.

### Perennial Energy Crops

Several perennial crops, such as switchgrass, produce biomass that can be used as a bio-energy feedstock. Demand for and production of these crops still is small and localized in Iowa, but the acreage is likely to increase. These crops improve soil physical properties, provide good soil cover, reduce erosion, and reduce P loss.

### Grazed Pastures

There are substantial areas of Iowa, especially in southern counties, in permanent pasture. Although there is little research comparing P loss from pasture and corn-soybean rotation in Iowa, pastures typically have lower soil erosion rates than a corn-soybean rotation on comparable land but higher dissolved P concentration in runoff because of fertilizer application and fecal P on the soil surface. Delivery of P to water bodies is highly affected by pasture management. Phosphorus delivery is greater with excessive and prolonged over-grazing and with unrestricted animal access to streams, compared with intensively managed rotational grazing and restricted animal access to streams.

### Land Retirement

The Conservation Reserve Program (CRP) is a long-term (10-15 year) perennial vegetation program intended to limit soil erosion. The established vegetation is a near "natural" system that has plant and animal habitat and soil improvement benefits that should result in reduced P loss.

## Edge-of-Field

### Wetlands (Targeted for Water Quality)

The performance of installed wetlands depends on the wetland-to-watershed ratio (wetland area compared to watershed area) with larger ratios having a greater impact on P removal. Several factors are involved with implementation of wetlands and their effectiveness, including land cost and availability and level of sediment P loading. Eventually, the effectiveness of wetlands for removing P declines due to P saturation. Wetlands installed or restored specifically for habitat benefit also may result in reduced P delivery to water bodies.

### Sediment Control

Several sediment delivery control practices are appropriate for edge-of-field to reduce sediment delivery. These include water and sediment control basins to capture sediment from a field or wetlands.

### Vegetative Buffers

A buffer is a vegetated area strategically placed between cropland and a stream or other water body, which acts as a filter. Buffers can have plant and animal habitat benefits, but a primary role is to reduce P delivery from fields to water bodies by removing particulate P from runoff water through filtration and sedimentation and removing dissolved P by plant uptake or soil binding. Riparian buffers also can reduce P delivery to water bodies by stabilizing stream banks.

## Performance of Phosphorus Loss Reduction Practices

The effectiveness of practices (Table 1) in reducing P loss and their effect on corn yield were evaluated based on research results. For consistency, individual years of data (site years) were extracted from the reviewed studies to allow for direct comparisons. Large variations in P reduction and yield effects were found for most practices, and the minimum and maximum values are reported. The average reported values were determined from the multiple available observations. Specific methods for calculating the values are described below. Great care was taken to ensure appropriate comparisons were being made from each study.

**Table 1. Practices with the largest potential impact on phosphorus load reduction. Corn yield impacts associated with each practice also are shown, since some practices may increase or decrease corn production. See text for information on value calculations.**

| | Practice | Comments | % P Load Reduction[a] | | | % Corn Yield Change[b] | | |
|---|---|---|---|---|---|---|---|---|
| | | | Min | Average (SD[c]) | Max | Min | Average (SD[c]) | Max |
| Phosphorus Management Practices | Phosphorus Application | Applying P based on crop removal - Assuming optimal STP level and P incorporation | 0[d] [0[e]] | 0.6[d] [70[e]] | 1.3[d] [83[e]] | | 0[f] | |
| | | Soil-Test P – No P applied until STP drops to optimum | 0[g] [35[h]] | 17[g] [40[h]] | 52[g] [50[h]] | | 0[f] | |
| | | Site-specific P management | 0[h] | | 14[h] | | 0[f] | |
| | Source of Phosphorus | Liquid swine, dairy, and poultry manure compared to commercial fertilizer – Runoff shortly after application | -64 | 46 (45) | 90 | -33 | -1 (13) | 73 |
| | | Beef manure compared to commercial fertilizer – Runoff shortly after application | -133 | 46 (96) | 98 | | | |
| | Placement of Phosphorus | Broadcast incorporated within 1 week compared to no incorporation, same tillage | 4 | 36 (27) | 86 | | 0[f] | |
| | | With seed or knifed bands compared to surface application, no incorporation | -50 [-20[i]] | 24 (46) [35[i]] | 95 [70[i]] | | 0[f] | |
| | Cover Crops | Winter rye | -39 | 29 (37) | 68 | -28 | -6 (7) | 5 |
| | Tillage | Conservation till – chisel plowing compared to moldboard plowing | -47 | 33 (49) | 100 | -6 | 0 (6) | 16 |
| | | No till compared to chisel plowing | 27 | 90 (17) | 100 | -21 | -6 (8) | 11 |
| Land Use Change | Crop Choice | Extended rotation | | [j] | | -27 | 7 (7)[k] | 15 |
| | Perennial Vegetation | Energy crops | -13 | 34 (34) | 79 | | -100[l] | |
| | | Land retirement (CRP) | | 75 | | | -100[l] | |
| | | Grazed pastures | 2 | 59 (42) | 85 | | -100[l] | |
| Erosion Control & Edge-of-Field Practices | Terraces | | 51 | 77 (19) | 98 | | | |
| | Wetlands | Targeted water quality | | [m] | | | | |
| | Buffers | | -10 | 58 (32) | 98 | | | |
| | Control | Sedimentation basins or ponds | 75 | 85 | 95 | | | |

a - A positive number is P load reduction and a negative number is increased P load.

b - A positive corn yield change is increased yield and a negative number is decreased yield. Practices are not expected to affect soybean yield.

c - SD = standard deviation.

d - Maximum and average estimated by comparing application of 200 and 125 kg $P_2O_5$/ha, respectively, to 58 kg $P_2O_5$/ha (corn-soybean rotation requirements) (Mallarino et al., 2002).

e - This represents the worst case scenario as data are based on runoff events 24 hours after P application. Maximum and average were estimated as application of 200 and 125 kg $P_2O_5$/ha, respectively, compared to 58 kg $P_2O_5$/ha (corn-soybean rotation requirements), considering results of two Iowa P rate studies (Allen and Mallarino, 2008; Tabbara, 2003).
f - Indicates no impact on yield should be observed.
g - Maximum and average estimates based on reducing the average STP (Bray-1) of the two highest counties in Iowa and the statewide average STP (Mallarino et al., 2011a), respectively, to an optimum level of 20 ppm (Mallarino et al., 2002). Minimum value assumes soil is at the optimum level.
h - Estimates made from unpublished work by Mallarino (2011) in conjunction with the Iowa P Index and Mallarino and Prater (2007). These studies were conducted at several locations and over several years and may, or may not, represent conditions in all Iowa fields.
i - Numbers are from a report by (Dinnes, 2004) and are the author's professional judgment.
j – Water quality data for P loss on extended rotations in Iowa are scarce compared to data for a corn-soybean rotation.
k - This increase is only seen in the corn year of the rotation – one of five years.
l - The number is -100, indicating a complete cropping change and therefore a corn yield of zero.
m - P retention in wetlands is highly variable and dependent upon such factors as hydrologic loading and P mass input.

## Calculations for Practice Performance

The following methods were used to determine the minimum, mean, and maximum reduction of P and impacts on corn yield for each practice. Impacts were calculated using the same approach for most practices, but for some practices, the method was different and in these instances, differences are explained. See "Appendix – Summary of Literature Reviewed" for more details on specific studies used for each practice. Although this document focuses only on P reduction, some of these practices may provide other benefits, such as N loss reduction or aesthetic and wildlife benefits. The additional benefits were not included in the comparisons made here.

### Phosphorus Reduction Minimum and Maximum

Minimum and maximum values for the source, placement, tillage, cover crop, crop choice, perennial crops, pastures, wetlands, buffers, and erosion control practices were calculated based on individual site-years from each study. For example, if there were 10 years of data for a potential reduction practice and the highest resulting P load for one of the years was 5% HIGHER than the corresponding "normal" practice, the P removal of that practice in that year would be -5% (or a 5% P load increase). If the lowest load for one of the years was a P load of 25% LOWER than the corresponding comparison practice, the P removal of the potential reduction practice would be 25% (or 25% decrease in P load). The standard deviations for each practice were calculated using all site-year data.

### Phosphorus Reduction Mean

The mean P load reduction values were based on reported load observations for a given practice and compared to a corn-soybean base scenario. This approach was used, rather than averaging reduction values for each observation, as the range of load values was substantial between studies and a large reduction in a study with a small load may tend to produce an inflated reduction. Not all studies were conducted in the same manner and could include runoff studies with simulated rainfall on small field plots, field runoff studies with large plots and natural rainfall, or small catchment studies.

## Yield Calculations

The effect of P reduction practices on corn yields was calculated as above for the minimum and maximum values. A negative change is a reduced yield, and a positive change is increased yield. Mean yield change for a potential P reduction practice from the "normal" practice is calculated by averaging all observed yields for the P reduction practice that is being compared, subtracting average observed yield of the "normal" practice, then dividing by the average observed yield of the practice being compared.

## Calculations Differing from Above

Reductions for other potential practices required different approaches (see footnotes to Table 1). In some cases, little relevant data were available for certain practices in Iowa, which limits the confidence of practice performance. Three practices that could not be implemented in the above manner were P application rate, the impact of STP reduction, and site-specific P management. The effects of P application practices and site-specific management are difficult to summarize due to variations in many confounding factors such as background STP, soil type, extent of incorporation, and occurrence of runoff events after application.

<u>P application rate:</u> Two methods were used to estimate the P application rate effects in Table 1. The first method represents the long-term impact, assuming that precipitation does not occur within 1 week of P application, and includes results from Iowa P Index modeling (Mallarino et al., 2002) by comparing the P loss assuming the soil is at the optimum STP level. The maximum P reduction in Table 1 is based on a comparison of a rate of 200 kg $P_2O_5$/ha (178 lb $P_2O_5$/ac) with a 62 kg $P_2O_5$/ha (56 lb $P_2O_5$/ac) rate, which is the average annual removal for a corn-soybean rotation assuming corn yield at 11.3 Mg/ha (180 bu/ac), soybean yield at 3.7 Mg/ha (55 bu/ac), and prevailing grain P concentrations in Iowa (Sawyer et al., 2002). The average value is based on 125 kg $P_2O_5$/ha (112 lb $P_2O_5$/ac) applied compared to 62 kg $P_2O_5$/ha (56 lb $P_2O_5$/ac). The 200 kg $P_2O_5$/ha (178 lb $P_2O_5$/ac) and 125 kg $P_2O_5$/ha (112 lb $P_2O_5$/ac) starting points are arbitrary, but could represent resulting P application rates if, for example, poultry (egg layer) manure is applied based on N rates or at disposal rates. However, once incorporated into the soil, there is very little change in P loss directly associated with increasing P application rates. The second method used to assess the effects of P application rate is considered a "worst case scenario" in which rainfall occurs about 24 hours after P application. Data sets from two studies conducted in Iowa (Allen and Mallarino, 2008; Tabbara, 2003) were used for this method and background STP levels were at or below optimum, so no compounding factors would be involved in estimates. The relationship between P application rate and P loss under these conditions was derived from these data using the Iowa P Index. For consistency, the same hypothetical application rates as the first method were employed.

<u>Soil-test P reduction:</u> The effect of reducing the STP level on P loss reduction was determined by assuming a reduction of STP from a current high level to an optimum level for corn and soybean crops (20 ppm) by eliminating P application. It was assumed no P would be applied until enough P was removed via crop harvest to reduce STP to the optimum level, and that once at the optimum level, P would only be applied on a crop removal basis. The reduction columns in Table 1 were determined based on estimated P loss from using the Iowa P Index for a 5 Mg/ha erosion rate. The maximum column was estimated by comparing an average STP of the two highest counties in Iowa [125 ppm from Mallarino et al. (2011a)], which fall in MLRAs 104 and 108C from Figure 1, to the P loss for an optimum STP level. The average removal column was determined based on reducing the average STP of all counties in Iowa (assumed at 40 ppm) to the optimum level of 20 ppm. There are several counties with estimated STP levels below optimum, and even two of the eight MLRAs have average estimates lower than optimum, indicating the minimum reduction obtainable by this practice is zero. The relationship between P loss and STP is linear, thus this practice can also be represented in terms of P loss reduction per unit STP reduction. Using the 5 Mg/ha erosion rate above, this relationship is approximately 0.025 kg P/ha reduced for every ppm STP reduced.

Site-specific P management: The effect of site-specific P management on P loss was difficult to assess because of STP variation within a field, plus the levels at which this variation occurs differ greatly across fields. The smallest loss reduction estimate assumes zero reduction when STP is uniform within a field or where STP values did not exceed the optimum level (20 ppm). Utilizing unpublished mean values from a recent study of 14 fields (Mallarino, 2012), an estimate of the maximum long-term benefit of site-specific P management was made. The approach used to estimate P loss reduction was the same as for the STP practice [using Mallarino et al. (2002) relationships], but considered the mean proportion of Iowa STP interpretation classes (Sawyer et al., 2002) and the observed mean STP levels for the 14 fields as follows (15-cm depth, Bray-1 method): Very high, 51% of field and 52 ppm; High, 21% of field and 25 ppm; Optimum, 11% of field and 18 ppm; Low, 9% of field and 12 ppm; and Very Low, 8% of field and 6 ppm. The primary assumption with this practice was that no P would be applied to soils with high or very high STP levels until STP levels decreased to the optimum level. Additionally, it was assumed soils testing low or very low would receive ISU recommended rates of 65 kg $P_2O_5$/ha and 90 kg $P_2O_5$/ha, which was the average for crops of the corn-soybean rotation (Sawyer et al., 2002), respectively, until optimum STP levels are obtained. All other factors relevant to estimate P loss according to the Iowa P index were maintained constant for the scenario. These reduction estimates do not assume the fields included in the research accurately represent the soils, landscape, and STP distribution of all Iowa corn and soybean fields.

Based on Iowa data (Mallarino and Prater, 2007), an estimate for STP drawdown rate is about 1 ppm P/year (15-cm sampling depth, Bray-1 or Mehlich-3 methods) with a corn-soybean rotation with average study yields of 9.5 Mg/ha (151 bu/ac) and 3.3 Mg/ha (49 bu/ac) for corn and soybeans, respectively. Likewise, for increasing STP by 1 ppm/per year, a net application rate (after P removal from harvest) of approximately 17 kg $P_2O_5$/ha would be needed (Mallarino and Prater, 2007). These relationships are averages across several research sites, and there was variation (especially the increase in STP) depending on soil type, application rates, crop yields, and erosion rates. Using these relationships with the unpublished STP data from the 14 sites outlined above, it would take approximately 30 years to reduce a very high testing soil (50 ppm) to optimum soil test levels with an annual average P loss reduction of 0.44%. Total long-term P loss reduction for this example compared to original soil tests was 14%.

**Estimates of Potential Phosphorus Load Reduction with Phosphorus Management Practices**

As described earlier, alternatives for reducing P loading to receiving waters fall into three main groups: P management practices, edge of field and erosion control practices, and land use change. Phosphorus management practices focus on the most effective or efficient use of P, or those that otherwise reduce its availability for transport to receiving waters. Edge-of-field technologies are designed primarily to settle sediment, or, in some cases, to retain dissolved P. These provide opportunities to remove P either in combination with the above practices or as stand-alone P reduction strategies. A third option is changing land use, with major focus on cropping systems that involve perennial vegetation cover, row crops with cover crops, or rotations of row crops with perennial forage crops for hay, pasture, or bioenergy production. **In all practice options, the goal is to maintain P in soil and reduce its transport from fields to receiving waters, especially during times of the year with greatest chance of loss. No single practice will reduce P transport to receiving waters to stated goals by EPA, such as a 45% reduction in waters leaving Iowa to the Gulf of Mexico. It will take a combination of practices tailored to the characteristics of the specific landform.**

This section describes the potential for reducing P transport to Iowa surface waters using various standalone practices and a few combined practice scenarios. Included for each of the scenarios is a discussion of the practice limitations, economic considerations, other ecosystem services, and potential for P reduction. The practices are grouped into P management, edge-of-field, and land use change practices.

Baseline P loads were estimated for each Major Land Resource Areas (MLRA) using existing data on crop yield, land use, hydrologic characteristics, soil-test P (STP), P application rate, and tillage. These data were used to parameterize the Iowa P Index, which was adapted for use at the MLRA scale. The Iowa P Index was used to estimate the potential P load reduction for each standalone practice or combination of practices. **It is important to note the estimates for standalone practices seldom are additive — one cannot add together reductions from multiple practices.**

Economic costs for each practice include estimates for implementing the practice at the field level and any potential impact on crop yield, specifically corn grain yield. An equal annualized cost (EAC) was computed so those practices with annualized costs and those with large initial capital costs could be appropriately compared. For the capital costs, a design life of 50 years and a discount rate of 4% were used. The price of corn was assumed to be $5/bushel. The cost of nitrogen ($0.50/lb), phosphate ($0.59/lb), and potash ($0.47/lb) along with other costs such as seed, lime, herbicides, etc. were obtained from (Duffy, 2011a).

Practice/scenario costs for implementation and potential for P load reduction were calculated by MLRA, and then accumulated for a statewide cost and reduction estimate.

## Background on Phosphorus Load Estimation

### Agricultural Background Information for Iowa

The current land use, P management practices being used, and STP levels are required so any water quality benefits resulting from the P reduction strategies can be estimated. Iowa has 10 Major Land Resource Areas (MLRAs) (Figure 1) (Table 2). Each has different characteristics, such as soils, landscape, precipitation, and temperature. The state was divided using these areas to distinguish between agricultural practices that may differ in benefit across the state. For purposes of using the Iowa P index, MLRA 102C was combined with MLRA 107A, and MLRA 115C was combined with MLRA 108C. Management was assumed to be consistent throughout the combined areas.

As presented in the following discussion, a range of data was used to develop background information. Although years from which the data were drawn may not be the same, an effort was made to represent the state as accurately as possible, given the available data.

**Figure 1. The 10 MLRAs in Iowa. Descriptions in Table 2.**

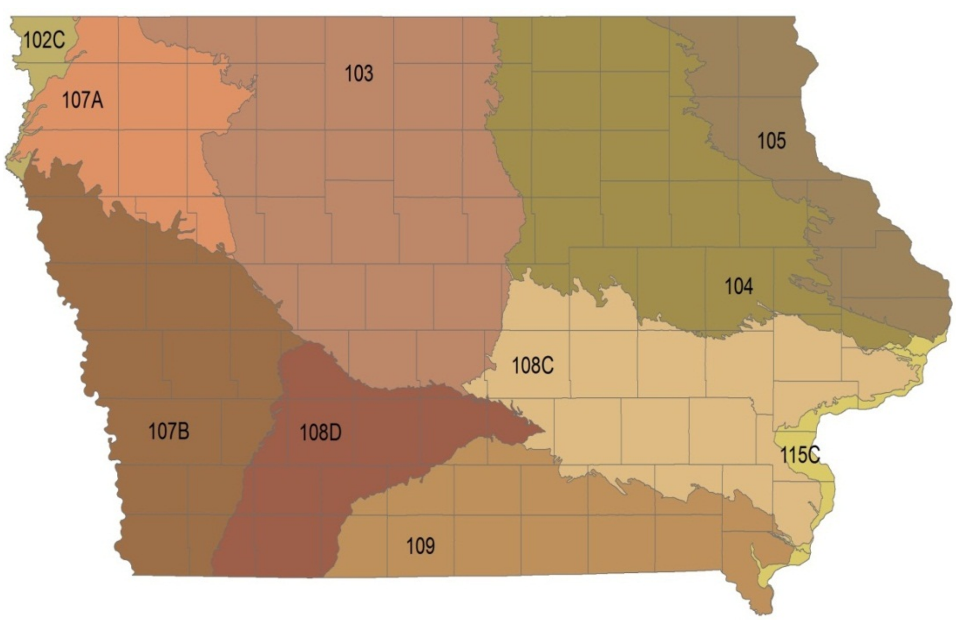


**Table 2. Description of the MLRAs in Iowa.**

| | | Landscape | | Climate | | |
|---|---|---|---|---|---|---|
| **MLRA** | **Description** | **Elevation m (ft)** | **Local Relief m (ft)** | **Total Precipitation mm (in)** | **Average Annual Temperature °C (°F)** | **Freeze Free days** |
| 102C | Loess Uplands | 335-610 (1,099-2,001) | 2-9 (7-30) | 585-760 (23-30) | 6-11 (43-52) | 170 |
| 103 | Central Iowa and Minnesota Till Prairies (aka. Des Moines Lobe) | 300-400 (984-1,312) | 3-6 (10-20) | 585-890 (23-35) | 6-10 (43-50) | 175 |
| 104 | Eastern Iowa and Minnesota Till Prairies | 300-400 (984-1,312) | 3-6 (10-20) | 735-940 (29-37) | 7-10 (45-50) | 180 |
| 105 | Northern Mississippi Valley Loess Hills | 200-400 (656-1,312) | 3-6 (10-20) | 760-965 (30-38) | 6-10 (43-50) | 175 |
| 107A | Iowa and Minnesota Loess Hills | 340-520 (1,115-1,706) | 3-30 (10-98) | 660-790 (26-31) | 7-9 (45-48) | 165 |
| 107B | Iowa and Missouri Deep Loess Hills | 185-475 (607-1,558) | 3-30 (10-98) | 660-1,040 (26-41) | 8-13 (46-55) | 190 |
| 108C | Illinois and Iowa Deep Loess and Drift – West-Central | 155-340 (509-1,115) | 3-6 (10-20) | 840-965 (33-38) | 8-11 (46-52) | 185 |
| 108D | Illinois and Iowa Deep Loess and Drift – Western | 210-460 (689-1,509) | 3-6 (10-20) | 840-940 (33-37) | 9-11 (48-52) | 185 |
| 109 | Iowa and Missouri Heavy Till Plain | 200-300 (656-984) | 3-6 (10-20) | 865-1,040 (34-41) | 9-12 (48-54) | 190 |
| 115C | Central Mississippi Valley Wooded Slopes - Northern | Similar to 108C | | | | |

***Crop Yield***

Total grain harvest (bushels) for both corn and soybean and total harvested land (acres) for both corn and soybean for each MLRA were determined by summing county estimates from the 2007 Agriculture Census (United States National Agricultural Statistics Service, 2009). Data from counties that are split between MLRAs were partitioned based on the percent of the county in each MLRA (Equation 1). For example, 96% of Audubon County is in MLRA 107B, while the other 4% is in MLRA 108D. Corn grain harvested in 2007 in Audubon County was 18,088,508 bushels (459,477,045 kg). Splitting the grain between MLRAs results in 17,364,968 bushels (441,097,963 kg) in MLRA 107B and 723,540 bushels (18,379,082 kg) in MLRA 108D.

**Equation 1**

$$Value_{MLRA} = \sum_{All\ Counties\ in\ MLRA} Value_{County} * \frac{\%County_{MLRA}}{100}$$

The number of harvested acres for each MLRA was also calculated this way. Once harvested grain and harvested area were summed for each MLRA, yield values were calculated (harvested grain/harvested area). Resulting yields are shown in Table 3.

**Table 3. Mean corn and soybean grain yields for each MLRA compiled from 2007 Agricultural Census. Two small MLRAs, 102C and 115C, have been incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | Corn Yield | | Soybean Yield | |
|---|---|---|---|---|
| | Mg/ha | bu/ac | Mg/ha | bu/ac |
| 103 | 10.7 | 170 | 3.4 | 50 |
| 104 | 10.7 | 171 | 3.4 | 51 |
| 105 | 10.6 | 170 | 3.4 | 50 |
| 107A | 9.9 | 158 | 3.4 | 51 |
| 107B | 9.6 | 153 | 3.3 | 49 |
| 108C | 10.8 | 173 | 3.4 | 51 |
| 108D | 9.4 | 150 | 3.3 | 49 |
| 109 | 9.6 | 153 | 3.1 | 47 |

Yields for corn in a continuous corn system were adjusted down while corn yields in a corn-soybean system were adjusted up to account for an approximate 8% yield reduction (Erickson, 2008) in a continuous corn system (Table 4).

**Table 4. Mean corn yields in corn-soybean and continuous corn systems for each MLRA compiled from the 2007 Agricultural Census with yield adjustments based on Erickson (2008). Two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | Corn Yield in Corn-Soybean | | Corn Yield in Continuous Corn | |
|---|---|---|---|---|
| | Mg/ha | bu/ac | Mg/ha | bu/ac |
| 103 | 11.0 | 175 | 10.1 | 161 |
| 104 | 11.0 | 176 | 10.2 | 162 |
| 105 | 11.2 | 179 | 10.4 | 165 |
| 107A | 10.1 | 161 | 9.3 | 148 |
| 107B | 9.8 | 156 | 9.0 | 143 |
| 108C | 11.1 | 177 | 10.2 | 163 |
| 108D | 9.5 | 151 | 8.7 | 139 |
| 109 | 9.7 | 155 | 9.0 | 143 |

***Crop Areas***

Crop areas were determined from NASS crop layer data for 2006 – 2010 using GIS methods. A summary can be found in Table 5 where CS represents a corn-soybean rotation, CC is continuous corn, EXT is an extended rotation, and PH is pasture or hay. A corn-soybean rotation is the dominant practice in Iowa, as well as in each MLRA, with the exception of 105, 108D, and 109, where PH is the dominant practice.

**Table 5. MLRA crop areas for corn-soybean rotation (CS), continuous corn (CC), various extended rotations (EXT), and pasture or hay (PH). The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | CS | CC | EXT | PH |
|---|---|---|---|---|
| | ha (ac) | ha (ac) | ha (ac) | ha (ac) |
| 103 | 1,917,134 (4,737,173) | 506,918 (1,252,577) | 77,125 (190,573) | 142,196 (351,362) |
| 104 | 1,293,724 (3,196,748) | 417,324 (1,031,193) | 111,299 (275,016) | 162,700 (402,026) |
| 105 | 154,347 (381,386) | 137,565 (339,918) | 81,381 (201,090) | 285,371 (705,142) |
| 107A | 810,924 (2,003,766) | 104,624 (258,522) | 45,886 (113,382) | 63,852 (157,776) |
| 107B | 1,189,034 (2,938,063) | 165,281 (408,404) | 113,560 (280,603) | 206,634 (510,586) |
| 108C | 916,735 (2,265,221) | 212,144 (524,201) | 133,846 (330,729) | 358,782 (886,538) |
| 108D | 388,642 (960,321) | 26,307 (65,004) | 80,779 (199,602) | 404,699 (999,998) |
| 109 | 235,615 (582,197) | 25,849 (63,872) | 81,675 (201,816) | 633,259 (1,564,762) |
| Iowa Total | 6,906,154 (17,064,873) | 1,596,013 (3,943,694) | 725,551 (1,792,812) | 2,257,495 (5,578,194) |

***Hydrologic Characteristics***

Tile drained areas were determined based on soil series identified as requiring drainage in the Iowa Drainage Guide and limited to slopes less than or equal to 2%. Drained land as a percentage of row crop area is shown in Table 6. Additionally, the tile drainage areas were used in conjunction with SSURGO drainage classes of Excessively Drained, Moderately Well Drained, Somewhat Excessively Drained, and Well Drained to determine the amount of “well drained” land as input into the Iowa P index. Tile drainage was used for MLRA 103, and Well Drained was used for all other MLRAs. Areas assumed to have tile drainage were classified as Drained Land.

**Table 6. Estimated land area with subsurface tile drainage (Drained Land) and soil area moderately well drained to excessively drained as defined by SSURGO soils data (Well Drained) as a percentage of row crop land for each MLRA in Iowa. The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | Drained Land (% Row crop) | Well Drained Land (% Row crop) |
|---|---|---|
| 103 | 67 | 33 |
| 104 | 32 | 49 |
| 105 | 17 | 89 |
| 107A | 37 | 63 |
| 107B | 25 | 80 |
| 108C | 44 | 59 |
| 108D | 36 | 62 |
| 109 | 70 | 19 |

Tile drainage, land slope, soil type, and land use affect the relationship between rainfall and runoff. Water yield (Table 7) from runoff and drainage used in this study was developed based on observed flow events in several watersheds and long-term precipitation.

**Table 7. Estimated mean water yield from the MLRAs in Iowa. The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | Water Yield | |
|---|---|---|
| | mm/yr | in/yr |
| 103 | 263 | 10.4 |
| 104 | 302 | 11.9 |
| 105 | 286 | 11.3 |
| 107A | 181 | 7.1 |
| 107B | 208 | 8.2 |
| 108C | 284 | 11.2 |
| 108D | 250 | 9.8 |
| 109 | 305 | 12.0 |

***Phosphorus Application***

Phosphorus application rates for each MLRA were estimated with Equation 2. Rates for fertilizer and manure at the county scale were taken from Jacobson et al. (2011). Since that study was designed to look at a total P balance for regions in the state, manure numbers included all cattle (both grain-fed and pastured). Since manure from pastured cattle is not applied to row crops, the manure from this cattle production system was not included in the analysis (leaving grain-fed cattle only). Replacement cattle numbers came from the 2002 Census of Agriculture (United States National Agricultural Statistics Service, 2007). The methods developed by Jacobson et al. (2011) used county-level data from both the 1997 and 2002 Census of Agriculture. Statewide fertilizer sales reported by the Association of American Plant Food Control Officials in 2008 were distributed among counties based on county-level fertilizer, lime, and soil conditioner expenditures for 1997 and 2002 as reported by the Census of Agriculture.

Phosphorus application rate to corn, soybean, and hay was determined by assuming producers apply only maintenance levels of P to replace what has been removed by the crop. This assumption was made in order to allocate applied P $Total\ County\ Phosphorus\ Application$ (*Total County P Application)* to the three primary crops. As P application and removal estimates did not agree for each county, the P removed by each crop ($Phosphorus_{Crop\ Removal}$) was divided by the total P removed across crops ($Phosphorus_{Total\ Removal}$) and this fraction was multiplied by the total county P application (Equation 2). This procedure allowed for consistent comparison of the relative proportion of P fertilizer applied to each crop. This calculation was used for each county before aggregating to the MLRA scale.

**Equation 2**

$$Phosphorus_{Crop\ Application} = \frac{Phosphorus_{Crop\ Removal}}{Phosphorus_{Total\ Removal}} * Total\ County\ P\ Application$$

The manure P values from Jacobson et al. (2011) were not adjusted to account for first-year crop availability because the upper bounds reported in Sawyer and Mallarino (2008) indicate it could be totally available in Iowa. In addition, application rate may be of less importance to P loss estimation than STP, as was discussed earlier.

The purpose of the above calculations was to more accurately determine the P application rate to all crops in each MLRA. Total P application rates were used in conjunction with current data on crop area (Table 5) to determine the total amount of P applied to each MLRA (Table 8). It was assumed the application rates have not changed significantly since the data were collected. No distinction was made between P applied as manure or commercial fertilizer when total application rates were calculated, as research has shown the amount of tillage, rather than P source, tends to be the primary driver of long-term P loss. However, as indicated in Table 1, when runoff occurs immediately following P application, there are substantial benefits of using manure instead of inorganic fertilizer to apply a specific P rate.

**Table 8. Total annual P application rates for each MLRA modified from Jacobson et al. (2011). This includes P from fertilizer and manure as applied to corn, soybean, and hay. The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| | Total $P_2O_5$ per Unit Area | | Total P Applied ($P_2O_5$) | |
|---|---|---|---|---|
| **MLRA** | **kg/ha** | **lb/ac** | **Mg** | **tons (2000 lbs)** |
| 103 | 54 | 48 | 141,980 | 156,504 |
| 104 | 52 | 47 | 103,986 | 114,623 |
| 105 | 63 | 56 | 41,175 | 45,387 |
| 107A | 76 | 68 | 77,521 | 85,451 |
| 107B | 45 | 40 | 74,651 | 82,287 |
| 108C | 54 | 48 | 87,389 | 96,328 |
| 108D | 40 | 36 | 35,833 | 39,498 |
| 109 | 47 | 42 | 46,174 | 50,897 |
| Iowa Total | 54 | 48 | 608,709 | 670,976 |

Table 9 provides the P application rates for corn, soybean, and hay. Average P removals for corn grain, soybean, and hay are 6.7, 13.3, 6.3 g $P_2O_5$/kg crop removed (Sawyer et al., 2002).

**Table 9. Calculated phosphorus application rates to corn, soybeans, and hay. The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| | Rate on Corn | | Rate on Soybean | | Rate on Hay | |
|---|---|---|---|---|---|---|
| **MLRA** | **kg $P_2O_5$/ha** | **lb $P_2O_5$/ac** | **kg $P_2O_5$/ha** | **lb $P_2O_5$/ac** | **kg $P_2O_5$/ha** | **lb $P_2O_5$/ac** |
| 103 | 66 | 59 | 40 | 35 | 38 | 34 |
| 104 | 63 | 56 | 39 | 35 | 45 | 40 |
| 105 | 71 | 64 | 47 | 42 | 57 | 51 |
| 107A | 89 | 81 | 58 | 53 | 60 | 55 |
| 107B | 54 | 48 | 35 | 31 | 35 | 32 |
| 108C | 65 | 58 | 42 | 38 | 44 | 39 |
| 108D | 49 | 44 | 32 | 29 | 31 | 28 |
| 109 | 60 | 54 | 40 | 36 | 36 | 32 |
| Iowa Total | 65 | 58 | 41 | 37 | 43 | 38 |

Mean STP estimates for each MRLA (Table 10) were calculated from Iowa county-based data from farmers' soil samples analyzed by the ISU Soil and Plant Analysis Laboratory from 2006 to 2010 (Mallarino et al., 2011a). Values for samples with calcareous soils (most in MRLA 103 and some in 107B) were adjusted based on Olsen P test results assuming Olsen extracts 60% P compared with Bray-1 (Mallarino, 1997).

**Table 10. Mean soil-test P for each MLRA in Iowa from Mallarino et al. (2011a). The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | Soil-Test P (ppm) |
|---|---|
| 103 | 30 |
| 104 | 27 |
| 105 | 27 |
| 107A | 32 |
| 107B | 28 |
| 108C | 27 |
| 108D | 19 |
| 109 | 11 |

The results for the different counties compared well with partial data shared by crop consultants. Although the MLRA averages are close to an optimum level of 16 to 20 ppm (Sawyer et al., 2002), some individual counties have excessively high STP values (131 ppm was the highest).

***Tillage practices***

Tillage estimates were compiled in 2008 by the Conservation Technology Information Center (CTIC). Categories included conventional tillage and conservation tillage, which was divided into no-till, mulch till, and ridge till for both corn and soybeans (Table 11). Ridge till was used in a small percentage of the crop area, and was lumped together with no-till.

**Table 11. Percent of no-till and mulch till for corn and soybean land for the MLRAs in Iowa. The two small MLRAs, 102C and 115C, were incorporated into MLRAs 107A and 108C, respectively.**

| MLRA | No-Till (%) | Mulch Till (%) |
|---|---|---|
| 103 | 8 | 44 |
| 104 | 20 | 38 |
| 105 | 24 | 30 |
| 107A | 11 | 45 |
| 107B | 44 | 24 |
| 108C | 35 | 33 |
| 108D | 42 | 29 |
| 109 | 33 | 24 |

***Data Compilation for use in the Iowa P Index***

The Iowa P Index is a quantitative risk assessment tool intended mainly to assess risk of P loss from individual agricultural fields, allow for comparisons of conservation and P management practices in relation to potential P loss, and estimate P delivered to nearest stream or water body. This model is comprehensive and estimates P loss, taking into account location in the state, soil type, STP, P application rate, tillage practices, source, timing and incorporation practices, runoff, erosion, and distance to the nearest stream or water body (Mallarino et al., 2002; NRCS, 2004). To satisfy the objectives of this effort, the science team adapted this tool to estimate P loads from MLRAs.

The process for collecting and analyzing MLRA-scale data for use in the Iowa P Index included several geospatial databases. Land use (row crop) data were extracted from the 2006 National Land Cover Database (NLCD) grid. Stream data are from the National Hydrography Dataset (NHD). Since the distance between the center of a crop field and the nearest stream or water body is an important parameter when estimating erosion and P loss with the P Index, information was gathered on row crop location in relation to the stream network, and seven distance classes were developed (0-500; 500-1,000; 1,000-2,000; 2,000-4,000; 4,000-8,000; 8,000-16,000; >16,000 feet). The distance classes were developed to approximate a relationship curve provided by Iowa P Index documentation (NRCS, 2004). All land was then placed into one of these categories determined by actual distance to a stream. Additionally, the distance of each class served as a boundary during the development of zones of analysis for soil parameters.

Another important parameter in the Iowa P Index is soil series, which can be determined from the Soil Survey Geographic (SSURGO) database. This database provides the erodibility factor, k, saturated hydraulic conductivity, Ksat, slope, and slope length parameters for each soil. Zonal statistics, or the statistics of soil parameters in each zone bound by distance class, were run on these data to determine the mean values for each distance class for each MLRA. The average slope and average slope length were determined for each distance class and then combined to obtain a slope length factor. Cover factors were determined based on land use (SCS-Iowa, 1990). After all data were gathered or estimated for each distance class, sheet and rill erosion rates were calculated using the Revised Universal Soil Loss Equation (RUSLE) and used as input in the Iowa P Index to estimate P loss. Row crop land was apportioned based on Tables 5 and 11 to determine amount of land in each crop and the proportion of tillage practices.

In addition to current cropping practices, information about P in the soil, based on the county-based STP summaries information, was evaluated by running zonal statistics to determine a mean value for each

MLRA. This was done with rainfall data as well, since annual precipitation is an important factor in erosion estimates.

The SSURGO database was cross-referenced with the NLCD database to determine the primary soils that are cropped. The resulting information was summarized by distance class for k, Ksat, and slope. Resulting estimates for soil parameters were compared to soils considered by the Iowa P Index within each distance class, and a representative soil was selected. Additionally, the resulting SSURGO analysis was used to determine the fraction of soils that were well-drained, as this affects P loss in the P-Index.

The current amount of land treated by terraces and contour farming was estimated based on best professional judgment of ISU Extension Agronomists for areas of the state where these practices would likely be prevalent. Specifically, contour farming was applied to 50% of the land in MLRA 105, and a combination of terraces and contour farming was applied to 50% of the land in MLRA 107b. To estimate the impact of contour farming, a RUSLE practice factor of 0.75 was used, and for a combination of terraces and contour farming, a practice factor of 0.5 was used. The P-Index model also incorporates contours and terraces in the runoff portion of the model, which was included where appropriate.

Finally, developed data were entered into the Iowa P Index along with P application rate (Table 9) for each distance class. The results were multiplied by the number of acres in each distance class in each MLRA to estimate a P load. Each practice or scenario was run by estimating the number of acres being implemented with the practice and developing the scenario within the P-Index.

## Phosphorus Management Practices

### Not Applying P on Acres with High or Very High Soil-Test P

This practice involves not applying P on fields where STP values exceed the upper boundary of the optimum level for corn and soybean in Iowa (20 ppm, Bray-1 or Mehlich-3 tests, 6-inch sampling depth). This practice would be employed until the STP level reaches the optimum level.

#### *Practice limitations, concerns, or considerations*

- No concerns when inorganic fertilizer is the P input for crops.
- Limitation to utilization of manure-N. When manure is applied, use of the P Index (which considers STP together with other source and transport factors) to assess potential impact of N-based manure on P loss is a reasonable option considering farm economics and other issues.
- Landlord/tenant contracts often require maintaining STP levels, even if higher than optimum.

#### *Costs/benefits*

The average estimated STP values from Mallarino et al. (2011) were used, along with the estimate of 1 ppm STP per year reduction in high or very high testing soils when growing a corn-soybean rotation without P application (Mallarino and Prater, 2007) for each MLRA to estimate the number of years required for not applying P. Cost savings were based on \$0.59/lb of phosphate ($P_2O_5$) and an application rate of 56 lb $P_2O_5$/ac (average annual need for a corn-soybean rotation with 180 bu/ac corn and 55 bu/ac soybean). This equates to \$36/ac/year savings in continuous corn and \$33/ac/year savings in a corn-soybean rotation. The acreage in continuous corn and corn-soybean rotation and number of years required to return county STP levels to optimum varied by MLRA. The annual EAC (benefit) of not applying P to high or very high STP soils is shown in table 12.

**Table 12. Cost for not applying P on soils testing high or very high. Costs amortized over 50 years.**

| MLRA | Average STP of each MLRA | Annual Cost of not Applying P to High or Very High STP Soils |
|---|---|---|
| | mg P/kg soil | $/ac |
| 103 | 30 | -12 |
| 104 | 27 | -9 |
| 105 | 27 | -9 |
| 107A | 32 | -14 |
| 107B | 28 | -10 |
| 108C | 27 | -9 |
| 108D | 19 | 0* |
| 109 | 11 | 0* |

* Average STP is below optimum and was not considered in this practice.

***Potential for load reduction (Scenario RR)***

Not applying P on those fields where STP values exceed the optimum level is estimated to reduce elemental P loading by 1,198 tons/year, which is approximately a 7% overall P load reduction at an annual farm-level cost of approximately -$263.5 million/year (net economic benefit) (Table 13).

**Table 13. Example Statewide Results for Individual Practices at Estimated Maximum Potential Acres, Phosphorus Reduction and Farm-Level Costs**

**Notes: Research indicates large variation in reductions. Some practices interact such that the reductions are not additive. Additional costs could be incurred for some of these scenarios due to industry costs or market impacts.**
**A positive $/lb P reduction, total cost or EAC is a cost. A negative $/lb P reduction, total cost or EAC is a benefit.**

| | Name | Practice/Scenario | P Reduction % (from baseline) | Potential Area Impacted for practice* (million ac) | Total Load (1,000 short ton) | Cost of P Reduction $/lb (from baseline) | Total EAC** (million $/year) | State Average EAC** ($/ac) |
|---|---|---|---|---|---|---|---|---|
| | BS | Baseline | | | 16.8 | | | |
| Phosphorus Management | CCa | Cover crops (rye) on all CS and CC acres | 50 | 21.0 | 8.3 | 60 | 1,022.9 | 49 |
| | Tnt | Convert all tillage to no-till | 39 | 16.1 | 10.3 | 14 | 186.4 | 12 |
| | Tct | Convert all intensive tillage to conservation tillage | 11 | 8.6 | 14.9 | -2 | -7.2 | -1 |
| | RR | P rate reduction in MLRAs that have high to very high soil test P | 7 | 25.8 | 15.6 | -110 | -263.5 | -11 |
| | CCnt | Cover crops (rye) on all no-till acres | 4 | 4.8 | 16.1 | 150 | 216.3 | 45 |
| | IN | Injection/band within no-till acres | 0.3 | 4.8 | 16.8 | 707 | 70.4 | 15 |
| Edge-of-Field**** | BF | Establish streamside buffers (35 ft) on all crop land*** | 18 | 0.4 | 13.7 | 14 | 88.0 | 231 |
| Land Use Changes | EC | Perennial crops (Energy crops) equal to pasture/hay acreage from 1987. Take acres proportionally from all row crop. This is in addition to current pasture. | 29 | 5.9 | 11.9 | 238 | 2,318 | 390 |
| | P/LR | Pasture and Land Retirement to equal acreage of Pasture/Hay and CRP from 1987 (in MLRAs where 1987 was higher than now). Take acres from row crops proportionally. | 9 | 1.9 | 15.3 | 120 | 365 | 192 |
| | EXT | Doubling the amount of extended rotation acreage (removing from CS and CC proportionally) | 3 | 1.8 | 16.3 | 53 | 54 | 30 |

* Acres impacted include soybean acres in corn-soybean rotation as the practice has a benefit to water quality from the rotation.
** EAC stands for Equal Annualized Cost (50 year life and 4% discount rate) and factors in the cost of any corn yield impact as well as the cost of physically implementing the practice. Average cost based on 21.009 million acres, costs will differ by region, farm and field.
*** Acres impacted for buffers are acres of buffers implemented and EAC are per acre of buffer.
**** This practice includes substantial initial investment costs.

**<u>Inject/Band P in All No-Till Acres</u>**

This practice involves injecting liquid P sources (fertilizer or manure) and banding solid inorganic fertilizers within all current no-till acres.

***Practice limitations, concerns, or considerations***

- For inorganic P fertilizers, it adds to the costs and does not increase (nor reduce) yield in Iowa.
- Possible benefits of injecting or banding inorganic P fertilizer containing N by improving N use efficiency.
- For liquid manure, this is a good practice to use manure-N efficiently.
- For solid manure, there is no practical way to do it yet, but engineering advances for prototypes being evaluated could make it practical in the future.

***Costs/benefits***

The cost of injecting or banding inorganic P fertilizer was estimated at $14.55 as per the 2012 Iowa Farm Custom Rate Survey (FM 1698, Iowa State University Extension). The cost of injecting liquid swine manure is estimated at $11.95 as per the 2012 Iowa Farm Custom Rate Survey. However, since no estimates of the proportion of inorganic P fertilizer versus liquid swine manure application are available, the more conservative estimate of $14.55 was used in estimating costs for this practice.

***Other services – ecosystem or environmental***

- More efficient use of liquid manure N.

***Potential for Phosphorus load reduction (Scenario IN)***

Injecting P within all current no-till acres in Iowa is estimated to reduce elemental P loading by 50 tons/year, which is less that 1% overall P load reduction at an annual farm-level cost of approximately $70,412,000/year (Table 13).

**<u>Convert All Intensive Tillage to Conservation Tillage</u>**

Tillage reduction will reduce P transport associated with soil erosion and surface runoff. This practice involves the conversion of all tillage acres to conservation tillage that covers 30 percent or more of the soil surface with crop residue, after planting, to reduce soil erosion by water.

***Practice limitations, concerns, or considerations***

- No clear data concerning impacts of this type of conservation tillage on possible corn yield reduction compared with moldboard plowing. However, data suggests the yield reduction is minimal in most conditions.
- These reduced tillage practices are significantly less efficient than no-till at controlling soil erosion and surface runoff.

***Costs/benefits***

To estimate the costs associated with conservation tillage systems, the publication *Estimated Costs of Crop Production in Iowa* (Duffy, 2012) was used to compare the difference between "conventional" or "intensive" tillage management practices ($\leq$20% residue after planting) and "conservation" tillage management practices (30% residue after planting). Table 14 illustrates the distribution of tillage in each MLRA and Table 15 highlights the EAC of this change in tillage.

**Table 14. Distribution of tillage in each MLRA. Base data from a Conservation Technology Information Center (CTIC) database.**

| | No-Till | Mulch Till | No-Till | Mulch Till |
|---|---|---|---|---|
| **MLRA** | **% of CC** | **% of CC** | **% of CS** | **% of CS** |
| 102C | 4 | 16 | 11 | 25 |
| 103 | 4 | 34 | 9 | 49 |
| 104 | 11 | 37 | 24 | 38 |
| 105 | 11 | 30 | 31 | 37 |
| 107A | 8 | 21 | 14 | 40 |
| 107B | 39 | 24 | 53 | 21 |
| 108C | 15 | 31 | 36 | 28 |
| 108D | 28 | 28 | 45 | 24 |
| 109 | 11 | 21 | 34 | 24 |
| 115C | 9 | 37 | 33 | 29 |

**Table 15. Average per acre EAC of converting from conventional tillage (<20% residue) to conservation tillage (30% residue) for continuous corn and corn-soybean by MLRA.**

| MLRA | Cost of converting from conventional tillage (<20% residue) to conservation tillage (30% residue) for CC and CS rotation - $/ac |
|---|---|
| 103 | -$0.95 |
| 104 | -$1.18 |
| 105 | -$2.66 |
| 107A | -$0.25 |
| 107B | -$0.38 |
| 108C | -$0.78 |
| 108D | $0.01 |
| 109 | -$0.23 |

### *Other services – ecosystem or environmental*

- Increases long-term soil productivity and crop yield.
- Reduces sediment loss, which extends the longevity of reservoirs.
- Reduces suspended and bedded sediments, thereby improving aquatic ecosystem integrity.

### *Potential for P load reduction (Scenario Tct)*

Conversion of all tillage to conservation tillage is estimated to reduce elemental P loading by 1,903 tons/year, which is about an 11% overall P load reduction at an annual farm-level cost of approximately -$7,209,000/year (net economic benefit) (Table 13).

**Convert All Tilled Area to No-Till**

Tillage reduction will reduce P transport associated with soil erosion and surface runoff. This practice involves the conversion of all tillage to no-till, whereby the soil is left undisturbed from harvest to planting except for strips up to 1/3 of the row width made with the planter (strips may involve only residue disturbance or may include soil disturbance). This practice assumes approximately 70 percent or more of the soil surface is covered with crop residue, after planting, to reduce soil erosion by water.

***Practice limitations, concerns, or considerations***

- No-till results in lower corn yield than with moldboard or chisel-plow tillage. However, the yield reduction is less or none for other minimum tillage options that, on the other hand, are less efficient at controlling soil erosion and surface runoff.
- No-till or conservation tillage does not affect soybean yield significantly.

***Costs/benefits***

The EAC of converting to no-till (70% residue) from either "conventional" (<20% residue) or "conservation" (30% residue) tillage systems were based on data from the publication *Estimated Costs of Crop Production in Iowa* (Duffy, 2012). Costs varied with average land rent in each MLRA. Also, since there is a 6% corn yield reduction when using no-till, there was a different cost for each MLRA associated variable MLRA yields. Tables 16 and 17 highlight the cost of converting to no-till.

**Table 16. Average per acre EAC of converting from conservation tillage (30% residue) to no-till (>70% residue) for continuous corn and corn-soybeans by MLRA.**

| MLRA | Cost of converting from conservation tillage (30% residue) to no-till (>70% residue) for CC and CS rotation - $/ac |
|---|---|
| 103 | $13.21 |
| 104 | $13.41 |
| 105 | $14.69 |
| 107A | $12.61 |
| 107B | $12.72 |
| 108C | $13.06 |
| 108D | $12.39 |
| 109 | $12.59 |

**Table 17. Average per acre EAC of converting from conventional tillage (≤20% residue) to no-till (≥70% residue) for continuous corn and corn-soybeans by MLRA.**

| MLRA | Cost of converting from conventional tillage (<20% residue) to no-till (>70% residue) for CC and CS rotation - $/ac |
|---|---|
| 103 | $10.32 |
| 104 | $10.64 |
| 105 | $12.76 |
| 107A | $9.32 |
| 107B | $9.51 |
| 108C | $10.08 |
| 108D | $8.96 |
| 109 | $9.29 |

For comparison, work done by the Center for Agricultural and Rural Development and Department of Economics at Iowa State University (Kling et al., 2007) reported an average 1997 to 2005 Environmental Quality Incentives Program (EQIP) payment of $14.88/ac and an Iowa Financial Incentive Program (IFIP) payment of $21.22 for conversion to no-till. Grain prices and land rent have both increased since the study period, which may partially explain the differences.

***Other services – ecosystem or environmental***

- Increases long-term soil productivity and crop yield.
- Reduces sediment loss, which extends the longevity of reservoirs.
- Reduces suspended and bedded sediments, thereby improving aquatic ecosystem integrity.

***Potential for P load reduction (Scenario Tnt)***

Conversion of all tillage to no-till is estimated to reduce elemental P loading by 6,544 tons/year, which is about a 39% overall P load reduction at an annual farm-level cost of approximately $186,390,000/year (Table 13).

**<u>Cover Crops</u>**

The cover crop in this practice/scenario is late summer or early fall seeded winter cereal rye. Winter rye offers benefits of easy establishment, seeding aerially or with drilling, growth in cool conditions, initial growth when planted in the fall, and continued growth in the spring.

***Practice limitations, concerns, or considerations***

- Impact on seed industry due to increased demand for rye seed.
- Row crops out of production to meet rye seed demand.
- New markets for cover crop seed production.
- Economic opportunities for seeding a cover crop.
- Livestock grazing.
- Corn and soybean planting equipment designed to manage cover crops in no-till.
- Negative impact on corn grain yield for species with spring growth.

***Costs/benefits***

The winter rye cover crop practice is an annual cost with little to no capital investment. Items included in the annual cost are seed and seeding, and cover crop termination (chemically killed and/or plowed down). Seeding at a rate of 60 lb/acre and a cost of $0.125/lb seed, the total seed cost would be $7.50/acre per year (Singer, 2011). There were several cost sources for seeding using a no-till drill, which range from $8.40/acre (Duffy, 2011) to $15/acre (Singer, 2011), with Edwards et al. (2011) estimating $13.55/acre.

To grow the primary crop, the cover crop must be terminated (chemically killed and/or plowed down). Glyphosate is the primary herbicide used for this procedure, and Singer (2011) suggested use at 24 oz product/acre with a cost of $0.083/oz, or $2.00/acre. Additionally, there is a cost associated with hiring spray equipment between $6 to $8/acre (Edwards et al., 2011).

The base cost of this practice (before any corn yield impact) ranges from $29/acre to $32.50/acre per year (value of $32.50/acre used for cost analysis). Any cost associated with a corn yield reduction due to the preceding rye cover crop depends on the baseline corn yields in each MLRA. The cost of implementing a rye cover crop, including corn yield impact, is shown in Table 18. From the review of literature, the estimated yield impact for corn following rye is -6%. No yield impact occurs with soybean following a preceding rye cover crop, therefore no soybean yield impact is included in the implementation cost.

**Table 18. Cost of using a rye cover crop. This cost is for operations, materials, and corn yield impact. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Cost of Implementing a Rye Cover Crop on Corn-Soybean Ground (EAC) | Cost of Implementing a Rye Cover Crop on Continuous Corn Ground (EAC) |
|---|---|---|
| | $/acre | $/acre |
| 102C | 40.5 | 83.5 |
| 103 | 42.5 | 86.5 |
| 104 | 42.5 | 87.5 |
| 105 | 42.5 | 86.5 |
| 107A | 40.5 | 83.5 |
| 107B | 39.5 | 81.5 |
| 108C | 43.5 | 87.5 |
| 108D | 39.5 | 80.5 |
| 109 | 40.5 | 81.5 |
| 115C | 43.5 | 88.5 |

***Other services – ecosystem or environmental***

- Wildlife habitat.
- Potential for P load reduction

*Scenario CCa: Plant a rye cover crop on all corn-soybean and continuous corn acres* - The same assumptions apply to this cover crop scenario as for the no-till only scenario. Any economic difference between the scenarios is due to increased acres, differences in corn yields, and corn acres in each MLRA. Incorporation of cover crops will force major changes in the agronomic practices where fall tillage is used. Implementing rye cover crops on all corn following soybean and continuous corn acres is estimated to reduce elemental P loading by 8,469 tons/year which is about a 50% overall P load reduction, with an annual farm-level cost of approximately $1,022,926,000/year (Table 13).

*Scenario CCnt: Plant a rye cover crop on all no-till acres* - The rationale for using this scenario is farmers currently using no-till are more likely to implement cover crops and the lack of fall tillage is conducive to timely establishment of fall-planted cover crops. As no-till corn is more common following soybean, continuous corn is considered separately. There is no assumption made about potential change in rye seed price or other establishment practices as rye cover crops are adopted. Implementing rye cover crops on the no-till acres is estimated to reduce elemental P loading by 720 tons/year, about a 4% overall P load reduction, with an annual farm-level cost of approximately $216,265,000/year (Table 13).

## Edge-of-Field Practices

### Buffers

#### *Practice limitations, concerns, or considerations*

Buffers have the potential to be implemented adjacent to streams to intercept overland flow and reduce P transport to receiving waters.

#### *Costs/benefits*

Costs of buffers can vary greatly depending on width, type of vegetation, and if substantial earthwork is required. For the analysis, cost of establishment and implementation was assumed to be $300/acre with an EAC of $13.96/acre/year. In addition, there would be a cost of land out of production which was assumed to be equal to the average cash rent for corn and soybean land for each MLRA (Edwards and Johanns, 2011a; Edwards and Johanns, 2011b). The EAC for buffer implementation by MLRA are shown in Table 19.

**Table 19. Cost of implementing buffers (cash rent for corn and soybean cropland, plus establishment EAC). (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Buffer Cost (EAC) - $/acre |
|---|---|
| 102C | 234 |
| 103 | 237 |
| 104 | 241 |
| 105 | 228 |
| 107A | 246 |
| 107B | 238 |
| 108C | 228 |
| 108D | 217 |
| 109 | 188 |
| 115C | 222 |

#### *Other services – ecosystem or environmental*

- Buffers would be expected to reduce nitrate-N load from shallow groundwater.
- Buffers would provide wildlife habitat benefits.
- Buffers would reduce greenhouse gas emissions.
- Buffer vegetation would sequester carbon.
- Buffers would stabilize stream banks and potentially reduce flood impacts.
- Buffers would improve aquatic ecosystem integrity.

***Potential for P load reduction***

*Scenario BF: Establishing 35 foot buffers on all crop land* - Establishing a 35-ft wide buffer on each side of agricultural streams that are not currently buffered would add buffers on 44,768 miles of agricultural streams for a total buffer area of 380,000 acres. Establishing buffers on all applicable cropland is estimated to have the potential to reduce elemental P loading by 3,090 tons/year, which is about an 18% overall P load reduction at an farm-level annual cost of approximately $88,044,000/year (Table 13).

## Land Use Change Practices

### Perennial Crops (Energy Crops) Replacing Row Crops

***Practice limitations, concerns, or considerations***

- Immediate limited market for perennials as energy crops.
- Market shifts in crop prices and demand.

***Costs/benefits***

Although there is not a current large market for perennial biomass crops as a source for energy or transportation fuel production, there are local and regional markets for those crops with current prices (example $50/ton). A publication from 2008 in the Ag Decision Maker series (Duffy, 2008) had estimates on the cost of production, transportation, and storage of switchgrass. At an assumed 4 ton/acre production level, the resulting revenue is $200/acre. The Ag Decision Maker costs factor in a land charge, and land rent for corn and soybean was used to represent the cost of switching from row crops to perennials. Since land rent is different in each MLRA, the resulting cost of producing energy crops varies by MLRA (Table 20).

**Table 20. Cost of producing a perennial energy crop, assuming 4 ton/acre production level and a sales price of $50/ton. (Note: A positive EAC is a cost. A negative EAC is a benefit. Included are cost of production, transportation, storage, land rent, estimated returns.)**

| MLRA | Cost of Producing Energy Crops (EAC) - $/acre |
|---|---|
| 102C | 399 |
| 103 | 402 |
| 104 | 405 |
| 105 | 392 |
| 107A | 411 |
| 107B | 402 |
| 108C | 392 |
| 108D | 382 |
| 109 | 353 |
| 115C | 387 |

***Other services – ecosystem or environmental***

- Increase wildlife habitat.
- Decrease erosion, surface runoff, and surface runoff transported pollutant export (e.g. P).
- Provide hydrologic services, that is, reduction of water runoff amount and rate.

*Potential for P load reduction (Scenario EC)*

This scenario switches corn and soybean row crop land to energy crops at the amount equivalent to reach the total number of acres in pasture/hay in 1987 for each MLRA (Table 21). Row crop acres were reduced proportionally for the corn-soybean rotation and continuous corn. This scenario is estimated to have the potential to reduce P loading by 4,900 tons/year, which is a 29% overall P load reduction at an annual cost of approximately $2,317,734,000 (Table 13).

**Table 21. Land area converted from corn and soybean to energy crops to reach the 1987 acres in pasture/hay for each MRLA.**

| MLRA | % of MLRA converted to energy crops | Acres converted to energy crops |
|---|---|---|
| 102C | 12 | 41,537 |
| 103 | 6 | 502,181 |
| 104 | 14 | 818,917 |
| 105 | 35 | 907,608 |
| 107A | 11 | 285,877 |
| 107B | 14 | 714,923 |
| 108C | 18 | 894,591 |
| 108D | 31 | 871,829 |
| 109 | 38 | 1,363,425 |
| 115C | 13 | 60,695 |

## Grazed Pasture and Land Retirement Replacing Row Crops

*Practice limitations, concerns, or considerations*

- Market and price shifts due to reduced row crop production.
- New markets for grass-fed beef.

*Costs/benefits*

The cost of switching land use from corn and soybean to pasture was calculated by subtracting the average cash rent received for pasture in each MLRA from the average cash rent for corn and soybean land (Edwards and Johanns, 2011a; Edwards and Johanns, 2011b). As there is limited data for both improved and unimproved pasture, the average cash rent of those two pasture categories was used for each MLRA. The resulting EACs for the practice implementation are shown in Table 22.

**Table 22. Cost of implementing pasture (cash rent for corn and soybean cropland, minus cash rent for pasture land). (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Pasture Cost (EAC) - $/acre |
|---|---|
| 102C | $150 |
| 103 | $169 |
| 104 | $171 |
| 105 | $159 |
| 107A | $173 |
| 107B | $159 |
| 108C | $159 |
| 108D | $148 |
| 109 | $122 |
| 115C | $145 |

Cost estimates for land retirement were based on income lost by taking land out of corn and soybean production (cash rent for corn and soybean) plus an annual maintenance cost. The maintenance was assumed to be mowing twice per year at a cost of $13.85/acre/mowing event ($27.70/acre/year) (Edwards et al., 2011). The EAC for each MLRA are shown in Table 23.

**Table 23. Cost of retiring corn and soybean row crop land. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Cost of Retiring Land (EAC) - $/acre |
|---|---|
| 102C | 248 |
| 103 | 251 |
| 104 | 254 |
| 105 | 242 |
| 107A | 260 |
| 107B | 251 |
| 108C | 241 |
| 108D | 231 |
| 109 | 202 |
| 115C | 236 |

***Other services – ecosystem or environmental***

- Increase wildlife habitat.
- Decrease soil erosion, surface runoff, and surface runoff transported pollutant export (e.g. P).
- Provide hydrologic services, that is, reduction of water runoff amount and rate.
- Increase carbon sequestration.
- Reduce greenhouse gas emissions.

***Potential for P load reduction (Scenario P/LR)***

This scenario increases the acreage of pasture and retired land to equal the pasture/hay and retired land acreage in 1987, which was the first time land was enrolled in the Conservation Reserve Program (CRP). Row crop acres were reduced proportionally for corn-soybean rotation and continuous corn. Some of the MLRAs have more land in pasture/hay and retired land now than in 1987, but the current amount was not adjusted down to the 1987 level. Research suggests that pasture/hay and land retirement reduces P loss by between 71% and 85% when compared to any land in corn or soybean. Statewide, this scenario impacts 1.9 million acres. Converting this amount of land from row crops to pasture and retired land (approximate 9% reduction in row crops) is estimated to have the potential to reduce P loading by 1,500 tons/year which is a 9% overall P load reduction at an annual cost of approximately $364,631,000 (Table 13).

**Extended Rotation (corn-soybean-alfalfa-alfalfa-alfalfa)**

For this analysis the extended rotation was assumed to be corn followed by soybean followed by three years of alfalfa.

***Practice limitations, concerns, or considerations***

- Reduce the amount of corn and soybean produced in Iowa.
- Market shift in product production (more alfalfa) and associated price for crops produced.
- Increased livestock production to feed alfalfa.
- Market shift as little fertilizer N is needed for corn following alfalfa.

***Costs/benefits***

As done with other practice costs related to perennial crops, the cost of the extended rotation is based on applicable cash rent values for each crop (Ag Decision Maker series, Duffy, 2008). The calculation used is shown in Equation 3.

**Equation 3**

$$\frac{3\ alfalfa\ years * \left(Cash\ Rent_{corn-soybean} - Cash\ Rent_{Alfalfa\ \ Hay}\right)}{5\ year\ total\ rotation}$$

This gives a range of $0/ac to $65/acre cost across the MLRAs and a state average of $35/acre before accounting for a corn yield improvement of 7% for the extended rotation. The resulting costs, after the corn yield improvement, are shown in Table 24.

**Table 24. The EAC cost of the extended rotation in each MLRA. (Note: A positive EAC is a cost. A negative EAC is a benefit.)**

| MLRA | Extended Rotation Cost (EAC) - $/acre | Extended Rotation Cost Including Increased Corn Yield (EAC) - $/acre |
|---|---|---|
| 102C | $0 | -$12 |
| 103 | $42 | $30 |
| 104 | $33 | $21 |
| 105 | $19 | $6 |
| 107A | $17 | $5 |
| 107B | $53 | $42 |
| 108C | $47 | $34 |
| 108D | $65 | $54 |
| 109 | $50 | $38 |
| 115C | $29 | $16 |

*Other services – ecosystem or environmental*

- Increased wildlife habitat.
- Decrease erosion, surface runoff, and surface runoff transported pollutant export.
- Provide hydrologic services, that is, reduction of water runoff amount and rate when land is in alfalfa.

***Potential for P load reduction (Scenario EXT)***

Increasing the acreage of extended rotations by doubling the current amount of extended rotations (and reducing proportionally the corn-soybean rotation and continuous corn) in each MLRA (Table 25) is estimated to have the potential to reduce P loading by 500 tons/year which is a 3% overall P load reduction at an annual cost of approximately $54,081,000 (Table 14).

**Table 25. Current extended rotation amount in each MLRA and the percent of land diverted from corn-soybean rotation and continuous corn for the scenario of doubling the amount of extended rotation (EXT).**

| MLRA | % of Row crop (current) | % of Row crop diverted to EXT from CS | % of Row crop diverted to EXT from CC |
|---|---|---|---|
| 102C | 8 | 6 | 2 |
| 103 | 3 | 2 | 1 |
| 104 | 6 | 5 | 1 |
| 105 | 22 | 12 | 10 |
| 107A | 4 | 4 | 0 |
| 107B | 8 | 7 | 1 |
| 108C | 11 | 9 | 2 |
| 108D | 16 | 15 | 1 |
| 109 | 24 | 21 | 2 |
| 115C | 10 | 8 | 3 |

## Combined Scenarios for Phosphorus Load Reduction

As is evident by results presented in Table 13, several individual practices do not achieve the needed P load reductions assuming a 45% reduction goal. As a result, a combination of practices may be needed. The combinations could be endless, but a few combined scenarios are highlighted below. Based on Iowa Department of Natural Resources estimates, nonpoint source P load reductions would need to achieve 29% of the overall target of 45%, with the remaining 16% P load reduction coming from point sources.

***Scenario PCS1***

This scenario assumes:

1. Phosphorus is not applied to all agricultural acres (CS, CC, EXT, and pasture) where STP values exceed the optimum level (20 ppm). This practice would be used until the STP level reaches the optimum level.
2. Conservation tillage is used on all CS and CC acres
3. Streamside buffers are established on CS and CC acres.

This scenario is estimated to have the potential to reduce elemental P loading by 5,066 tons/year which is approximately a 30% overall P load reduction at an annual farm-level cost of approximately -$182,669,000 (net economic benefit) (Table 26).

***Scenario PCS2***

This scenario assumes:

1. Phosphorus is not applied to 56% of agricultural acres (CS, CC, EXT, and pasture) where STP values exceed the optimum level (20 ppm). This practice would be used until the STP level reaches the optimum level.
2. No-till is used on 56% of tilled CS and CC acres.
3. Streamside buffers are established on 56% of CS and CC acres.

This scenario is estimated to have the potential to reduce elemental P loading by 4.878 tons/year which is approximately a 29% overall P load reduction at an annual farm-level cost of approximately -$42,994,000 (net economic benefit) (Table 26).

***Scenario PCS3***

This scenario assumes:

1. Phosphorus is not applied to 53% of agricultural acres (CS, CC, EXT, and pasture) where STP values exceed the optimum level (20 ppm). This practice would be used until the STP level reaches the optimum level.
2. No-till is used on 53% of tilled CS and CC acres.
3. Cover crops are used on all no-till CS and CC acres.

This scenario is estimated to have the potential to reduce elemental P loading by 4,945 tons/year which is approximately a 29% overall P load reduction at an annual farm-level cost of approximately $449,857,000 (Table 26).

***Scenario PCS4***

This scenario assumes:

1. Phosphorus is not applied to 63% of agricultural acres (CS, CC, EXT, and pasture) where STP values exceed the optimum level (20 ppm). This practice would be used until the STP level reaches the optimum level.
2. No-till is used on 63% of tilled CS and CC acres and cover crops established on no-till acres, except for MLRA 103 and 104.

This scenario is estimated to have the potential to reduce elemental P loading by 4,847 tons/year which is approximately a 29% overall P load reduction at an annual farm-level cost of approximately $189,533,000 (Table 26).

***Scenario PCS5***

This scenario assumes:

1. Phosphorus is not applied to 48% of agricultural acres (CS, CC, EXT, and pasture) where STP values exceed the optimum level (20 ppm). This practice would be used until the STP level reaches the optimum level.
2. No-till is used on 48% of tilled CS and CC acres and cover crops established on no-till acres.
3. Streamside buffers are established on 48% of CS and CC acres.

This scenario is estimated to have the potential to reduce elemental P loading by 4,869 tons/year, which is approximately a 29% overall P load reduction at an annual farm-level cost of approximately -$33,184,000 (net economic benefit (Table 26).

**Table 26. Example Statewide Combination Scenarios that Achieve Targeted P Reductions and Associated Nitrate-N Reductions**

**Notes: Estimated EAC based on 21.009 Million Acres of Corn-Corn and Corn-Soybean Rotation.**
**Research indicates large variation in reductions. Some practices interact such that the reductions are not additive.**
**Additional costs could be incurred for some of these scenarios due to industry costs or market impacts.**

| | | Phosphorus | Nitrate-N | Cost of P Reduction $/lb (from baseline) | Total EAC Cost* (million $/year) | Average EAC Costs ($/acre) |
|---|---|---|---|---|---|---|
| Name | Practice/Scenario** | % Reduction (from baseline) | | | | |
| BS | Baseline | | | | | |
| PCS1 | Phosphorus rate reduction on all ag acres (CS, CC, EXT, and pasture); Conservation tillage on all CS and CC acres; Buffers on all CS and CC acres | 30 | 7 | -18.03 | -182.7 | -$8 |
| PCS2 | Phosphorus rate reduction on 56% of all ag acres (CS, CC, EXT, and pasture); Convert 56% of tilled CS and CC acres to No-Till; Buffers on 56% CS and CC acres | 29 | 4 | -4.41 | -43.0 | -$2 |
| PCS3 | Phosphorus rate reduction on 53% of all ag acres (CS, CC, EXT, and pasture); Convert 53% of tilled CS and CC acres to No-Till; Cover crops on No-till CS and CC acres | 29 | 14 | 45.76 | 449.9 | $20 |
| PCS4 | Phosphorus rate reduction on 63% of ag acres (CS, CC, EXT, and pasture); Convert 63% of tilled CS & CC acres to No-till and cover crops on No-till crop acres except for MLRAs 103 and 104 | 29 | 9 | 19.55 | 189.5 | $8 |
| PCS5 | Phosphorus rate reduction on 48% of ag acres (CS, CC, EXT, and pasture); Convert 48% of tilled CS and CC acres to No-till with Cover Crop on No-till acres; Buffers on 48% CS and CC acres | 29 | 16 | -3.41 | -33.2 | -$1 |

***EAC stands for Equal Annualized Cost (50-year life and 4% discount rate) and factors in the cost of any corn yield impact as well as the cost of physically implementing the practice. Average cost based on 21.009 million acres, costs will differ by region, farm and field.**
****These practices include substantial initial investment costs.**

## Future Research Needs

A number of potential practices were discussed in this document that need further investigation concerning current use or adoption in Iowa and the impact on P loss reduction. Future Iowa research focused on nutrient reduction strategies for different practices should include:

**Assessment of current status**

- Better estimates of soil-test P levels around the state
- Better data on actual fertilizer and manure P application rates
- Current status of conservation practices, such as cover crops, terraces, contour farming, water and sediment control basins, ponds

**Phosphorus management**

- Impacts on water quality of variable-rate fertilizer and manure P application technology
- Development of commercially viable inorganic P fertilizer materials without N, so N and P management can be handled separately if needed
- Methods and management to reduce the N:P ratio of animal manures
- Field research based on large plots or catchments to study the impacts on P loss of alternative P management practices
- Validation of the Iowa P index as an edge-of-field and watershed scale assessment tool

**In-field and edge-of-field soil and water conservation practices**

- An efficient method to estimate ephemeral gully erosion and delivery of sediment
- Living mulch impacts on water quality
- Water quality data comparing extended rotations, pastures, and land retirement to a corn-soybean rotation
- Cover crop management techniques adapted to Iowa to limit the risk to corn yield reduction including development of new cover crop species and varieties
- Direct measurement of P loss from field edge and to surface water systems
- Sediment delivery ratio as influenced by the distance factor and role of road ditches and other channelized flow
- Development and evaluation of management practices to reduce stream bank erosion and sediment delivery
- Efficacy of alternative surface inlets

To quantify water quality improvements by implementing any new technology or ideas or determine the effectiveness of P reduction practices on a MLRA/statewide scale, it is important to have information about the starting point (i.e., background information about crop yields, land use, hydrologic characteristics, P application rates to crops). Although assumptions have been made in this effort to categorize background information, more accurate information about current agricultural practices would improve estimates.

## Appendix A – Literature Reviewed

Not all literature listed here was used in determining practice impacts on P loss reduction; however, all research work was reviewed for applicability to this P reduction strategy project. As part of this effort, data were added to a spreadsheet table for compilation and comparison. Comments in the following text similar to "data were added to the table" indicate that the water quality or agronomic data were compiled into the dedicated spreadsheet. Tables and figures displayed in the appendix are for informational purposes and have labels and numbers from the original publication source, which are not consistent with the numbering in the previous part of this document.

The following table (Sharpley et al., 2001) is presented for comparison to the practices in Table 1.

*Table 8.* Best Management Practices for control of nonpoint sources of agricultural P and N

| Practice | Description | Impact on loss[a] | |
|---|---|---|---|
| | | P | N |
| *Source Measures* | | | |
| Feed additives | Enzymes increase nutrient utilization by animals | Decrease | Decrease |
| Crop hybrids | Low phytic-acid corn reduces P in manure | Decrease | Neutral |
| Manure management | Compost, lagoons, pond storage; barnyard runoff control; transport excess out of watershed | Decrease | Decrease |
| Rate added | Match crop needs | Decrease | Decrease |
| Timing of application | Avoid autumn and winter application | Decrease | Decrease |
| Method of application | Incorporated, banded, or injected in soil | Decrease | Decrease |
| Crop rotation | Sequence different rooting depths | Neutral | Decrease |
| Manure amendment | Alum reduces $NH_3$ loss and P solubility | Decrease | Decrease |
| Soil amendment | Flyash, Fe oxides, gypsum reduce P solubility | Decrease | Neutral |
| Cover crops/residues | If harvested can reduce residual soil nutrients | Decrease TP | Increase DP |
| Plowing stratified soils | Redistribution of surface P through profile | Decrease | Neutral |
| *Transport Measures* | | | |
| Cultivation timing | Not having soil bare during winter | Decrease | Decrease |
| Conservation tillage | Reduced and no-till increases infiltration and reduces soil erosion | Decrease TP<br>Increase DP | Decrease<br>Increase $NO_3$ |
| Grazing management | Stream exclusion, avoid overstocking | Decrease | Decrease |
| Buffer, riparian, wetland areas, grassed waterways | Removes sediment-bound nutrients, enhances denitrification | Decrease TP<br>neutral DP | Decrease |
| Soil drainage | Tiles and ditches enhance water removal and reduce erosion | Decrease TP<br>Increase DP | Decrease TN<br>Increase $NO_3$ |
| Strip cropping, contour plowing, terraces | Reduces transport of sediment-bound nutrients | Decrease<br>Neutral DP | Decrease<br>Neutral $NO_3$ |
| Sediment delivery structures | Stream bank protection and stabilization, sedimentation pond | Decrease | Decrease |
| Critical source area treatment | Target sources of nutrients in a watershed for remediation | Decrease | Decrease |

[a] TN is total N, $NO_3$ is nitrate, TP is total P, and DP is dissolved P.

(Smart et al., 1985)
This was an extensive watershed study done in Missouri. And, although not directly applicable to Iowa, the trend in P concentration with different types of land use was interesting and is shown in the following table.

**Table 4. Mean concentration of water quality variables in streams draining single land use watersheds in the Missouri Ozarks during summer 1979.**

| Variable | | Land use: Urban $n = 45$ | Pasture $n = 45$ | Forest $n = 32$ |
|---|---|---|---|---|
| Total P† | mg/L | 0.106 | 0.046 | 0.020 |
| Total dissolved P | mg/L | 0.079 | 0.031 | 0.014 |
| Total N | mg/L | 11.5 | 3.37 | 0.92 |
| $NO_3^-$-N | mg/L | 2.14 | 1.32 | 0.02 |
| $NO_2^-$-N | mg/L | 0.10 | 0.01 | 0.01‡ |
| $NH_4^+$-N | mg/L | 0.10 | 0.02 | 0.02 |
| $Ca^{2+}$ | mmol/L | 1.73 | 1.55 | 1.02 |
| $Mg^{2+}$ | mmol/L | 0.28 | 0.13 | 0.94 |
| $Na^+$ | mmol/L | 0.74 | 0.20 | 0.02 |
| $K^+$ | mmol/L | 0.26 | 0.14 | 0.06 |
| Alkalinity | mg/L | 164.5 | 166.5 | 215.5 |
| $SO_4^{2-}$ | mmol/L | 0.39 | 0.02 | 0.02 |
| $Cl^-$ | mmol/L | 1.27 | 0.20 | 0.05 |
| $SiO_2$ | mg/L | 17.3 | 19.9 | 17.3 |
| Turbidity | JTU | 6.0 | 3.4 | 1.1 |
| Chlorophyll *a* | | | | |
| Benthic | mg/m² | 46.6 | 41.1 | 16.1 |
| Suspended | μg/L | 3.0 | 4.3 | 1.3 |

† Geometric means for all variables except arithmetic means for $Ca^{2+}$, alkalinity, and turbidity.
‡ Underlined means are not different at the 0.05 level of significance using a least significant difference test.

(Johnson et al., 1982; Koehler et al., 1982)
As referenced by (Ritter, 1988), these papers compare land uses in a number of states around the country (see below). Dataset was not used as no background information was provided. Note the data from Table 3 below was attributed to Johnson et al. (1982), but the citation should be Koehler et al. (1982). There was a large amount of variability, but forests tend have the lowest estimated P loads.

TABLE 2

Comparative Values of Some Nonpoint Sources[a]

| | Total N | | Total P | |
|---|---|---|---|---|
| | mg/L | kg/ha/yr[+] | mg/L | kg/ha/yr[+] |
| *Lower Limit for Algal Blooms | -- | -- | 0.025 | -- |
| *Maximum Level for Domestic Water Supply | 10 | -- | -- | -- |
| *Precipitation (U.S.) | 0.73 - 1.27 | 5.6 - 10.0 | 0.02 - 0.04 | 0.05 - 0.10 |
| *Precipitation (OH) | 2.0 - 2.8 | 12.8 | -- | -- |
| *Precipitation (Coastal DE) | -- | 44.6 - 45.4 | -- | 1.45 - 1.48 |
| *Precipitation (MN) | -- | -- | 0.011 - 0.042 | 0.10 |
| *Forested (OH) | 0.54 - 0.89 | 2.1 | 0.011 - 0.020 | 0.04 |
| *Forested (OH) | -- | 2.59 - 4.61 | -- | -- |
| *Forested (MN) | -- | -- | 0.04 - 1.20 | 0.08 |
| *Silvicultural Piedmont (VA)1 | 1.1 - 1.8 | 2.7 | 0.12 - 0.19 | 0.28 |
| *Agricultural Piedmont (VA) | 1.1 - 3.2 | 4.4 | 0.10 - 0.60 | 0.54 |
| *Agricultural Watersheds (Coastal DE) | -- | 14.4 - 15.7 | -- | 0.39 - 0.46 |
| *Farmland (OH) | 0.90 - 3.11 | 5.1 | 0.020 - 0.023 | 0.06 |
| *Upland Native Prairie (MN) | -- | 1.0 | -- | 0.13 |
| *Grassland - Rotational Grazing (OK) | 1.52 - 1.64 | 1.47 | 0.56 - 0.83 | 0.89 |
| *Grassland - Continuous Grazing (OK) | 2.58 - 3.25 | 6.84 | 1.29 - 1.32 | 3.24 |
| *Grassland - Rotational Grazing (TX) | 0.64 | -- | 0.04 | -- |
| *Grassland - Continuous Grazing (TX) | 0.94 | -- | 0.07 | -- |
| *Land Applied Dairy Manure (WI) | -- | 4.0 | -- | 0.08 |
| *Land Applied Dairy Manure (WI) | -- | 2.8 - 8.0 | -- | 0.4 - 1.7 |
| *Land Applied Dairy Manure (MN) | 13.2 - 62.3§ | -- | 1.8 - 4.9 | -- |
| *Land Applied Dairy Manure (SC) | 10.3 - 11.8§ | 11.8 - 16.6§ | 7.5 - 8.9 | 8.2 - 13.5 |
| *Land Applied Dairy Manure (AL) | -- | 0.8 - 3.2 | -- | -- |
| *Land Applied Dairy Manure (MN) | -- | 2.8 - 3.7 | -- | 0.5 - 0.6 |
| *Seepage from Stacked Manure (US) | 1,800 - 2,350 | -- | 190 - 280 | -- |
| *Seepage from Stacked Manure (WI) | 1,315 - 2,641 | -- | 51 - 156 | -- |
| *Feedlot Runoff (US) | 920 - 2,100 | 100 - 1,600 | 290 - 360 | 10 - 620 |
| *Feedlot Runoff (Great Plains Region) | 3,000 - 17,500 | -- | 47 - 300 | -- |
| *Dairy Barnyard Runoff (VT) | 78 - 3,953 | -- | 7 - 255 | -- |
| *Dairy Barnyard Runoff (NY) | -- | -- | 8.5 - 39.5 | -- |

[+]Normalized to precipitation of 76 cm/yr

*Surface Runoff

§$NO_3$-N

[a]Taken from Johnson et al. (12)

TABLE 3

Comparative Magnitude of Some Nonpoint Sources[a]

| | Total N | | Total P | |
|---|---|---|---|---|
| | mg/L | kg/ha/yr[+] | mg/L | kg/ha/yr[+] |
| *Precipitation (US) | .73 - 1.27 | 5.6 - 10 | -- | .05 - .10 |
| Lower Limit for Algal Blooms | -- | -- | .025 | -- |
| *Maximum Level - Domestic Water Supply | 10 | -- | -- | -- |
| *Precipitation (OH) | 2.0 - 2.8 | 12.8 | -- | -- |
| *Forest (OH) | .54 - .89 | 2.1 | .011 - .020 | .04 |
| *Farmland (OH) | .90 - 3.11 | 5.1 | .020 - .023 | .06 |
| *Precipitation (Coastal DE) | -- | 44.6 - 45.4 | -- | 1.45 - 1.48 |
| *Ag Watersheds (Coastal DE) | -- | 14.4 - 15.7 | -- | .39 - .46 |
| *Precipitation (MN) | -- | -- | .011 - .042 | .10 |
| *Forest (MN) | -- | -- | .04 - 1.2 | .08 |
| *Upland Native Prairie (MN) | -- | 1.0 | -- | -- |
| *Grassland - 112 kg N/ha (NC) | -- | 2.3 | -- | -- |
| *Grassland - 44 kg N/ha (NC) | -- | 8.4 | -- | -- |
| *Grassland - Rotate Graze (OK) | 1.52 - 1.64 | 1.5 | .56 - .83 | .89 |
| *Grassland - Continuous Graze (OK) | 2.58 - 3.25 | 6.8 | 1.29 - 1.32 | 3.24 |
| *Corn - 204 kg N/ha (Coastal GA) | .17 - .43§ | .1 - .2§ | -- | -- |
| **Corn - 204 kg N/ha (Coastal GA) | 7.07 - 10.31§ | 12.4 - 12.8§ | -- | -- |
| *Silvicultural Piedmont (VA) | 1.1 - 1.8 | 2.7 | .12 - .19 | .28 |
| *Agricultural Piedmont (VA) | 1.1 - 1.8 | 2.7 | .12 - .19 | .28 |
| *Poorly-Drained Coastal Plain (VA) | 1.7 - 2.3 | 1.6 | .19 - .31 | .21 |
| *Well-Drained Coastal Plain (VA) | 1.5 - 4.1 | 4.9 | .41 - .65 | .88 |

[+]Normalized to precipitation of 76 cm/yr

*Surface Runoff

§$NO_3$-N

**Subsurface Flow

[a]Taken from Johnson et al (15)

## Soil-Test Phosphorus

This may be one of the most important factors for P delivery when values are excessively high. A report by (Dinnes, 2004) indicates that applying P based on the STP level balanced with crop use could reduce P loss by 35% to 50% on an annual basis and by 40% over the long term. These reductions would likely only be realized, however, in areas with excessively high STP levels, and from Table 10, the estimated average STP level for the different MLRAs is not excessively high.

(Mallarino, 2011)
This presentation highlighted the relatively small contribution tile drainage makes on total P levels leaving a site. Concentrations in tile drainage do start to increase when STP levels increase to more than 80 ppm (Bray-1 or Mehlich-3 methods). Additionally, the author suggests the risk of P loss is minimal with low to optimal STP.

(Klatt et al., 2003)
This paper reviewed the relationship between STP and total P concentration in five watersheds. There were also two watersheds that had P loads measured. The monitoring timeline was between 1998 and 2000 (two water years included August 1998 to July 1999 and August 1999 to July 2000). The watersheds included in this study were mixed watersheds so the data cannot be directly used here, however, P load from August 1998 to July 1999 indicates the watershed with a higher percentage of perennial crops is lower while the August 1999 to July 2000 time period indicates the opposite. Two tables are shown here to compare the watersheds. The data were not added to the practice table.

**Table 5. Summary of selected management practices for the fields of the Clear Lake agricultural watershed.**

| | Area | | | | | |
|---|---|---|---|---|---|---|
| | | | Water monitored basins† | | | |
| Management practice | All basins | | 1 | 2 | 3 | 4 |
| | ha | % | | | | |
| **Primary tillage** | | | | | | |
| Chisel or disk | 591 | 58 | 41 | 63 | 84 | 11 |
| Moldboard plow | 248 | 24 | 18 | 10 | 16 | 78 |
| No-till | 25 | 2 | 0 | 27 | 0 | 0 |
| Ridge-till | 112 | 11 | 30 | 0 | 0 | 11 |
| Subsoiling‡ | 48 | 5 | 11 | 0 | 0 | 0 |
| **Cropping System** | | | | | | |
| Alfalfa, legume–grass pasture | 89 | 8 | 0 | 7 | 21 | 0 |
| Continuous corn | 54 | 5 | 0 | 0 | 0 | 0 |
| Corn–soybean rotation | 969 | 87 | 100 | 93 | 79 | 100 |
| **Time and method of P application** | | | | | | |
| Fall, injected and/or incorporated | 457 | 47 | 61 | 33 | 24 | 17 |
| Spring, injected and/or incorporated | 142 | 15 | 20 | 0 | 0 | 4 |
| Fall, surface applied | 241 | 25 | 9 | 22 | 76 | 0 |
| Spring, surface applied | 134 | 13 | 10 | 45 | 0 | 79 |
| **P source** | | | | | | |
| Fertilizer | 1002 | 90 | 79 | 100 | 100 | 100 |
| Manure | 110 | 10 | 21 | 0 | 0 | 0 |

† Very little and incomplete information was received for Basin 5 and data are not shown.
‡ In addition to no-till or chisel or disk tillage.

**Table 6. Precipitation, total P concentration, and total P loading from the Clear Lake agricultural watershed during a two-year evaluation period.**

| | | P concentration for five basins | | | | | P loads for two basins† | |
|---|---|---|---|---|---|---|---|---|
| Month | Precipitation | 1 | 2 | 3 | 4 | 5 | 2 | 3 |
| | mm | $\mu g\ L^{-1}$ | | | | | $g\ ha^{-1}$ | |
| 1998 | | | | | | | | |
| August | 140 | 372 | 606 | 314 | 332 | 338 | 478 | 348 |
| September | 46 | 691 | 242 | 330 | 209 | 671 | 62 | 161 |
| October | 88 | 185 | 127 | 100 | 150 | 124 | 63 | 60 |
| November | 28 | 150 | 135 | 90 | 130 | 150 | 21 | 17 |
| December | 10 | 237 | 179 | 213 | 205 | 190 | 17 | 19 |
| 1999 | | | | | | | | |
| January | 36 | 198 | –‡ | – | – | – | 64 | 60 |
| February | 41 | 817 | 550 | 502 | 441 | – | 209 | 248 |
| March | 32 | 595 | 311 | 343 | 337 | – | 93 | 81 |
| April | 208 | 186 | 201 | 176 | 158 | – | 231 | 158 |
| May | 192 | 388 | 259 | 396 | 365 | 273 | 244 | 334 |
| June | 131 | 411 | 499 | 219 | 351 | 245 | 362 | 373 |
| July | 283 | 319 | 894 | 343 | 388 | 398 | 418 | 198 |
| M/T§ | 1235 | 379 | 364 | 275 | 279 | 299 | 2262 | 2057 |
| August | 55 | 160 | 545 | 173 | 265 | 290 | 41 | 19 |
| September | 55 | 158 | 1088 | 230 | 196 | 245 | 18 | 6 |
| October | 28 | 145 | 138 | 144 | 219 | 866 | 1 | 6 |
| November | 19 | 780 | 393 | 377 | 481 | 423 | 2 | 3 |
| December | 19 | 256 | 148 | 59 | 85 | 466 | 26 | 41 |
| 2000 | | | | | | | | |
| January | 27 | 214 | 142 | – | – | – | 36 | 62 |
| February | 41 | 1158 | 980 | 1263 | 672 | 654 | 377 | 536 |
| March | 31 | 121 | – | 150 | 84 | 98 | 20 | 22 |
| April | 43 | 365 | 332 | 386 | 383 | 413 | 34 | 48 |
| May | 121 | 138 | 179 | 184 | 153 | 421 | 52 | 120 |
| June | 132 | 367 | 479 | 283 | 842 | 382 | 80 | 28 |
| July | 143 | 206 | 785 | 238 | 251 | 301 | 72 | 60 |
| M/T | 714 | 339 | 474 | 317 | 330 | 414 | 759 | 951 |
| Storm flow¶ | | 823 | 911 | 655 | 573 | 787 | | |

† Water discharge was continuously monitored only for Basins 2 and 3.
‡ No sample was collected. Loads for Basins 2 and 3 were calculated using the average P concentration of the previous and following month.
§ Means/total. Precipitation and loading are annual totals while P concentrations are monthly averages.
¶ Mean total water P concentration measured during or within 24 h of 15 storm flow events.

(Sharpley et al., 2001)

Although this study was not focused on Iowa, the authors show an interesting trend between STP and dissolved P in runoff and tile drainage. Having curves like this would be beneficial for Iowa.

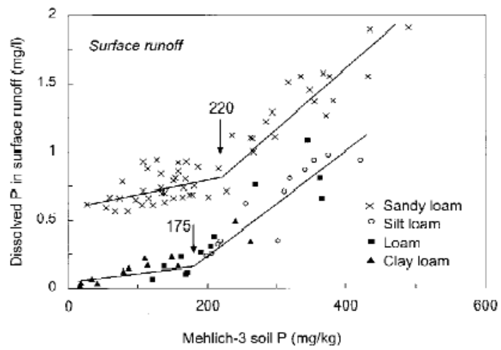



*Figure 3.* Relationship between the concentration of dissolved P in surface runoff and Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Co., PA (adapted form McDowell and Sharply, 2001).

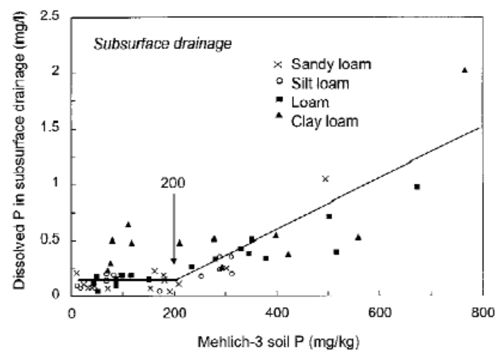



*Figure 4.* Relationship between the concentration of dissolved P in subsurface drainage from 30-cm deep lysimeters and the Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Cp., PA (adapted from McDowell and Sharply, 2001).

## Phosphorus Application Rate

There are a number of studies that have investigated P application rate. Results seem to indicate the placement (broadcast, injected, incorporated, etc.) along with time after application of first runoff event, and STP, are probably more important factors when considering P loss. Two studies (Allen and Mallarino, 2008; Tabbara, 2003) were used for the rate practice as these were done in Iowa and report background STP at or below optimum.

(Allen et al., 2006)
This paper reports findings on the relationship between P application rate and various forms of soil P. The goal was to compare soil P tests on different soils in and around Iowa. The relationships were developed with indoor rainfall simulation, and trends for all soils are the same — with increasing P application, the result is increasing levels of P in runoff. Although interesting and possibly useful in the future, these data were not added to the practice table.

(Allen and Mallarino, 2008)
This study looks at the relationship between P application rate, incorporation into the soil, and the number of days after application that rain occurs. The study was done on two Iowa soils, and relationships were developed to match observed data. This work will have a significant impact on estimating load from P applied systems and should make a good tool to compare against the P-Index. Main conclusions were that generally, after 15 days P loss from incorporated and unincorporated plots

with runoff is not much different (except one site in one year). Total P, bioavailable P and dissolved P all have similar trends. Of course, the higher the application rate the larger the impact of incorporation. Phosphorus application rates ranged from 0 to over 108 kg P/ha. Data were estimated from figures supplied in the publication for the 24-hour treatment, and, where appropriate, the 15-day treatment. Best fit lines were also supplied in the publication. This dataset was used along with the Tabbara (2003) study as an example of the impact of rate after different lengths of time between P application and P loss.

(Schuman et al., 1973)
This study is described under the “Grazed Pastures” section. Data were added to the practice table comparing the corn treatments with 39 kg P/ha to the corn treatments with 97 kg P/ha.

(Gessel et al., 2004)
This paper is described in the “Phosphorus Source” section as it was a manure-focused paper. The dataset was added to the practice table.

## Phosphorus Source

Similar to “Phosphorus Application Rate” it seems other factors such as STP and placement are likely more important than the source. Although not considered in this study, the addition of manure has been shown to enhance soil health and reduce the volume of runoff from a given site (Gilley and Risse, 2000), as well as possibly increase fauna (worm) activity (Converse et al., 1976).

Economically speaking, a paper by (Singer et al., 2010) suggests that using compost is more economically beneficial when compared to commercial fertilizer.

(Tabbara, 2003)
This study focused on comparing liquid swine manure to commercial fertilizer. Although the final P application rates were not the same (liquid swine high rate was 121 kg total P/ha compared to 158 kg total P/ha for fertilizer, and liquid swine low rate was 62 kg total P/ha compared to 74 kg total P/ha), the authors came to the conclusion a rainfall occurring 24 hours after application would cause more P to leave the commercial fertilizer treatments than the liquid swine manure treatments. This was attributed to the higher solubility of fertilizer P when compared to liquid swine manure. This paper also compared P incorporation strategies (broadcast with no incorporation vs. incorporated) and found incorporation was more effective at limiting P loss. Data have been assimilated into the practice table, and a linear interpolation was done between fertilizer and liquid swine manure numbers to directly compare application rate.

(Kovar et al., 2011)
This study was conducted in Iowa and included rainfall simulations in 2007 and 2008 on plots fertilized with liquid swine manure applied in two ways compared to commercial P. Additionally, the study investigated the impact of cover crops on runoff and P load. These data were not used here due to variability in rainfall applied to the plots in the study, which did not allow for a direct comparison between practices. Additionally, the rainfall events did not occur the same number of days after manure application, which may have influenced how much P was lost. The authors do suggest, however, that the addition of a cover crop may not increase the dissolved reactive P lost.

(Barbazan et al., 2009)
This study focused on yield differences when using liquid swine manure and commercial fertilizer. The authors conclude there are no differences between P availability between the two sources. Additionally, adding more fertilizer did NOT further increase yields.

(Lawlor et al., 2011)
This paper from Gilmore City, Iowa, highlights the differences in adding commercial fertilizer with adding liquid swine manure. All yield data has been added to the table as site years, although a linear interpolation was done to make direct nitrogen application rate comparisons as N application rates were sometimes substantially different and P was generally not limiting.

(Bakhsh et al., 2005)
This paper was summarized in the “Phosphorus Application Rate” section as there were no directly comparable rates of liquid swine manure and commercial fertilizer. Yields have been added to the practice table.

(Rakshit, 2002)
This thesis had two years of data from multiple farms with multiple liquid swine application rates. Although there were no direct comparisons to commercial fertilizer in the study, the multiple rates allowed for linear interpolation between nitrogen rates for yield comparison as P was generally not limiting. All data were added to the practice table, but there tended to be a slight yield decrease when comparing.

(Chinkuyu et al., 2002)
This research conducted at Ames, Iowa, was a 3-year study (1998 to 2000) looking at the application of laying hen manure. The treatments are spring-applied UAN at 168 kg N/ha, spring-applied laying hen manure at 168 kg N/ha (actual total N application rates of 115, 219, and 117 kg N/ha for 1998 to 2000), and spring-applied laying hen manure at 336 kg N/ha (actual application rates of 254, 324, and 324 kg N/ha for 1998 to 2000). There was also an associated lysimeter study with the same treatments. The 168 kg N/ha manure treatment had actual rates of 167, 169, and 162 kg N/ha, while the 336 kg N/ha manure treatment had 337, 338, and 325 kg N/ha applied. Although this was a N treatment study, it was assumed that P was not a limiting factor, and yield results were added to the practice table as a manure vs. commercial fertilizer comparison.

(Ruiz Diaz and Sawyer, 2008; Ruiz Diaz et al., 2011)
These papers were used for yield numbers from poultry manure applications. Results show little yield impact (positive or negative) of using manure. Data were added to the practice table.

(Ginting et al., 1998b)
This paper is described in the “Tillage and Residue Management” section.

(Eghball et al., 2000)
See description under the “Tillage and Residue Management” section.

(Andraski et al., 2003)
See description under the “Tillage and Residue Management” section. Data were added to the practice table.

(Allen and Mallarino, 2008)
See description under the “Phosphorus Application Rate” section.

(Bundy et al., 2001)
This study is described in the “Placement of Phosphorus” section. Data has been added to the practice table.

(Zhao et al., 2001)
This small plot study using rainfall simulation in southern Minnesota in 1997 compared two types of tillage (moldboard and ridge till) and two sources of P (beef manure and urea). Results showed in the

moldboard system the manure treatment had lower P loss than urea, but in the ridge till system the manure treatment has substantially more P loss than urea. Also, overall, the ridge till system had lower P loss from surface runoff than the moldboard system. Interestingly, tile drainage from the ridge till system is higher than the moldboard system. Data were added to the practice table for tillage and source.

(Gessel et al., 2004)
This study was conducted in Morris, Minn., between 1998 and 2001 and compared water quality results (runoff) and yield results from plots with different rates of manure application. There were no significant differences in total P loss with any of the treatments; however, the treatment with no manure (no P) and the treatment with the highest manure (and P) rate had the lowest total P loss (2.3 kg P/ha and 2.2 kg P/ha, respectively). The two mid-level manure treatments were approximately 2.5 kg P/ha. The only statistically significant difference in yields was for soybeans, where the no application and low application rates produced lower yields (2.2 compared to 2.5 Mg/ha). Although a manure study, there was not a comparable fertilizer treatment so the dataset was estimated from a figure and added to the practice table under the "Phosphorus Application Rate" section.

(Mallarino et al., 2010a)
This study was done in O'Brien County, Iowa, and compared no-till and chisel plow systems with and without manure (liquid swine). The dataset reported is for 2008, 2009, and half of 2010 and includes P loss and crop yields. The general trend was the chisel plow plots lost more P than the no-till plots and the fertilized plots lost more P than the manure plots. Although not specifically stated, the assumption is made here that fertilizer P and manure P application rates were the same. The dataset was added to the practice table under tillage, source, and placement.

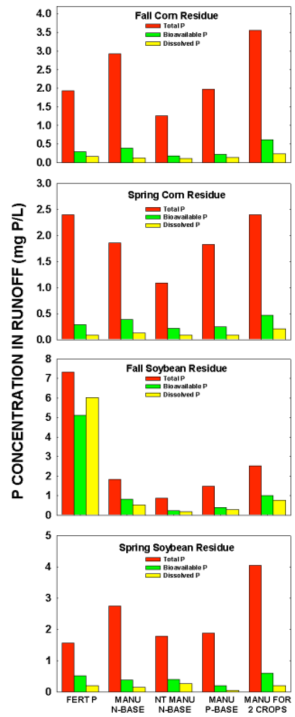
(Mallarino et al., 2010b)
This paper summarizes the same project as described in (Mallarino et al., 2010a).

(Mallarino et al., 2011b)
This is an update to (Mallarino et al., 2010a) and data has been added to the practice table.

(Mallarino and Haq, 2012)
This report to the Iowa Egg Council looked at P concentrations in rainfall simulated runoff using inorganic fertilizer and poultry manure with or without treatment. The study only reported concentrations; however, the study shows a reduction in P concentrations when using additives such as alum or gypsum with manure application. The study also found higher P concentrations in fertilized plots when compared to manured plots. As P loads were not reported, the dataset was not added to the practice table.

(Mallarino et al., 2005)
This report presented findings from a rainfall simulation runoff study looking at P runoff concentrations at the Northeast Research and Demonstration Farm in Iowa. Although not reported, the authors suggest P load trends were similar to concentrations, which indicate no-till treatments receiving manure at a rate governed by nitrogen demand generally had the lowest total P concentrations, while P applied to chisel plowed systems based on P needs tended to have the next lowest concentrations. Highest concentrations were seen when applying manure for 2 crops in a chisel plowed system except in the fall soybean residue, where fertilizer P resulted in the highest concentrations. As this dataset did not report loads, it was not added to the practice table; however, the following figure outlines the findings.

Fig. 2. Effect of management systems on runoff P concentrations




(Mallarino and Haq, 2007)
This rainfall simulation study investigated relationships between STP and runoff P loss from 2004 until 2006 in many farmers' fields. During 2005 and 2006, work at 21 fields evaluated P loss when 100 lb $P_2O_5$/acre were applied without incorporation into the soil using inorganic fertilizer, liquid swine manure, solid beef feedlot manure, and poultry manure. Simulated rainfall was applied within 24 hours of the P application. Results showed good correlations between STP and total or dissolved P loss only when fertilizer was not applied between the soil sampling date and the runoff events. The total and dissolved P losses always were highest for fertilizer, intermediate for liquid swine manure, and lowest for poultry and beef manures. Differences between poultry and beef manures were small, inconsistent, and varied among fields and seasons, but on average runoff P tended to be slightly higher for poultry manure.

(Mallarino and Haq, 2008)
This rainfall simulation study in 2006 and 2007 investigated the differences between poultry manure and commercial fertilizer in regards to P loss in runoff. A large number of poultry manure types were used at multiple locations (17 total fields). Phosphorus application rate was 100 lb total $P_2O_5$/ac for all sources. Slopes for all sites ranged between 2.5 and 7% and all trials were run on soybean residue with no tillage or incorporation. Rainfall simulation was done within 24 hours of P application and was run long enough to get 30 minutes of continuous runoff. The general trend was that poultry manure, no matter the type, had similar P loss in runoff, which was lower than the loss from fertilizer. This dataset (as estimated from reported figures) was added to the practice table in three sets (fall 2006, spring 2007, and fall 2007), as this is how it was reported.

(Daverede et al., 2004)
This study is described in the “Placement of Phosphorus” section. Data has been added to the practice table.

(Wortmann and Walters, 2006)
This research was conducted in Nebraska to evaluate soil P test prediction of P concentration in runoff and to determine the residual effects of composted manure on runoff P loss and leaching of P. The research was conducted from 2001 to 2004 under natural runoff events with plots of 11-m length. Runoff and sediment losses were 69 and 120% greater with no compost than with residual compost treatments. Runoff P concentration increased as STP increased, but much P loss occurred with the no-compost treatment as well. Authors concluded that the residual effect of compost application in reducing sediment and runoff loss was evident more than 3 yr after application and should be considered in P indices.

(Wortmann and Walters, 2007)
Research was conducted in 2004 and 2005 under natural rainfall to determine the residual effects of previously applied compost, plowing of soil with excessive STP, and application of additional compost after plowing on volume of runoff and loss of sediment and P in runoff. Inversion plowing greatly decreased P levels in the surface soil and over the following year reduced runoff by 35% and total P loss by 51% compared with the unplowed compost treatments. Sediment loss was increased with plowing compared with the unplowed compost applied treatments but less than with the no-compost treatment. Unplowed compost-amended soil continued to reduce sediment loss but exhibited increased DRP loss even 5 yr after the last application. Plowing to invert excessively high-P surface soil was effective in reducing runoff and DRP loss.

## Placement of Phosphorus

Phosphorus not incorporated into the soil can be readily lost. (Dinnes, 2004) suggests deep tillage incorporation compared to surface broadcast could show a -75 to 50% reduction on an annual basis and a long term average of -15% reduction; shallow tillage incorporation compared to surface broadcast could show a -75 to 40% reduction on an annual basis and a long term average of -10% reduction; and knifing or injecting compared to surface broadcast could show a -20 to 70% reduction on an annual basis with a long term average of 35% reduction. Reasons behind this logic are that the possibility of a runoff-producing storm is the same with no incorporation or incorporation, and if a runoff producing storm occurs when the soil is disturbed, more sediment may leave the site.

(Tabbara, 2003)
See study description under “Phosphorus Source”, which describes the incorporation techniques investigated. Data from this paper was reformatted and added to the practice table.

(Sharpley et al., 2001)
Not done in Iowa, however, the trend shown for application method/incorporation is telling and is likely the same trend that would be observed in any soil.

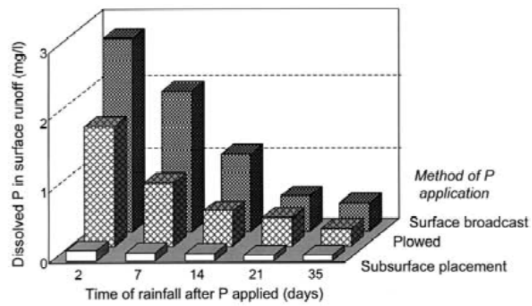


*Figure 6.* The effect of application method and timing of rainfall after application of dairy manure (100 kg p?ha) on the concentration of P in surface runoff from a grassed Berks silt loam.

(Allen and Mallarino, 2008)
See study description under “Phosphorus Application Rate.”

(Timmons et al., 1973)
This study was done in west-central Minnesota with rainfall simulation in 1968 and 1969 with a P application rate of 168 kg P/ha (150 lb P/ac). The authors found no significant differences between unfertilized plots and those where the P was incorporated by plowing and disking. Unincorporated plots had the highest P loss. This data has been added to the practice table.

(Andraski et al., 2003)
This study was described in the “Tillage and Residue Management” section. Data were added to the practice table in this section to account for the no-till and chisel plow incorporation methods.

(Bundy et al., 2001)
This rainfall simulation study done in Arlington and Madison, Wis., compares a number of parameters; however, for this study the data for tillage and source were used. Additionally, the tillage data (chisel plow compared to no-till) was used to compare incorporation vs. no incorporation. The general trends were that manure treatments tended to have a lower P load than inorganic fertilizer, and P loss decreases with increased surface residue. Data has been added to the practice table.

(Baker and Laflen, 1982)
This rainfall simulation study was conducted in Iowa and compared incorporated and unincorporated fertilizer application as well as multiple levels of residue cover. This study only reported dissolved nutrients; however, the trends were strong. As expected, erosion reduced with increasing residue. Unexpectedly, orthophosphate loads were fairly consistent for all residue amounts at ~0.13 kg PO4-P/ha. The one exception was the 1500 kg/ha treatment, which had the most residue and the lowest PO4-P load at 0.05 kg PO4-P/ha. Additionally, there was very little difference in the placement of the fertilizer. Data were not added to the table since the study did not report total P.

(Kovar et al., 2011)
This study is described in the “Cover Crops” section. The data were added to the practice table.

(Mallarino et al., 2010a)
This study was described in the “Phosphorus Source” section. Data were added to the practice table.

(Mallarino et al., 2011b)
This study was described in the “Phosphorus Source” section. Data has been added to the practice table.

(Daverede et al., 2004)
This study, done in northwest Illinois between 1999 and 2001, compares phosphorus loss with different sources and different application types or placement techniques on soybeans. Results show that when P is surface applied, the risk for P loss is high when runoff occurs after the first month but reduces significantly after 6 months. There were no significant differences between source when the P was incorporated or injected and a runoff event occurred one month after application. Six months after application there were no significant differences between any of the treatments. The dataset was added to the practice table for source and placement.

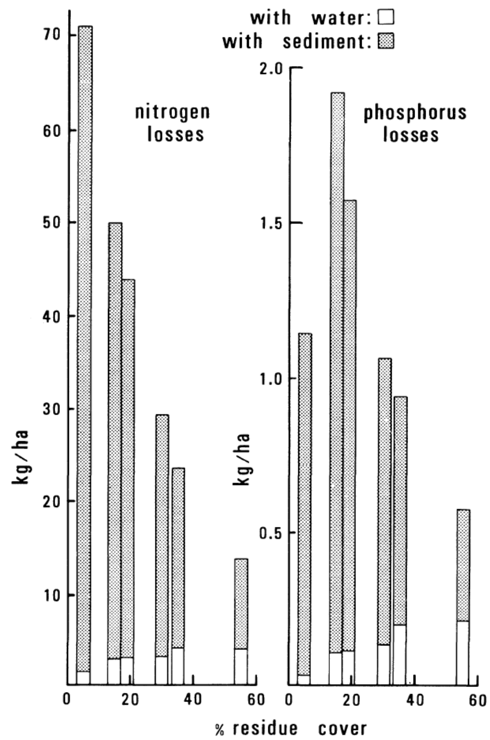
## Tillage and Residue Management

Overall, reduced tillage tends to decrease P loss due to limiting soil erosion. There are also additional benefits in increasing soil organic matter near the surface (Dick et al., 1991; Lal et al., 1990); however, these will not be covered by this project. The comparison between surface runoff volume between tillage practices is not directly covered here; however, the P load from each tillage practice factors in runoff. It should be noted that no-till systems tend to have slightly greater runoff volume than chisel plowing (Ritter, 1988).

Sediment is not directly used with this effort; however, it is recognized that the majority of P moves with sediment and as such, soil erosion is an important process. A paper by (Laflen and Colvin, 1981) shows a very strong relationship between soil erosion and residue cover on several soils in Iowa. The trend is of decreasing erosion with increasing residue cover.

A paper by (Singer et al., 2010) suggests moldboard plowing is the most economical tillage type, when not using compost; however, when using compost, both chisel plowing and no-till is more profitable.

(Barisas et al., 1978)
This was a small plot study with rainfall simulation (1.4 hour storm in the afternoon at 6.35 cm/hr followed by a 1 hour storm the next morning at 6.35 cm/hr followed by a 0.5 hour storm at 12.7 cm/hr) investigating several types of tillage (conventional, till-plant, chisel plow, disk, ridge-plant, and fluted coulter). The three soil types included in this study were Kenyon, Tama, and Ida with slopes of 4.8, 4.7, and 12.2, respectively. P fertilizer was added at 67 kg/ha as $P_2O_5$ (29 kg P/ha). Soluble P ($PO_4$-P) concentrations were measured in runoff water. These concentrations were generally lower with less residue and had the trend: conventional < till < disk < chisel < coulter < ridge for the Ida soil, conventional < till < chisel < ridge < disk < coulter for the Kenyon soil, and conventional < till < chisel < disk < coulter for the Tama soil. Bottom line trend is that as residue increases, P loss with water increases, but P loss with sediment decreases. The net P loss decreases with increasing residue cover (illustrated in the following figure). Data were estimated from the figures provided and added to the practice table. Tillage practices are described in (Laflen et al., 1978).




(McIsaac et al., 1993)
This study was done in Illinois comparing no-till, ridge-till, and moldboard plow on a Catlin silt loam soil (1.5 to 4% slope) and no-till, ridge-till, chisel plow, and moldboard plow on a Tama silt loam soil (6-13% slope). Loads were calculated for those treatments running up and down the slope. The rainfall simulation used was at 64 mm/hr for one hour. Basic findings were that increased tillage works to reduce dissolved P loss. Although this work was done in Illinois, the data were added to the practice table for comparison as the trend is similar to what other researchers have found.

(Kanwar and Baker, 1993)
This study focused on nitrate; however, yield data associated with tillage type was also included. The study found approximately a 7% yield decrease in the no-till treatment when comparing to moldboard plowing.

(Andraski et al., 1985)
This study was conducted in Wisconsin and compares conventional tillage with chisel, till-plant, and no-till. Although residue coverage was not reported in the paper, till-plant generally has lower than 30% residue cover and will not be considered conservation tillage. The study consisted of monitored rainfall events in September of 1980 and June and July of 1981 with monitored runoff from rain simulations in 1982 and 1983. As this study was only single runoff events, the P delivery numbers are low; however, there were opportunities for direct comparisons to be made. Initial P levels were similar in all trials 39, 45, 58, and 50 ppm for conventional, chisel, till-plant, and no-till, respectively in 1980). Values did not significantly change when re-measured in 1983 (39, 48, 54, 62 ppm). Data for total P and dissolved P loss were added to the practice table.

(Ginting et al., 1998b)
This study from west central Minnesota had conflicting results when comparing corn grain yield in ridge till and moldboard plow treatments. Overall there were little yield differences between tillage treatments, but the authors comment that in cold wet years, having more residue will likely reduce yields. This study also investigated any impact of manure on yields. Manure seemed to have an impact when using a ridge till system with optimal growing degree days, but any significance was lost in the moldboard plow treatments. Data were added to the table but the 1993 data were an average of both manure and no manure treatments.

(Ginting et al., 1998a)
This paper was a companion to the one above and contains the P data from the previously described study (Ginting et al., 1998b). Basic findings were that conventional tillage has more P loss than ridge tillage and that using manure as the P source generally reduces P loss. Total P, dissolved P, and particulate P were measured and estimates from figures in the paper were added to the practice table under the tillage and the manure sections. The tillage study compared moldboard plowing to ridge till. Moldboard is not the "normal" here in Iowa, so the study is not directly applicable to this effort, and the results are only shown for reference.

(Burwell et al., 1975)
This was a natural rainfall study done in west-central Minnesota (1966 through 1971). The authors investigated continuous clean cultivated ground, continuous corn, corn in rotation, oats in rotation, and hay in rotation. Phosphorus results were broken into seasonal periods, however, these were combined to produce an annual number when entered into the practice table. The general trend for total P was decreasing with increasing land cover (i.e., fallow at >5 kg/ha and hay in rotation <0.5 kg/ha). Although this is an interesting trend, no direct comparisons could be made to a corn-soybean rotation. which is common in Iowa. These data were not added to the practice table in this section.

(Eghball et al., 2000)
This research was done in Council Bluffs, Iowa, on a Monona Soil with 12% slopes with rainfall simulation. The study focused on comparing no-till with disked conditions, but also included nutrient application sources (manure, inorganic, and none). The type of manure is not explicitly stated, however, discussion in the introduction is about beef and dairy. Phosphorus in the inorganic fertilizer plots came from diammonium phosphate and was applied at 12 kg P/ha before spring tillage. There was no fertilizer incorporation in the no-till plots and immediately incorporated in the tillage plots. Findings suggest that less P is lost in no-till systems (when initially dry or wet) and more P may be lost from inorganic fertilizer (initially dry conditions). There was little in the way of statistical significance, but the data were entered in the practice table for tillage and source as there were definite trends (the buffer plots were not used in the tillage and source analysis). This study also used grass hedges between plots, which were added to the buffer section of the practice table.

(Laflen and Tabatabai, 1984)
This rainfall simulation study was done at two locations in Iowa. The duration of the rainfall was 60 minutes with, as expected, decreasing P levels as rainfall progressed. Additionally, the site with steeper slopes lost more P. The three tillage categories investigated were moldboard plow, chisel plow, and no-till. Phosphorus loss was decreasing in that order also. Data were added to the practice table.

(McIsaac et al., 1995)
This rainfall simulation study was done on a Catlin silt loam and a Tama silt loam in Illinois. Trends show that increased cover (no-till or strip-till) produces increased dissolved P runoff. This is similar to other studies. The chisel plow treatment in this study had the lowest dissolved P levels. Total P levels were not reported so the data were not added to the practice table.

(Mostaghimi et al., 1988)
This rainfall simulation study was done in Virginia with no-till and conventional tillage treatments along with no P application, subsurface injection of P, and surface application of P. The study found that total P is lower in the subsurface injection treatments than in the surface application treatments. Additionally, no-till treatments have lower P losses than conventional tillage systems. As this study was done in Virginia, no data were added to the practice table.

(Johnson et al., 1979)
This small watershed study was conducted near Castana, Iowa, from 1972 to 1975 on Monona, Ida, and Napier soils. There were six watersheds in the study and the authors point out results could be impacted by variations in watershed characteristics (slope, shape, etc.). The P application rate used in this study was 37 kg P/ha. Conventional tillage in this study was disking, plowing, disking and planting. The till-plant tillage in this study included disking and planting using a till-planter. The ridge-plant treatment only used a planter. Corn yields were also measured with this study and found that treatments tended to be similar, but till-plant was generally higher. The three year average of the treatments was 6.72, 7.48, and 6.59 Mg/ha for the conventional, till-plant, and ridge-plant treatments. Unfortunately, sampling methods changed after 1973 by only analyzing runoff samples for available P, and no nutrient data were collected in 1972. The 1973 data set was estimated from a figure in the publication but not added to the practice table as the study did not utilize chisel plowing.

(Andraski et al., 2003)
This rainfall simulation study was near Madison and Lancaster, Wis. Soils were Plano silt loam and Rozetta silt loam. The study included manure history and tillage treatments. The Madison manure treatments had dairy manure applied in the spring at a P rate of 88 kg P/ha with immediate incorporation into the soil. There were several manure application histories: 1995 and 1998 application, 1996 and 1999 application, and annually from 1994 to 1999. Tillage consisted of chisel plowing and field cultivating in the spring. The Lancaster site had fall surface applied dairy manure from 1993 to 1997 with fall chisel plowing (followed by disking before planting) and a no-till treatment. Phosphorus application rate at Lancaster was 79 kg P/ha on the manure treatments. All data is from rainfall simulations conducted in 2000 before planting and after harvest. There was no yield data available. All data were added to the practice table for both the tillage treatments and the manure treatments.

(Bundy et al., 2001)
This study was described in the "Placement of Phosphorus" section. Data has been added to the practice table.

(Randall et al., 1996)
This extension publication outlined research done at the research farm at Waseca, Minn., and included corn yield data for 1974 through 1977 and 1986 through 1988 with different tillage practices. No-till tended to have lower yields, however, the author comments it is not significant. The study also found moldboard plowing in the spring was less productive than in the fall. The data from 1974 to 1977 was reported as an average yield and the average was used for each year for analysis. Data has been added to the practice table.

(Baker and Laflen, 1982)
This study was described in the “Placement of Phosphorus” section. The data were not added to the practice table.

(Gold and Loudon, 1989)
This natural rainfall study was conducted from the middle of 1981 to the early part of 1984 in Michigan comparing moldboard-plow plots with chisel-plow plots. The study used a corn, dry beans, sugar beet, corn rotation. The moldboard-plow plots lost more P than the chisel-plow plots (1.2 kg P/ha/study period for moldboard and 0.83 kg P/ha/study period for chisel). Although informative, this dataset was not added to the practice table because this rotation is not used in Iowa.

(Mallarino et al., 2010a)
This study was described in the “Phosphorus Source” section. Data were added to the practice table.

(Mallarino et al., 2011b)
This study was described in the “Phosphorus Source” section. Data were added to the practice table.

(Singer et al., 2004)
This research was done near Boone, Iowa, and reported corn yields under different tillage practices between 1996 and 2002. The study also reported the impact of compost (bedded swine manure). Although the rotation used in the study was corn-soybean-wheat, corn yields were reported for each year of the study for each tillage practice so the data were added to the practice table. There was little difference in the practices.

(Singer et al., 2007)
This was a continuation (2003 and 2004) of the (Singer et al., 2004) study, but included additional information on nutrients contained in the crops. Corn yield data were added to the practice table.

(Kaiser et al., 2009)
This study reports results from rainfall simulation trials between 2004 and 2006 around Iowa. The primary focus of the study was to compare P loss with different application rates of poultry manure; however, since there was not a comparable commercial fertilizer treatment, only the tillage effect was examined here. Results show tillage reduces total P loss when compared to no-till and the more manure is added, the more P is lost. The dataset was added to the practice table; however, the compounding factor of inconsistent rainfall timing limited the use.

## Cover Crops

Limited data is available on the impact of cover crops on P delivery; however, (Dinnes, 2004) suggests that cover crops in applicable areas in Iowa may reduce P loads by 10 to 70% (50% over the long term).

(Kaspar et al., 2003)
This report summarizes work done on research plots west of Ames, Iowa. The study involved multiple treatments, however, only the cover crop (rye) and check (control) treatments are considered here. All plots were fertilized with 200 lb/ac of UAN, which was surface applied in the spring before corn. Each treatment had four replicates. Corn yields from 2000 and 2002 were 164 and 198 for the control plots while 164 and 176 for the cover crop plots. Soybean yields in 2001 were 46 for the control plots and 44 for the cover crop plots, which was not significantly different. This data has been summarized by (Kaspar et al., 2007).

(Kaspar et al., 2007)
This cover crop study in Iowa reported a corn yield response in year 1 of -9.7% with no change in year 3 and no change in soybean yield response in year 2 with a -6.7% response in year 4. Site year data has been added to the table for yield.

(Kaspar et al., 2001)
This study focused on the effects of small grain cover crops (rye and oat) on runoff and erosion. The study was performed near Ames, Iowa, between 1996 and 1998. Runoff and erosion were measured in a rainfall simulation setup. Authors found that in two of three years, interrill erosion rates were statistically lower than the control when using a rye cover crop and statistically lower in one of three years when using an oat cover crop. In two of two years rill erosion rates were statistically lower than the control with both cover crop treatments, and the rye cover crop was statistically lower than the oat cover crop. No P data were included in the paper, so the dataset was not added to the practice table.

(Qi et al., 2011)
This paper from Iowa looks at yields from a rye cover crop (on both corn and beans), and a living mulch (kura clover) with corn. Over the 4 years of the study, a rye cover crop before the corn phase showed a corn yield of 8.1 Mg/ha with a yield of 8.4 Mg/ha for the control. Rye before soybeans showed a soybean yield of 2.5 Mg/ha with a bean yield of 2.8 Mg/ha on the control. The kura clover living mulch was a continuous corn system which had a 4-year average yield of 2.8 Mg/ha. Site years have been added to the table for yield.

(Strock et al., 2004)
This paper is from southern Minnesota with three years of data. There was no statistically significant change in observed crop yields for either corn or soybeans and rye. The site years for yield have been added to the table. There was no statistically significant difference in yields.

(Pederson et al., 2010)
This report shows information from 4 years (2007 to 2010). There is a reduction in yield with the addition of a cover crop when comparing to spring UAN at 150 lbs N/ac. The study was conducted at the NERF site near Nashua, Iowa.

(Sawyer et al., 2010)
Results from ISU outlying research farms shows a substantial decrease in corn yields with the addition of a cover crop. There is little impact on soybean yields. This paper looked at information from four locations.

(PFI, 2011)
This report shows a significant reduction in corn yield at two locations in the study. There was one location where the cover crop treatment had a significantly increased corn yield. In general there was no significant difference in plots with cover crops compared to conventional agriculture.

(Kovar et al., 2011)
This rainfall simulation study done in Boone County, Iowa, was done in 2007 and 2008. The study compared plots with no P added, liquid swine manure knife injected, and liquid swine manure applied with a low-disturbance applicator. The study also included cover crop treatments. The P application rate was 53 kg P/ha for the knifed in plots and 88 kg P/ha for the low disturbance plots. Results showed more P was lost in the low disturbance plots in 2007 (more than in the control or the knifed in plots). In 2008 the no manure plots lost more P followed by the knifed in plots. In 2007 the presence of cover crops had no impact on P loss, but in 2008, P loss was significantly reduced with a cover crop. All data were added to the practice table.

## Cropping Changes (Extended Rotations and Crop Choice)

Any crop with increased residue will likely have increased dissolved P loss, but minimize erosion and the P lost with eroded soil.

(Dinnes, 2004)
This study reviews literature from around the country, very little is relevant to Iowa. The authors do make an attempt at estimating the applicability in Iowa (best professional judgment), which is 0% to 90% reduction in P load annually (50% over the long term).

(Benoit, 1973)
This study was done in Vermont, and not specifically included in this research; however, the conclusions on P were interesting. This study was on sloping soils that were tile drained and investigated nitrogen and P movement with different crops. Authors found up to 0.02 mg/L P was present in subsurface drainage (seemingly not dependent on crop) and up to 2.0, 2.0, and 3.0 mg/L lost from surface drainage for alfalfa, corn, and hay-pasture, respectively. These crops averaged 0.8, 0.7, and 0.9 mg/L for alfalfa, corn, and hay-pasture, respectively. This supports other studies showing more P loss (in the dissolved form) from land with more vegetative cover.

(Burwell et al., 1975)
This paper was described in the "Tillage and Residue Management" section. Again, no direct comparison could be made to a corn-soybean rotation so the data were not added to the practice table.

(Rehm et al., 1998)
This webpage from the University of Minnesota has a table with P loss of various land uses. These land uses are grass, no-till corn, conventional corn, and wheat/summer fallow and have total P losses of 7.05, 2.94, 13.75, and 1.43 lb P/ac, respectively. Additionally, this page has comparisons of tillage systems and placement; however, the tillage work was done in Indiana and the placement work was done in Virginia. Although specific references for the crop choice data were not provided, the data were added to the practice table.

(Young and Mutchler, 1976)
This study was done in Morris, Minn., with alfalfa and corn on frozen soils and was completed between 1972 and 1974. The overall message is that tillage in the fall will reduce P loss when planning on applying manure on frozen soils or on snow. If manure is applied during frozen conditions to alfalfa, much of the applied P is lost. Data were not added to the practice table, as manure application to frozen soils is not a common practice.

(Mallarino and Rueber, 2010)
This report from the Northern Research and Demonstration Farm in Iowa highlights corn yields with extended rotations. Data were summarized and added to the practice table.

(Kanwar et al., 2005)
This 6-year study had several plots with strip intercropping (corn/soybean/oat interseeded in berseem clover), an extended rotation (alfalfa/alfalfa/alfalfa/corn/soybean/oat), and a conventional rotation (corn/soybean). All fertilization was done in the spring with a sidedress application based on the late spring nitrate test (LSNT). Yields from all treatments were added to the practice table.

(Huggins et al., 2001)
This 3-year study from Minnesota investigated what happens with conversion from a continuous alfalfa or a CRP cropping system to a corn-corn-soybean rotation. This rotation does not exactly fit the intended rotation for this project, but it has been added to the practice table and will contribute to information about continuous corn and corn-soybean rotations.

(Liebman et al., 2008)
This 4-year study from Iowa investigates a number of cropping rotations including a 2-year (corn-soybean), a 3-year (corn-soybean-small grain + red clover green manure), and a 4-year (corn-soybean-small grain + alfalfa-alfalfa hay). There was a yield and economic analysis of the different rotations. Fertilizer was managed based on soil testing and included composted manure, urea applied at planting, and sidedressed UAN as needed. Phosphorus and potassium were also applied as needed. Crop yields were added to the practice table, but not the economic values.

## Perennial Crops/Perennial Biomass Crops

The advantage of perennial crops is the increased soil cover, which reduces soil erosion. Although dissolved P loss will likely increase, total P loss should decrease. Additionally, it may be possible to use perennial crops for reducing P levels in high P soils (Gaston et al., 2003). The Gaston study compared a number of crops with switchgrass and alfalfa resulting in the largest soil P change.

(Andrews, 2010)
This thesis reports rainfall simulation runoff P for several crop types including continuous corn, corn-soybeans, and switchgrass. Additionally, there are several management treatments as well – manure, fertilizer, and no nutrients. Each of the two switchgrass treatments was compared to an average of the corn followed by soybean and soybean followed by corn treatment so a comparison to a corn-soybean rotation could be made. The dataset was added to the practice table.

## Perennial Cover (Land Retirement – CRP)

The advantage of perennial crops is the increased soil cover, which reduces soil erosion. Although dissolved P loss will likely increase, total P loss should decrease.

(Schroeder and Kovar, 2008)
This study done in central Iowa investigates differences in soils under a continuously cropped system and a 13-year-old CRP system on the edge of the cropped ground. Although no runoff or P transport data is available, the study findings indicate CRP buffer locations may retain less P than crop ground, which would be a concern when using buffers or vegetated filter strips for P reduction. The paper doesn't mention, however, that there would still be sediment reduction, and dissolved P may increase. This dataset was not useable here and was not added to the practice table.

(Panuska et al., 2007)
This study was done in Wisconsin using the Wisconsin P-Index. Although results are based on modeling, the trend shown (decreasing P loss with increasing soil cover) is expected when comparing P loss from CRP and various row crops. Additionally, the presence or absence of manure has little to no impact on P loss. This dataset was not included in the practice table as results were based on modeling.

(Jokela and Russelle, 2010)
This magazine article comments on the reduction of P with the addition of perennial cover. Additionally, RUSLE 2 model results are shown with estimates of soil loss, which show a 90% reduction when moving from corn silage to alfalfa. Phosphorus reduction would have the same trend. These data were not included in the practice table as results were from modeling and did not specifically report P loss.

## Grazed Pastures

Unlike other perennial systems, grazed pastures may have increased P due to dung and increased erosion due to compaction and hoof damage; however, erosion is generally less than from cropping systems. Additionally, there are several ways to manage a pasture system including excluding livestock from streams, intensive grazing, rotational grazing, and seasonal grazing. (Dinnes, 2004) suggests, in any given year, there may be a 65 to 90% reduction in total P when comparing livestock exclusion to intensive grazing with a long term average of 75%; a -100 to 75% reduction in total P when comparing rotational grazing to intensive grazing with a long term average of 25%; a 0 to 80% reduction in total P when comparing seasonal grazing to intensive grazing with a long term average of 50%.

(Zaimes et al., 2008b)
This study investigated the total P in soil under multiple land uses (rotationally and intensively grazed pastures with and without cattle fenced out, row cropping) and conservation practices associated with the land uses. A number of sites across Iowa were included in this study in order to investigate impacts of soil and land form. No significant differences were observed in total P soil concentrations between the riparian areas in the study, however, central Iowa tended to have the lowest values. Authors suggest that once elevated, soil P is difficult to decrease with conservation practices. Authors also suggest limiting erosion is likely an important factor when attempting to limit P delivery to streams. There were no useable/comparable water quality data in this paper.

(Schwarte et al., 2011)
This study was conducted in 2008 and 2009 in central Iowa (near Nevada) and investigated six 12.1 ha cool-season grass pastures. All data were collected as part of a rainfall simulation study. Soils were listed as Ackmore and Nodaway silt loams. There was no fertilizer applied for three years before or during the study. As the treatments were continuous stocking with restricted cattle access, continuous stocking with unrestricted access, and rotational stocking, there was no useable control comparison, however, the authors provide a relationship for P loss on pastures based on the percentage of bare ground:

The $R^2$ value on this relationship is 0.4302 and x is the percentage of bare ground. As this was not directly applicable to this project, the data were not added to the practice table.

(Nellesen et al., 2011)
This study was at the same location as (Schwarte et al., 2011) on the same plots but using 2005 to 2007 data. This study used natural rainfall rather than simulations. There were no statistically significant differences in annual P loss with any of the grazing treatments, but the continuously grazed unrestricted treatments tended to have higher loads (13.2 g P/m of stream as a 3-year average). The rotationally grazed treatments study average was 10.3 g P/m of stream and the continuously grazed restricted access treatments averaged 5.5 g P/m of stream. There were some significant differences in certain months of the study. As this was not directly applicable to this project, the data were not added to the practice table.

(Haan et al., 2003)
Refer to (Haan et al., 2006) for information on this study, as they are the same.

(Haan et al., 2006)
This pasture study was conducted near Nevada, Iowa, as a rainfall simulation from 2001 to 2003. Pasture slopes were 0-15 degrees with bromegrass on Downs silt loams, Gara loam, and Colo-Ely complex. No additional P was applied during the study period. Results showed that more intensely grazed pastures have more runoff and a higher P load. In this study slope had little impact on P loss. Conclusions were the more ground cover, the less P loss. As this was not directly applicable to this project, the data were not added to the practice table.

(Schuman et al., 1973)
This was a small watershed study in the Missouri Valley Deep Loess Soils in Treynor, Iowa, from 1969 to 1971. Specific soil types were Monona, Ida, and Napier silt loams. Slopes ranged from anywhere between 2 and 18%. There were four treatments, three with corn as the primary crop and one with bromegrass. The corn treatments had a 39 kg P/ha treatment and two 97 kg P/ha treatments (one cropped on the contour and one with level terraces). Rate data has been added to the "Phosphorus Application Rate" section. The comparison made here is between corn ground and bromegrass with rotational grazing. A P application rate of 39 kg P/ha was applied to both watersheds. There was little difference in P loss between the treatments in 1969, but the bromegrass treatment was substantially lower in 1970 and 1971. Data has been added to the practice table.

(Smith et al., 1992)
This study in Oklahoma looked at different grazing management techniques. The findings show the extent of vegetation is likely a better indicator of P loss (with erosion) than vegetation type or management scheme. Authors suggest the presence of permanent vegetation reduces P loss below levels expected for tilled croplands. As this study was done in Oklahoma and no direct comparisons are made to a corn-soybean rotation, the dataset was not included in the practice table.

(Alexander et al., 2004)
Based on watershed modeling with the Spatially Referenced Regression on Watershed attributes (SPARROW) model, nationally P loads from pastured areas are approximately 18% lower than cropland (0.9 kg P/ha/yr compared to 1.1 kg P/ha/yr). As this was national modeling data, values were not added to the practice table.

## Wetlands

Wetlands have potential to remove P from influent water primarily by allowing sediment to settle out; however, dissolved P can also be removed if the presence of iron or aluminum-rich materials is high (Kadlec and Knight, 1996). Additionally, sorption sites in wetland soils can become saturated with P, and, if the water chemistry changes, wetlands could become a source of P. Although limited data is available for wetlands in the Midwest, (Dinnes, 2004) suggests on an annual basis, a wetland would remove between -20 and 50% of the P with a long-term removal of 20%.

(Kovacic et al., 2000)
Although this study was done in eastern Illinois, it was reviewed as there is limited data available on P removal in Iowa. The three wetlands reviewed here were monitored between 1995 and 1997 and show a P removal in six of the nine site years. The wetland-to-watershed ratio ranged from 3.1% to 5.9% with P release more common in the wetland with a relatively larger drainage area. Data were added to the practice table, but only for comparative purposes as Iowa-specific data should be available in the near future.

(Miller et al., 2002)
Although this study was done in eastern Illinois, it was reviewed as there is limited data available on P removal in Iowa. The study ran from 1994 to 1997; however, the wetlands primarily received water from tile drained watersheds, which had very little P. Additionally, only orthophosphate concentrations were reported so the dataset was not included in the practice table.

(Kadlec and Hey, 1994)
This wetland study was conducted north of Chicago, Illinois, in 1989 and 1990 and consisted of four wetlands in-line with a river. Water was pumped into these wetlands and allowed to free flow out. This paper only reported total P concentrations and the authors suggest 75% of the P was being removed on average (at least in the first two years of running). The authors also note any long term removal of P would come in the form of sedimentation. The dataset was not added to the practice table.

## Buffers

This section includes information on traditional buffers on the edge of a field as well as buffers in the field. This could also include grassed waterways, however, the focus is on actual buffers. Several factors are important in buffer performance including land slope, buffer width, buffer vegetation, and the presence/absence of concentrated water flow. That being said, in-field buffers may provide a 20 to 70% reduction in total P annually with a long-term reduction of 50% (Dinnes, 2004). Edge-of-field buffers may provide 25 to 65% reduction in Iowa with a longer-term reduction of 45% (Dinnes, 2004).

(Lee et al., 1999)
This study detailed a rainfall simulation on switchgrass and cool-season grass buffers. Sediment, total P, and PO4-P were measured with removals calculated. The switchgrass buffers performed better for every pollutant in every case, as did increasing the width of the buffer. Although only for a single storm and only a simulation, removal data were added to the practice table.

(Zaimes et al., 2008a)
This study is a companion to (Zaimes et al., 2008b) and investigates streambank erosion rates from different agricultural systems. Erosion results showed more streambank erosion from the row crop system with an average erosion rate of 239 mm/yr over a 3-year period. In contrast, riparian forest buffers showed an average of 15 mm/yr over the same period in northeast Iowa and 46 mm/yr in central Iowa. Continuous and intensive rotational pastures were between 101 and 171 mm/yr. Associated with this erosion is P loss, which had a similar trend to erosion (see table below). Since streambank contributions are not being specifically investigated at this point, it will not be reported.

TABLE 5. Soil and Total Phosphorus Losses From Streambank Erosion Under Different Riparian Land-Uses in Three Iowa Regions.[1]

| | Stream Reach Length | Severely Eroding Streambank | | | | Streambank Soil Loss[3] | | | Streambank Phosphorus Loss[4] | |
|---|---|---|---|---|---|---|---|---|---|---|
| Riparian Land-Use | Total (km) | Length % | SD[5] | Area ($m^2$) | Bulk Density[2] ($tonne/m^3$) | Total (tonne/year) | Unit length (tonne/km/year) | Streambank Soil Phosphorus Concentrations[2] (kg/tonne) | Total (kg/year) | Unit Length (kg/km/year) |
| Central | | | | | | | | | | |
| Row-cropped fields | 1.6 | 44 (6) | a | 1657 | 1.23 | 487 | 304 | 354 | 172 | 108 |
| Continuous pastures | 1.7 | 39 (6) | ab | 1999 | 1.35 | 448 | 264 | 349 | 156 | 92 |
| Rotational pastures | 1.3 | 25 (6) | b | 899 | 1.31 | 122 | 94 | 398 | 49 | 37 |
| Grass filters | 1.6 | 16 (6) | bc | 615 | 1.16 | 76 | 47 | 303 | 23 | 14 |
| Riparian forest buffers | 1.4 | 14 (6) | bc | 430 | 1.24 | 25 | 18 | 350 | 9 | 6 |
| Northeast | | | | | | | | | | |
| Continuous pastures | 1.6 | 38 (5) | a | 1935 | 1.15 | 381 | 238 | 518 | 197 | 123 |
| Intensive rotational pastures | 1.5 | 27 (5) | a | 1125 | 1.20 | 230 | 153 | 432 | 99 | 66 |
| Cattle fenced out of streams | 0.8 | 11 (6) | b | 203 | 1.16 | 5 | 6 | 464 | 2 | 3 |
| Riparian forest buffers | 0.8 | 10 (6) | b | 244 | 1.10 | 4 | 5 | 479 | 2 | 2 |
| Southeast | | | | | | | | | | |
| Continuous pastures | 1.8 | 54 (5) | a | 2661 | 1.32 | 355 | 197 | 360 | 128 | 71 |
| Rotational pastures | 1.5 | 54 (6) | a | 2403 | 1.36 | 399 | 266 | 459 | 183 | 122 |
| Intensive rotational pastures | 0.7 | 32 (6) | b | 371 | 1.28 | 87 | 124 | 531 | 46 | 66 |
| Cattle fenced out of stream | 0.3 | 16 (6) | bc | 239 | 1.32 | 18 | 61 | 555 | 10 | 34 |
| Grass filters | 0.7 | 16 (6) | c | 289 | 1.29 | 15 | 22 | 406 | 6 | 9 |

[1]The mean rainfall that each riparian land-use reach received was used as a covariate to estimate streambank erosion rate. In parentheses is the standard error.
[2]Data from Zaimes *et al.*, 2008.
[3]The product of streambank erosion rate, bulk density, and severely eroding bank area for each riparian land-use within a region was used to estimate its *total soil loss* from the streambanks. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use of a region. Streambank *soil loss per unit of stream length* was estimated by dividing the streambank phosphorus loss for each riparian land-use by its total stream reach length within a region.
[4]The product of streambank erosion rate, bulk density, severely eroding bank area, and average soil phosphorus concentration for each riparian land-use within a region, was used to estimate its *total phosphorus loss* from the streambanks. Streambank erosion rate was the average rate of all the pin plots in the riparian land-use of a region. Streambank *phosphorus loss per unit of stream length* was estimated by dividing the streambank soil loss for each riparian land-use by its total stream reach length within a region.
[5]SD, significant differences. In this column the different letters indicate significant differences ($p$-value <0.10) among riparian land uses.

(Osborne and Kovacic, 1993)
This research was done in eastern Illinois in 1988 and 1989. The study setup included an entirely cropped area up to the stream, a cropped area with a forested buffer (16 m wide), and a cropped area with a grass buffer (39 m wide). Although drainage concentrations were not monitored, data from shallow and deep lysimeters as well as piezometers was reported and will be used here. Results are averaged over two years (a corn/soybean rotation), and will be reported double in the site year table to maintain annual weighting for this study. Data were estimated from the provided figure in the publication. In brief, both buffers tended to increase P concentrations in the groundwater with other data suggesting P is reduced in surface runoff. Surface runoff data were added to the practice table as concentrations.

(Lee et al., 2003)
This study considers two buffers (switchgrass at 7.1 m and a combination switchgrass and bushy vegetation at 16.3 m) and includes 1997 and 1998 data. The authors report results from the three largest storms of the two years. Although these are not annual values, they serve as a good comparison between runoff from crop ground before and after buffers. Dataset was added to the practice table.

(Lee et al., 2000)
This study considers two buffers (switchgrass at 7.1 m and a combination switchgrass and bushy vegetation at 16.3 m). Authors present results from rainfall simulation in this paper. Results show between 46 and 93% reduction in total P depending on the length and intensity of rainfall. Dataset was added to the practice table.

(Eghball et al., 2000)
See discussion under the "Tillage and Residue Management" section.

(Udawatta et al., 2002)
This small watershed study in northeast Missouri ran from 1997 to 1999 and focused on two buffer practices — grass strips on the contour and agroforestry strips on the contour. The strips were 4.5 m wide with 36.5 m spacing. All watersheds ran through a grassed waterway before samples were collected, so results may be artificially low. The goal of the paper was to come up with predictions on sediment/P/nitrogen loss; however, they reported average annual loss of the two practices when compared to the control (no buffers). Over the three year period, the contour grass buffers had a slightly higher P loss than the control (1.1 kg P/ha/yr compared to 1.0 kg P/ha/yr); however the authors suggest the reductions started to occur in 1998, which showed a 3.7% reduction with the grass buffers and an 18% reduction with the agroforestry buffers. Data has been added to the practice table and reproduced three times for the 3-year average.

(Young et al., 1980)
This rainfall simulation study was done in west central Minnesota using runoff from feedlots and buffers with various types of vegetation. The buffer with corn reduced total P the most when compared with orchardgrass, sorghum-sudangrass, or oats, which was likely due to higher infiltration rates on recently tilled and planted (simulated rainfall 30 to 45 days after planting). The other treatments were also tilled and planted; however, corn is likely the fastest growing crop. The dataset was not added as it was not completely applicable to this study.

(Webber et al., 2010)
This natural rainfall study was done in central Iowa looking at different sized buffers filtering runoff from grazed land with differing grazing management schemes. Data showed there were no significant differences between orthophosphate loads from buffers that were 10% of drainage area or 20% of

drainage area, although the larger buffer tended to have lower orthophosphate loads. Total P loads were not reported so these data were not added to the practice table.

(Schroeder and Kovar, 2008)
See description in the "Perennial Cover (Land Retirement – CRP)" section. The dataset was not added to the practice table.

## Erosion Control Practices and Structures

This section includes terraces and any other practice that may be used to limit erosion or P loss. Estimated annual reduction in Iowa for terraces is -20 to 90% with a long-term average of 50% (Dinnes, 2004). Ponds are generally not built for sediment removal in the agricultural setting but may be effective at removing sediment, and any P sorbed to that sediment.

(Hanway and Laflen, 1974)
This study investigated nutrient losses from tile-outlet terraces. There was no real control with this work to compare P loss from terraced and non-terraced ground. Information from the three-year study was added to the table for possible future comparison. Additionally, the authors make the case that P concentrations in surface runoff had the same trends as sediment concentrations. Phosphorus concentrations in tile drainage water were much lower than in surface runoff. Soluble P concentrations were NOT related to sediment in tile water or runoff, were generally low in both tile water and runoff water (lower in tile), and were related to the crop-available P (STP) in the surface soil.

(Schuman et al., 1973)
This study was described in the "Grazed Pastures" section. Data from the level terrace treatment was added to the practice table compared to the other corn treatment at the same P application rate.

(Burwell et al., 1974)
This study was conducted in 1970 and 1971 and compared two watersheds in southwest Iowa. The control was a contour farmed 33.6 ha watershed and the practice was level terraces on 85% of a 157.5 ha watershed approximately 18 km away. Results show the level terrace practice can reduce total P by between 50 and 60% when compared to contour farmed ground. The data from the contour farmed watershed is similar, although not the same, as that reported by (Schuman et al., 1973). Since this paper did not reference the other, they are assumed to be different. Data were added to the practice table.

## Phosphorus Loss in Drainage

This is for informational purposes only and is intended to provide justification for not emphasizing loss in drainage water with this study. Although loss of P in drainage will not be considered here, there is a possibility for P levels to increase with managed drainage by around 10% over the long term (Dinnes, 2004). Additionally, a study by Allen et al., (2012) shows very low concentrations moving in subsoil. Soil-test P trailed off to trace amounts as samples were taken at increasing distances from the P source after only 0.75 to 1.0 m.

(Hanway and Laflen, 1974)
See description under "Erosion Control Practices and Structures" where the study was described.

(Baker et al., 1975)
This study was done at the Iowa State Agronomy and Agricultural Engineering farm in Boone County, Iowa. Drainage phosphate-P concentrations ranged from 0 to 0.009 kg/ha from the plots, which had an oat, corn, oat, corn, soybean rotation from 1969 to 1973. Although this data cannot be directly compared to anything, the data set was added to the table for purposes of cataloguing expected P concentrations leaving tile-drained landscapes.

(Benoit, 1973)
See study description under “Cropping Changes (Extended Rotations and Crop Choice).”

(Fraterrigo and Downing, 2008)
This paper reviewed parameters that had an impact on lake total P and found a slight correlation between tile-drained land and “low transport capacity” watersheds, and no correlation in “high transport capacity” watersheds. Authors suggest tile drainage in the low-transport watersheds changes the P form from what it would have been (particulate P) to dissolved P. Additionally, this paper found a correlation between urban (commercial) land use, point sources (wastewater treatment), and agricultural land to total P in lakes. Also, a major factor was the type of lake. Although this study was done in Iowa, it was not used as there was not useable data for this project.

## References Cited


Alexander, R., R. Smith, and G. Schwarz. 2004. Estimates of diffuse phosphorus sources in surface waters of the United States using a spatially referenced watershed model. Water Science and Technology 49:1-10.

Allen, B., and A. Mallarino. 2008. Effect of liquid swine manure rate, incorporation, and timing of rainfall on phosphorus loss with surface runoff. Journal of Environmental Quality 37:125-137.

Allen, B., A. Mallarino, J. Klatt, J. Baker, and M. Camara. 2006. Soil and surface runoff phosphorus relationships for five typical USA midwest soils. Journal of Environmental Quality 35:599-610.

Allen, B.L., A.P. Mallarino, J.F. Lore, J.L. Baker, and M.U. Haq. 2012. Phosphorus lateral movement through subsoil to subsurface tile drains. Soil Science Society of America Journal 76.

Andraski, B., D. Mueller, and T. Daniel. 1985. Phosphorus losses in runoff as affected by tillage. Soil Science Society of America Journal 49:1523-1527.

Andraski, T., L. Bundy, and K. Kilian. 2003. Manure history and long-term tillage effects on soil properties and phosphorus losses in runoff. Journal of Environmental Quality 32:1782-1789.

Andrews, A.A. 2010. Phosphorus loss with surface runoff as affected by bioenergy-based residue and nutrient management systems for an Iowa loam soil. MS Thesis, Iowa State Univ., Ames, IA.

Baker, J.L., and J.M. Laflen. 1982. Effects of corn residue and fertilizer management on soluble nutrient runoff losses. Transactions of the ASAE 25:344-348.

Baker, J.L., K.L. Campbell, H.P. Johnson, and J.J. Hanway. 1975. Nitrate, phosphorus, and sulfate in subsurface drainage water. Journal of Environmental Quality 4:406-412.

Bakhsh, A., R.S. Kanwar, and D.L. Karlen. 2005. Effects of liquid swine manure applications on NO3-N leaching losses to subsurface drainage water from loamy soils in Iowa. Agriculture, Ecosystems and Environment 109:118-128.

Barbazan, M., A. Mallarino, and J. Sawyer. 2009. Liquid swine manure phosphorus utilization for corn and soybean production. Soil Science Society of America Journal 73:654-662.

Barisas, S.G., J.L. Baker, H.P. Johnson, and J.M. Laflen. 1978. Effect of tillage systems on runoff losses of nutrients, a rainfall simulation study. Transactions of the ASAE 21:893-897.

Benoit, G.R. 1973. Effect of agricultural management of wet sloping soil on nitrate and phosphorus in surface and subsurface water. Water Resources Research 9:1296-1303.

Bermudez, M., and A. Mallarino. 2007. Impacts of variable-rate phosphorus fertilization based on dense grid soil sampling on soil-test phosphorus and grain yield of corn and soybean. Agronomy Journal 99:822-832.

Bundy, L., T. Andraski, and J. Powell. 2001. Management practice effects on phosphorus losses in runoff in corn production systems. Journal of Environmental Quality 30:1822-1828.

Burwell, R.E., D.R. Timmons, and R.F. Holt. 1975. Nutrient transport in surface runoff as influenced by soil cover and seasonal periods. Soil Science Society of America Journal 39:523-528.

Burwell, R.E., G.E. Schuman, R.F. Piest, R.G. Spomer, and T.M. McCalla. 1974. Quality of water discharged from two agricultural watersheds in southwestern Iowa. Water Resources Research 10:359-365.

Chinkuyu, A.J., R.S. Kanwar, J.C. Lorimor, H. Xin, and T.B. Bailey. 2002. Effects of laying hen manure application rate on water quality. Transactions of the ASAE 45:299-308.

Converse, J.C., G.D. Bubenzer, and W.H. Paulson. 1976. Nutrient losses in surface runoff from winter spread manure. Transactions of the ASAE 19:517-519.

Daverede, I., A. Kravchenko, R. Hoeft, E. Nafziger, D. Bullock, J. Warren, and L. Gonzini. 2004. Phosphorus runoff from incorporated and surface-applied liquid swine manure and phosphorus fertilizer. Journal of Environmental Quality 33:1535-1544.

Dick, W., E. McCoy, W. Edwards, and R. Lal. 1991. Continuous application of no-tillage to Ohio soils. Agronomy Journal 83:65-73.

Dinnes, D.L. 2004. Assessments of practices to reduce nitrogen and phosphorus nonpoint source pollution of Iowa's surface waters. Iowa Department of Natural Resources; USDA-ARS National Soil Tilth Laboratory, Ames, Iowa.

Eghball, B., J. Gilley, L. Kramer, and T. Moorman. 2000. Narrow grass hedge effects on phosphorus and nitrogen in runoff following manure and fertilizer application. Journal of Soil and Water Conservation 55:172-176.

Erickson, B. 2008. Corn/soybean rotation literature summary.

Fraterrigo, J., and J. Downing. 2008. The influence of land use on lake nutrients varies with watershed transport capacity. Ecosystems 11:1021-1034.

Gaston, L.A., T.L. Eilers, J.L. Kovar, D. Cooper, and D.L. Robinson. 2003. Greenhouse and field studies on hay harvest to remediate high phosphorus soil. Communications in Soil Science and Plant Analysis 34:2085-2097.

Gburek, W.J., and W.R. Heald. 1974. Soluble phosphate output of an agricultural watershed in Pennsylvania. Water Resources Research 10:113-118.

Gburek, W.J., and A.N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. J. Env. Qual. 27:267-277.

Gessel, P.D., N.C. Hansen, J.F. Moncrief, and M.A. Schmitt. 2004. Rate of fall-applied liquid swine manure: Effects on runoff transport of sediment and phosphorus. Journal of Environmental Quality 33:1839-1844.

Gilley, J.E., and L.M. Risse. 2000. Runoff and soil loss as affected by the application of manure. Transactions of the ASAE 43:1583-1588.

Ginting, D., J. Moncrief, S. Gupta, and S. Evans. 1998a. Interaction between manure and tillage system on phosphorus uptake and runoff losses. Journal of Environmental Quality 27:1403-1410.

Ginting, D., J. Moncrief, S. Gupta, and S. Evans. 1998b. Corn yield, runoff, and sediment losses from manure and tillage systems. Journal of Environmental Quality 27:1396-1402.

Gold, A.J., and T.L. Loudon. 1989. Tillage effects on surface runoff water quality from artificially drained cropland. Transactions of the ASAE 32:1329-1334.

Haan, M.M., J.R. Russell, W.J. Powers, J.L. Kovar, and J.L. Benning. 2006. Grazing management effects on sediment and phosphorus in surface runoff. Rangeland Ecology & Management 59:607-615.

Haan, M.M., J.R. Russell, W.J. Powers, S.K. Mickelson, S.I. Ahmed, J.L. Kovar, and R.C. Schultz. 2003. Effects of grazing management on sediment and phosphorus in runoff, p. 381-386, *In* T. J. Sauer, (ed.) Animal, Agricultural and Food Processing Wastes IX; Proceedings of the Ninth International Symposium. ed. ASAE, St. Joseph, MI.

Hanway, J.J., and J.M. Laflen. 1974. Plant nutrient losses from tile-outlet terraces. Journal of Environmental Quality 3:351-356.

Hongthanat, N., J. Kovar, and M. Thompson. 2011. Sorption indices to estimate risk of soil phosphorus loss in the Rathbun Lake watershed, Iowa. Soil Science 176:237-244.

Huggins, D.R., G.W. Randall, and M.P. Russelle. 2001. Subsurface drain losses of water and nitrate following conversion of perennials to row crops. Agronomy Journal 93:477-486.

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

Johnson, D.D., J.M. Kreglow, S.A. Dressing, R.P. Maas, F.A. Koehler, and F.J. Humenik. 1982. Best management practices for agricultural nonpoint source control, 1. Animal Waste. North Carolina State Univ., Raleigh, NC.

Johnson, H.P., J.L. Baker, W.D. Shrader, and J.M. Laflen. 1979. Tillage system effects on sediment and nutrients in runoff from small watersheds. Transactions of the ASAE 22:1110-1114.

Jokela, B., and M. Russelle. 2010. Benefits of perennial forages for soils, crops and water quality. Progressive Forage Grower. Available at http://progressiveforage.com/index.php?option=com_content&view=article&id=3246:benefits-of-perennial-forages-for-soils-crops-and-water-quality&catid=83:other-forage&Itemid=137.

Kadlec, R.H., and D.L. Hey. 1994. Constructed wetlands for river water-quality improvement. Water Science and Technology 29:159-168.

Kadlec, R.H., and R.L. Knight. 1996. Treatment wetlands Lewis Publishers, Boca Raton, FL.

Kaiser, D.E., A.P. Mallarino, M.U. Haq, and B.L. Allen. 2009. Runoff phosphorus loss immediately after poultry manure application as influenced by the application rate and tillage. Journal of Environmental Quality 38:299-308.

Kanwar, R.S., and J.L. Baker. 1993. Tillage and chemical management effects on groundwater quality. p. 455-459 Proc. Agricultural Research to Protect Water Quality1993. Soil and Water Conservation Society.

Kanwar, R.S., R.M. Cruse, M. Ghaffarzadeh, A. Bakhsh, D.L. Karlen, and T.B. Bailey. 2005. Corn-soybean and alternative cropping systems effects on NO3-N leaching losses in subsurface drainage water. Applied Engineering in Agriculture 21:181-188.

Kaspar, T., J. Radke, and J. Laflen. 2001. Small grain cover crops and wheel traffic effects on infiltration, runoff, and erosion. Journal of Soil and Water Conservation 56:160-164.

Kaspar, T., D. Jaynes, T. Moorman, and T. Parkin. 2003. Reducing nitrate levels in subsurface drain water with organic matter incorporation. Final Report to the American Farm Bureau Foundation for Agriculture. USDA-ARS National Soil Tilth Laboratory, Ames, IA.

Kaspar, T.C., D.B. Jaynes, T.B. Parkin, and T.B. Moorman. 2007. Rye cover crop and gamagrass strip effects on NO3 concentration and load in tile drainage. Journal of Environmental Quality 36:1503-1511.

Klatt, J.G., A.P. Mallarino, J.A. Downing, J.A. Kopaska, and D.J. Wittry. 2003. Soil phosphorus, management practices, and their relationship to phosphorus delivery in the Iowa Clear Lake agricultural watershed. Journal of Environmental Quality 32:2140-2149.

Kleinman, P., A. Sharpley, B. Moyer, and G. Elwinger. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. Journal of Environmental Quality 31:2026-2033.

Koehler, F.A., F.J. Humenik, D.D. Johnson, J.M. Kreglow, S.A. Dressing, and R.P. Maas. 1982. Best management practices for agricultural nonpoint source control, II. Commercial Fertilizer. North Carolina State Univ., Raleigh, NC.

Kovacic, D.A., M.B. David, L.E. Gentry, K.M. Starks, and R.A. Cooke. 2000. Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. Journal of Environmental Quality 29:1262-1274.

Kovar, J., T. Moorman, J. Singer, C. Cambardella, and M. Tomer. 2011. Swine manure injection with low-disturbance applicator and cover crops reduce phosphorus losses. Journal of Environmental Quality 40:329-336.

Laflen, J.M., and T.S. Colvin. 1981. Effect of crop residue on soil loss from continuous row cropping. Transactions of the ASAE 24:605-609.

Laflen, J.M., and M.A. Tabatabai. 1984. Nitrogen and phosphorus losses from corn-soybean rotations as affected by tillage practices. Transactions of the ASAE 27:58-63.

Laflen, J.M., J.L. Baker, R.O. Hartwig, W.F. Buchele, and H.P. Johnson. 1978. Soil and water loss from conservation tillage systems. Transactions of the ASAE 21:881-885.

Lal, R., T. Logan, and N. Fausey. 1990. Long-term tillage effects on a Mollic Ochraqualf in North-West Ohio. III. Soil Nutrient Profile. Soil & Tillage Research 15:371-382.

Lawlor, P.A., M.J. Helmers, J.L. Baker, S.W. Melvin, and D.W. Lemke. 2011. Comparison of liquid swine manure and ammonia nitrogen application timing on subsurface drainage water quality in Iowa.

Lee, K.-H., T. Isenhart, and R. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. Journal of Soil and Water Conservation 58:1-8.

Lee, K.-H., T.M. Isenhart, R.C. Schultz, and S.K. Mickelson. 1999. Nutrient and sediment removal by switchgrass and cool-season grass filter strips in central Iowa, USA. Agroforestry Systems 44:121-132.

Lee, K.-H., T. Isenhart, R. Schultz, and S. Mickelson. 2000. Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. Journal of Environmental Quality 29:1200-1205.

Liebman, M., L.R. Gibson, D.N. Sundberg, A.H. Heggenstaller, P.R. Westerman, C.A. Chase, R.G. Hartzler, F.D. Menalled, A.S. Davis, and P.M. Dixon. 2008. Agronomic and economic performance characteristics of conventional and low-external-input cropping systems in the central Corn Belt. Agronomy Journal 100:600-610.

Mallarino, A.P. 1997. Interpretation of soil phosphorus tests for corn in soils with varying pH and calcium carbonate content. J. Prod. Agric. 10:163-167.

Mallarino, A. 2011. Phosphorus loss through subsurface tile drainage, *In* R. Christianson, (ed.), Presentation made to the 2011 Drainage Research Forum in Okoboji, Iowa.

Mallarino, A.P. 2012. Proportion of very high, high, optimum, low, and very low soil test phosphorus at 14 study sites. Iowa State Univ. (unpublished).

Mallarino, A.P., and M.U. Haq. 2007.On-farm demonstration of the impact of phosphorus soil-test levels and management on phosphorus loss with surface runoff. Final Project Report to the Iowa Dep. of Nat. Resources. Iowa State Univ., Ames, IA.

Mallarino, A.P., and M.U. Haq. 2008. Phosphorus loss with surface runoff shortly after application of poultry manure and fertilizer without incorporation into the soil. Final Project Report to the Iowa Egg Council. Iowa State Univ., Ames, IA.

Mallarino, A.P., and M.U. Haq. 2012. Alum and gypsum treated poultry manure and fertilizer phosphorus losses in runoff without incorporation into the soil. Final Project Report to the Iowa Egg Council. Iowa State Univ., Ames, IA.

Mallarino, A.P., M.U. Haq, and M.J. Helmers. 2010a. Impacts of crop, biomass harvest systems, and nutrient management on soil and phosphorus loss with surface runoff. Iowa State Univ., Ames, IA. Progress Report, Integrated Farm/Livestock Management Project 07IFLM002.

Mallarino, A.P., M.U. Haq, and M.J. Helmers. 2011b. Impacts of crop, biomass harvest systems, and nutrient management on soil and phosphorus loss with surface runoff. Progress Report, Integrated Farm/Livestock Management Project 07IFLM002. Iowa State Univ., Ames, IA.

Mallarino, A.P., M.U. Haq, M.J. Helmers, A.A. Andrews, C.H. Pederson, and R.E. Rusk. 2010b. Tillage, cropping, harvest, and nutrient management systems impacts on phosphorus loss with surface runoff: a research update. Proc. Integrated Crop Management, Ames, IA. Iowa State Univ. Extension.

Mallarino, A.P., M.U. Haq, and K. Pecinovsky. 2005. Runoff phosphorus loss as affected by tillage, fertilizer, and swine manure phosphorus management in corn-soybean production systems. Annual reports. Northeast Iowa Research and Demonstration Farm. Iowa State Univ., Ames, IA.

Mallarino, A., B. Hill, and K. Culp. 2011a. Soil-test P summaries. Iowa State Univ. soil testing and plant analysis laboratory, 2006-2010 (unpublished).

Mallarino, A.P., and J. Prater. 2007. Corn and soybean grain yield, phosphorus removal, and soil-test response to long-term phosphorus fertilization strategies. p. 241-253 Proc. Integrated Crop Management, Ames, IA2007. Iowa State Univ. Extension Agribusiness Education Program.

Mallarino, A., and D. Rueber. 2010. Yield of corn, soybean, and oats as affected by long-term crop rotation and nitrogen fertilization of corn. Annual reports. Northern Iowa Research and Demonstration Farm. Iowa State Univ., Ames, IA.

Mallarino, A.P., and D.J. Wittry. 2005. Pilot implementation of an environmental phosphorus assessment tool for Iowa. Final Project Report. Iowa Department of Natural Resources, Des Moines, IA.

Mallarino, A., and D. Wittry. 2010. Crop yield and soil phosphorus as affected by liquid swine manure phosphorus application using variable-rate technology. Soil Science Society of America Journal 74:2230-2238.

Mallarino, A., B. Stewart, J. Baker, J. Downing, and J. Sawyer. 2002. Phosphorus indexing for cropland: Overview and basic concepts of the Iowa phosphorus index. Journal of Soil and Water Conservation 57:440-447.

McIsaac, G., J.K. Mitchell, and M.C. Hirshi. 1993. Nitrogen and phosphorus concentrations in runoff from corn and soybean tillage systems. p. 230-232 Proc. Agricultural Research to Protect Water Quality, Minneapolis, Minnesota1993. Soil and Water Conservation Society.

McIsaac, G.F., J.K. Mitchell, and M.C. Hirschi. 1995. Dissolved phosphorus concentrations in runoff from simulated rainfall on corn and soybean tillage systems. Journal of Soil and Water Conservation 50:383-388.

McKenna, D. 2009. Alternatives and costs of reducing agricultural nutrient losses to surface water, pp. 25. Illinois Department of Agriculture.

Miller, P.S., J.K. Mitchell, R.A. Cooke, and B.A. Engel. 2002. A wetland to improve agricultural subsurface drainage water quality. Transactions of the ASAE 45:1305-1317.

Mostaghimi, S., J.M. Flagg, T.A. Dillaha, and V.O. Shanholtz. 1988. Phosphorus losses from cropland as affected by tillage systems and fertilizer application method. Water Resources Bulletin 24:735-742.

Nellesen, S., J. Kovar, M. Haan, and J. Russell. 2011. Grazing management effects on stream bank erosion and phosphorus delivery to a pasture stream. Canadian Journal of Soil Science 91:385-395.

NRCS. 2004. Iowa Technical Note No. 25: Iowa Phosphorus Index, pp. 32. NRCS - Iowa, ftp://ftp-fc.sc.egov.usda.gov/IA/technical/Technot25Aug04.pdf.

Osborne, L.L., and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258.

Panuska, J., L. Good Ward, and R. Wolkowski. 2007. Converting CRP land to corn: Minimizing phosphorus loss, *In* U. o. Wisconsin, (ed.), Vol. A3831. University of Wisconsin-Extension, Madison, WI.

PDEP. 2006. Pennsylvania Stormwater Best Management Practices Manual.

Pederson, C., R. Kanwar, M. Helmers, and A. Mallarino. 2010. Impact of liquid swine manure application and cover crops on groundwater water quality. Iowa State Univ., Ames, IA.

PFI. 2011. Cover crop effect on cash crop yield: year 2. Practical Farmers of Iowa, Ames, IA.

Qi, Z., M.J. Helmers, R.D. Christianson, and C.H. Pederson. 2011. Nitrate-nitrogen losses through subsurface drainage under various agricultural land covers. Journal of Environmental Quality 40:1578.

Rakshit, S. 2002. Liquid swine manure as a nitrogen source for corn and soybean production. Soil Science, Iowa State Univ., Ames, Iowa.

Randall, G.W., W.E. Lueschen, S.D. Evans, and J.F. Moncrief. 1996. Tillage best management practices for corn-soybean rotations in the Minnesota River Basin, *In* U. o. Minnesota, (ed.), Vol. FO-6672. Univ. of Minnesota Extension.

Rehm, G.J., J. Lamb, M. Schmitt, G. Randall, and L. Busman. 1998. Agronomic and environmental management of phosphorus, Vol. FO-06797-B. University of Minnesota Extension.

Ritter, W.F. 1988. Reducing Impacts of Nonpoint Source Pollution from Agriculture - A Review. Journal of Environmental Science and Health Part a-Environmental Science and Engineering & Toxic and Hazardous Substance Control 23:645-667.

Ruiz Diaz, D., and J. Sawyer. 2008. Plant-available nitrogen from poultry manure as affected by time of application. Agronomy Journal 100:1318-1326.

Ruiz Diaz, D., J. Sawyer, and A. Mallarino. 2011. On-farm evaluation of poultry manure as a nitrogen source for corn. Soil Science Society of America Journal 75:729-737.

Sawyer, J., J. Pantoja, and D. Barker. 2010. Nitrogen fertilization of corn grown with cover crop. Iowa State Univ., Ames, IA.

Sawyer, J.E., and A.P. Mallarino. 2008. Using manure nutrients for crop production, *In* I. S. U. U. Extension, (ed.), Vol. PMR 1003. Iowa State Univ. Extension, Ames, IA.

Sawyer, J.E., A.P. Mallarino, R. Killorn, and S.K. Barnhart. 2002. A general guide for crop nutrient and limestone recommendations in Iowa, *In* I. S. U. U. Extension, (ed.). Iowa State Univ. Extension, Ames, IA.

Schilling, K., T. Isenhart, J. Palmer, C. Wolter, and J. Spooner. 2011. Impacts of land-cover change on suspended sediment transport in two agricultural watersheds. Journal of the American Water Resources Association 47:672-686.

Schindler, D., R. Hecky, D. Findlay, M. Stainton, B. Parker, M. Paterson, K. Beaty, M. Lyng, and S. Kasian. 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proceedings of the National Academy of Sciences of the United States of America 105:11254-11258.

Schindler, D.W. 1971. Carbon, nitrogen, and phosphorus and the eutrophication of freshwater lakes. Journal of Phycology 7:321-329.

Schroeder, P.D., and J.L. Kovar. 2008. Comparison of the phosphorus sorption characteristics of a conservation reserve buffer and an adjacent crop field. Communications in Soil Science and Plant Analysis 39:2961-2970.

Schuman, G.E., R.G. Spomer, and R.F. Piest. 1973. Phosphorus Losses from four agricultural watersheds on Missouri Valley Loess. Soil Science Society of America Journal 37:424-427.

Schwarte, K., J. Russell, J. Kovar, D. Morrical, S. Ensley, K. Yoon, N. Cornick, and Y. Cho. 2011. Grazing management effects on sediment, phosphorus, and pathogen loading of streams in cool-season grass pastures. Journal of Environmental Quality 40:1303-1313.

SCS-Iowa. 1990. Predicting rainfall erosion losses. USDA NRCS.

Sekely, A., D. Mulla, and D. Bauer. 2002. Streambank slumping and its contribution to the phosphorus and suspended sediment loads of the Blue Earth River, Minnesota. Journal of Soil and Water Conservation 57:243-250.

Sharpley, A., R. McDowell, and P. Kleinman. 2001. Phosphorus loss from land to water: integrating agricultural and environmental management. Plant and Soil 237:287-307.

Sharpley, A., P. Kleinman, R. McDowell, M. Gitau, and R. Bryant. 2002. Modeling phosphorus transport in agricultural watersheds: Processes and possibilities. Journal of Soil and Water Conservation 57:425-439.

Singer, J., C. Chase, and K. Kohler. 2010. Profitability of cropping systems featuring tillage and compost. Agronomy Journal 102:450-456.

Singer, J.W., S.D. Logsdon, and D.W. Meek. 2007. Tillage and compost effects on corn growth, nutrient accumulation, and grain yield. Agronomy Journal 99:80-87.

Singer, J.W., K.A. Kohler, M. Liebman, T.L. Richard, C.A. Cambardella, and D.D. Buhler. 2004. Tillage and compost affect yield of corn, soybean, and wheat and soil fertility. Agronomy Journal 96:531-537.

Smart, M.M., J.R. Jones, and J.L. Sebaugh. 1985. Stream Watershed relations in the Missouri Ozark Plateau Province. Journal of Environmental Quality 14:77-82.

Smith, S.J., A.N. Sharpley, W.A. Berg, J.W. Naney, and G.A. Coleman. 1992. Water-quality characteristics associated with Southern Plains grasslands. Journal of Environmental Quality 21:595-601.

Strock, J.S., P.M. Porter, and M.P. Russelle. 2004. Cover cropping to reduce nitrate loss through subsurface drainage in the northern U.S. corn belt. Journal of Environmental Quality 33:1010-1016.

Sylvan, J.B., Q. Dortch, D.M. Nelson, A.F. Maier Brown, W. Morrison, and J.W. Ammerman. 2006. phosphorus limits phytoplankton growth on the Louisiana Shelf during the period of hypoxia formation. Environmental Science & Technology 40:7548-7553.

Tabbara, H. 2003. Phosphorus Loss to Runoff Water Twenty-Four hours after application of liquid swine manure or fertilizer. Journal of Environmental Quality 32:1044-1052.

Timmons, D.R., R.E. Burwell, and R.F. Holt. 1973. Nitrogen and phosphorus losses in surface runoff from agricultural land as influenced by placement of broadcast fertilizer. Water Resources Research 9:658-667.

Udawatta, R.P., J.J. Krstansky, G.S. Henderson, and H.E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. Journal of Environmental Quality 31:1214-1225.

United States. National Agricultural Statistics Service. 2007. 2002 census of agriculture., pp. 1 online resource, Vol. 1 - 2. U.S. Dept. of Agriculture, National Agricultural Statistics Service. Washington, D.C.

United States. National Agricultural Statistics Service. 2009. 2007 census of agriculture, pp. 1 CD-ROM, Vol. 1-2. U.S. Dept. of Agriculture, National Agricultural Statistics Service, [Washington, D.C.].

USEPA. 2007. Hypoxia in the Northern Gulf of Mexico, an update by the EPA Science Advisory Board. EPA-SAB-08-003. USEPA, Washington, DC.

Webber, D., S. Mickelson, S. Ahmed, J. Russell, W. Powers, R. Schultz, and J. Kovar. 2010. Livestock grazing and vegetative filter strip buffer effects on runoff sediment, nitrate, and phosphorus losses. Journal of Soil and Water Conservation 65:34-41.

Wilson, C., R. Kuhnle, D. Bosch, J. Steiner, P. Starks, M. Tomer, and G. Wilson. 2008. Quantifying relative contributions from sediment sources in Conservation Effects Assessment Project watersheds. Journal of Soil and Water Conservation 63:523-532.

Wittry, D., and A. Mallarino. 2004. Comparison of uniform- and variable-rate phosphorus fertilization for corn-soybean rotations. Agronomy Journal 96:26-33.

Wortmann, C.S., and D.T. Walters. 2006. Phosphorus runoff during four years following composted manure application. Journal of Environmental Quality. 35:651-657.

Wortmann, C.S., and D.T. Walters. 2007. Residual effects of compost and plowing on phosphorus and sediment in runoff. Journal of Environmental Quality. 36:1521-1527

Young, R.A., and C.K. Mutchler. 1976. Pollution potential of manure spread on frozen ground. Journal of Environmental Quality 5:174-179.

Young, R.A., T. Huntrods, and W. Anderson. 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. Journal of Environmental Quality 9:483-487.

Zaimes, G., R. Schultz, and T. Isenhart. 2008a. Streambank soil and phosphorus losses under different riparian land-uses in Iowa. Journal of the American Water Resources Association 44:935-947.

Zaimes, G., R. Schultz, and T. Isenhart. 2008b. Total phosphorus concentrations and compaction in riparian areas under different riparian land-uses of Iowa. Agriculture Ecosystems & Environment 127:22-30.

Zhao, S.L., S.C. Gupta, D.R. Huggins, and J.F. Moncrief. 2001. Tillage and nutrient source effects on surface and subsurface water quality at corn planting. Journal of Environmental Quality 30:998-1008.

## References


Chase, C., K. Delate, M. Liebman, and K. Leibold. 2008. Economic Analysis of Three Iowa Rotations, pp. 12. Iowa State University: University Extension, Ames, Iowa.

Christianson, L., J. Tyndall, and M. Helmers. In Preperation. Financial Comparison of Seven Nitrate Reduction Strategies for Midwestern Agricultural Drainage.

Duffy, M. 2008. Estimated Costs for Production, Storage and Transportation of Switchgrass, *In* I. S. University, (ed.), Vol. A1-22. Iowa State University University Extension, Ames, Iowa.

Iowa State University. 2011a. Production Costs of Corn Following Beans. Iowa State University, Ames, IA.

Iowa State University. 2011b. Low-till drilled soybeans following corn. Iowa State University, Ames, Iowa.

Duffy, M. 2012. Estimated Costs of Crop Production in Iowa - 2012. Iowa State University Extension and Outreach.

Iowa State University Extension. 2011a. Ag. Decision Maker: Cash Rental Rate Estimation. Release 2011. Iowa State University Extension, Ames, IA.

Edwards, W., and A. Johanns. 2011b. Cash Rental Rates for Iowa 2011 Survey. Iowa State University: University Extension, Ames, IA.

Edwards, W., A. Johanns, and A. Chamra. 2011. 2011 Iowa Farm Custom Rate Survey, *In* I. S. University, (ed.), Vol. A3-10. Iowa State University, Ames, IA.

Kling, C., S. Rabotyagov, M. Jha, H. Feng, J. Parcel, P. Gassman, and T. Campbell. 2007. Conservation Practices in Iowa: Historical Investments, Water Quality, and Gaps. Iowa State University: Center for Agricultural and Rural Development and Department of Economics, Ames, Iowa.

Mallarino, A., B. Hill, and K. Culp. 2011. ISU soil testing and plant analysis laboratory soil-test P summaries, 2006-2010.

Mallarino, A.P., and J. Prater. 2007. Corn and soybean grain yield, phosphorus removal, and soil-test response to long-term phosphorus fertilization strategies. p. 241-253 Proc. Integrated Crop Management, Ames, IA2007. Iowa State University University Extension Agribusiness Education Program.

Miller, G.A., T.E. Fenton, B.R. Oneal, B.J. Tiffany, and C.L. Burras. 2010. Iowa Soil Properties and Interpretations Database.

Singer, J. 2011. Cover Crops: Why, How, and How Much?, *In* R. Christianson, (ed.), A copy of a presentation given by Jeremy ed.

# Section 2.4
# Other Considerations Beyond Farm-Level Costs of Nutrient Reduction Practices

Prepared by the Iowa State University Science Team
May 2013

The Iowa NPS Nutrient Reduction Science Assessment identified a set of practices to reduce nitrogen (N) and phosphorus (P) reaching surface water. The analysis included the farm level cost to implement a practice, but did not include the full economic cost or benefit of a practice or scenario. It also does not include off-farm cost and benefits related to implementing and monitoring practices. This section addresses other considerations, both positive and negative, that have not been factored into the analysis. These considerations are not fully vetted and deserve a more in-depth analysis, but the methods, results and costs/benefits are unique to the scenario being considered. Thus, this section raises questions that also should be considered when evaluating practice adoption and policy decisions. In addition, the changes described will be implemented over time rather than immediately. As a result, the cost and benefits may be moderated as markets adjust and capital replacement occurs over time.

Much like the soils and climate of the Corn Belt, the Gulf of Mexico is a natural resource important to the region and the nation. Protecting the eco-system also protects the economy based on fishing and tourism. Nutrients from the upper Mississippi basin contribute to Gulf hypoxia, which threatens the Gulf.

Closer to home, practices that reduce nutrient loss to the Gulf also help protect water quality in Iowa streams and lakes. Improved water quality can reduce water treatment costs for communities, plus increase recreational opportunities, which leads to additional recreational spending locally.

The economic analysis in the Science Assessment does not include these types of benefits. There are studies that have estimated cost savings to municipalities and households of reduced nutrients in surface water, or the economic benefit of greater recreational activity associated with cleaner water bodies. The objective of the Science Assessment was to identify and model the effectiveness of specific practices at reducing N and P from reaching the Gulf of Mexico, plus estimate the cost and cost per unit of nutrient removed when implementing each practice. It was beyond the scope of that analysis to also calculate the benefits of each practice.

The cost estimates in the analysis are based on prices and costs in 2012: $5.00/bu corn, $12.50/bu soybeans, $0.50/lb. nitrogen and $0.59/lb. phosphate. Yields, land rental rates and the cost to construct wetland, bioreactors and other structures are based on estimates for 2012. If input and output prices or costs change from these levels, so will the cost of implementing the practices. Lower grain prices will lower the cost of adopting practices that have a yield reduction. A market for biomass for energy production will make land use changes less costly. Lower fertilizer prices will lessen the incentive to reduce application rates.

The cost and cost effectiveness of practices differed widely across practices and combinations of practices. Likewise, the effectiveness and predictability of a practice may differ by weather conditions, location in the state and other management decisions.

The equal annualized cost to implement the three scenarios that meet both the N and P reduction objective ranged from near $77 million to more than $1.2 billion. The initial investment necessary to implement these three scenarios ranged from $1.2 to $4 billion. These investments will be made over many years. Kling, et al. estimated that Iowa farmers invested $435 million on seven conservation practices between 1997 and 2004[1]. Thus, the level of initial investment under the three scenarios is achievable over time.

It is important that individual farmers or localized groups of farmers, such as a watershed or drainage district, be allowed the flexibility to choose the combination of practices that will achieve water quality goals at the most effective costs. Given the best available information, farmers, alone or in groups, are able to find the lower cost and lower risk strategies more effectively than a mandate directed from the state or national level.

The cost of adopting practices to achieve targeted reductions in N and P were estimated including the farm level and, where noted, allied-industry level costs. It is important to recognize that while cost estimates for the individual farmer may be relatively straightforward to calculate, it is more difficult to estimate the economic impact if the majority or all farmers adopt the practice.

The cost estimates are based on current dollars and current technologies. As new technologies emerge and farmers find more efficient ways to implement practices, the adoption costs can be expected to decline. The investment and annual costs are estimated average costs. The costs are expected to be lower for practices installed in ideal locations, but higher than average for locations less well suited for a practice. Scenarios that assume high implementation levels may have higher-than-expected costs, as more above-average cost installations are used.

## Price impacts of supply changes

Some of the practices have an impact on corn and soybean production area or yield. The impact of changing supplies on corn and soybean prices can be large. Dr. Chad Hart, ISU Grain Marketing Economist, estimates for a one million bushel increase (or decrease) in corn supplies, corn prices tend to decrease (or increase) by $0.00136 per bushel. For soybeans, the same expected price change is $0.00625 per bushel. For every one percent change in the supply of alfalfa, there would be a corresponding 0.8 percent price change in the opposite direction.

While commodity price increases are a gain to the producer, they are a loss to the user. Based on historical relationships, a 10-cent price change in corn impacts Iowa net farm income by $110 million in the same direction. Given a 2.3 billion bushel corn crop, gross income to corn producers of a dime per bushel increase would be expected to increase $230 million. Thus, net farm income does not change at the same rate as grain prices. Furthermore, income of businesses beyond the farm gate impacted by higher corn prices, specifically ethanol returns, are not included as part of net farm income.

## Cover crops, wetlands and bioreactors

Cover crop seed production is another cost that must be counted differently if widespread adoption is expected. The USDA reported the United States planted 1.3 million acres of rye in 2011 with only 242,000 acres harvested. To seed 60% of Iowa's 23.4 million acres of corn and soybeans in 2012 at seeding rates of one bushel per acre with a seed harvest of approximately 45 bushels per acre would require 312,000 acres (1.3% of Iowa's rowcrop acreage) acres of rye for seed production, more than was

[1] *Catherine Kling, Sergey Rabotyagov, Manoj Jha, Hongli Feng, Josh Parcel, Philip Gassman, Todd Campbell,* Conservation Practices in Iowa: Historical Investments, Water Quality, and Gaps. *Center for Agricultural and Rural Development and Department of Economics, Iowa State University.* July 22, 2007

harvested in the United States in 2011. To raise this much seed in Iowa reduces corn and soybean production, but increases sales of rye seed or reduces cost for rye seed purchased by saving seed. Cover crops also impact corn production due to an estimated 6% reduction in corn yields following rye cover crops. One of the combination scenarios in the Science Assessment uses cover crops on 60% of the 21 million continuous corn and corn-soybean acres. Assuming 170-bushel corn yield, production would be reduced by 77.1 million bushels.

Widespread use of bioreactors will require trees be planted to provide the woodchips. It is estimated 111,000 acres (0.5% of Iowa's rowcrop acreage) of trees would be needed to supply chips for bioreactors if used at the maximum level.

Wetlands are estimated to have a 10-acre pool and 35-acre buffer per 1,000 acres of cropland treated. To treat all 10.261 million acres possible would require 462,000 acres (2% of Iowa's rowcrop acreage) of wetlands and buffer.

Even if it is assumed the wetlands, rye seed production and wood chips come from low productivity land, the total impact on production is large. These three practices, if adopted on the maximum acres possible, would take approximately 885,000 acres (3.8% of Iowa's rowcrop acreage) out of corn and soybean production. The expected long-term price impact, including reduced yield on cover crop acres, would be approximately $0.20 (4%) per bushel on corn and approximately $0.09 (0.7%) on soybeans.

Based on these changes in yield and price, farm income from corn and soybean production would decrease slightly (the increased price does not offset the reduced production) before accounting for the losses to the grain user sector. The production of rye, wood chips and wetlands do generate potential income or cost savings. However, if other states also adopted these practices, the price impacts would be larger as more acres are impacted, leading to decreased crop production. If other states do not adopt these practices, the higher prices would encourage production in those states, partially offsetting the price increase for Iowa grain farmers but increasing net farm income in those states. Grain users, meat, milk, egg and ethanol producers and export customers would be negatively impacted by higher grain prices. Moving corn and soybean production out of Iowa to other regions, particularly those not well suited for row crop production, could generate negative environmental impacts in those regions. The added wetlands, trees and rye seed production increases landscape diversity within Iowa.

### Fall to Spring N application

Another example of a practice that has costs beyond the farm level is shifting from fall application of N to spring application. Dr. Dan Otto, ISU Extension Economist, estimated the annualized infrastructure cost (storage, handling and application equipment) to shift all fall fertilizer application from fall to spring at $397.34 million.

It is assumed 25% of the nitrogen is applied in the fall. Twenty-five percent of the estimated state average application of 171lbsN/acre means 43lbsN/acre is applied in the fall. However, the recommended maximum return to nitrogen (MRTN) is 156lbsN/acre. Reducing N application rates to the MRTN level means it is not necessary to build the entire additional infrastructure Otto assumed would be needed, thus lowering the needed investment.

The industry currently applies an estimated 128lbsN/acre in the spring. The difference between the 156lbsN/acre capacity and the current 128lbsN/acre is 28lbsN/acre. This is 65% of the 43lbsN/acre capacity that Otto recommended building. Otto's estimate was $397.34 million annually for the added capacity, but only 75% of that was for nitrogen, or $297.75 million. At 65% of that capacity is $194 million annually for infrastructure costs that would need to be added to move to spring-only application.

Moving application of liquid swine manure from fall to spring creates added costs for pork producers and commercial manure applicators. Most manure storage is built to hold a year's supply or more of manure. Shifting from fall to spring will cause logistical problems in the transition year because there is typically not enough storage to forgo fall pump out and additional land will be required to empty storage in the spring after manure had been applied to the fields in the fall. The application time window is narrower in the spring than the fall. It will require additional equipment and labor to apply the same amount of manure in fewer days and thus increase the cost of manure application.

An additional consideration in changing from fall to spring fertilizer application is timeliness of farming operations. If fertilization is moved to a spring application without changing spring operations, there will be less time available for planting the crop. Conversely, if tillage operations change, there may be more time available. The two main factors to consider when evaluating the impact of changing field operations are the number of days suitable for fieldwork and the time it takes for each operation performed. The time it takes per operation and to a lesser extent, the days available, will be influenced by the power unit and the size of the implement.

Corn and soybean yields have an optimum planting date. In the Iowa latitudes, May 10 is the critical planting date for corn. After that date, yields begin to decline. Field trials by Iowa State University have documented this pattern. Planting delayed two weeks results in a 10% reduction in yield and a delay of four weeks could lead to a 25% yield reduction.

The National Agricultural Statistics Service provides a weekly estimate of the days suitable for fieldwork. Iowa State University Extension compiled these estimates from 1958 through 2007. For Iowa from April 2 through May 13, there was a median of 20.6 days suitable for fieldwork. Obviously the days suitable for fieldwork and the first day when fieldwork is possible will vary by year and region of the state. However, having an estimate of the median number of days is necessary to estimate the timeliness cost of changing operations or the timing of the operations.

The second component for calculating potential timeliness yield loss is estimating the amount of time for all of the operations performed. ISU Extension publication AgDM A3 -24, *Estimating the Field Capacity of Farm Machines*, provides an estimate of the time for a variety of operations and sizes of implement.

As an example, assume a 1,500-acre farm using 12 hours per day following a disk/cultivate tillage regime. A 33-foot tandem disk is estimated to cover 19.2 acres in an hour. That means a farmer could cover 230 acres in a day, so it would take 6.5 days to tandem disk (1500/230). A 50-foot field cultivator can cover 33.9 acres an hour or 407 acres per 12-hour day. With 1,500 acres it would take 3.7 days. A 24-row, 30-inch planter covers 21.8 acres an hour or 262 acres in a 12-hour day. Planting would add another 5.7 days for a 1,500-acre farm. Finally, a 17-knife anhydrous applicator would cover 16.2 acres an hour or 194 acres a day. This means for a 1,500-acre farm with large equipment and using a disk/cultivator tillage system, it would take 6.5 + 3.7 + 7.7 + 5.7 = 23.6 days.

The number of days for fieldwork in this hypothetical example would exceed the median number of days available, assuming the goal was to be planted by May 10. A farmer would suffer yield loss if all the operations had to be performed in the spring.

The fieldwork estimate does not include maintenance or travel. Therefore, a 12-hour day is appropriate for the examples. The total number of days needed for fieldwork to avoid planting delays depends on the size of the equipment, the number and type of operation, and days available. The losses could be serious in some situations. With $5 corn and a 1.5-bushel per day yield loss, a farmer with 1,500 acres of corn would lose $11,250 for every day of delay. In the example above, planting would be at least three days beyond May 10. Therefore, this hypothetical farmer would have a $33,750 loss due to delayed planting. Applying the yield loss to the 25% of the acres that would shift from fall to spring fertilizer application is predicted to reduce total corn production by approximately 16 million bushels, and the price would be expected to increase approximately $0.02/bushel.

## Extended rotations

Moving acres from continuous corn or corn-soybean rotation to a corn-soybean-alfalfa-alfalfa-alfalfa rotation reduces N application and corn and soybean production while increasing hay supplies. Increasing supply would lead to lower prices. Acreage of alfalfa in Iowa has decreased from 1.9 million acres in 1989 to 820,000 acres in 2011 and annual production dropped from 5.7 million tons to 2.8 million tons. Prices increased from $84 a ton to $134 a ton over the same time period. The resulting elasticity is -0.8. This means for every one percent change in the supply of alfalfa, there is a corresponding 0.8 percent change in price in the opposite direction. A scenario that doubles the acres in an extended rotation would increase the supply of alfalfa 100% but cut the price by 80%. It would reduce the supply of corn and soybeans resulting in higher prices for these commodities.

A scenario that implements an extended rotation on 25% of the acres reduces corn and soybeans 1.89 and 1.26 million acres, respectively, and increases alfalfa by 3.15 million acres. Prices are estimated to increase $0.40-0.45/bushel for corn and $0.35-0.40/bushel for soybeans. Alfalfa acres nearly triple and prices are expected to decline by 230 percent unless new demand from beef or dairy cattle, sheep or horses emerges. The corn and soybean prices do not increase enough to offset the lost acres and the decrease in alfalfa price outweighs the increase in alfalfa supplies. Gross income to crop farmers selling these three commodities is expected to decline. And while dairy and beef cow producers benefit because of lower-priced alfalfa, beef feedlots, hog and poultry producers are negatively impacted by higher corn and soybean prices. The price changes also dramatically change the economics of the practice, as such market forces will impact how quickly and how far adoption of extended rotations will proceed.

## Non-economic costs and benefits

In addition to economic factors beyond the scope of the Science Assessment, the nitrogen and phosphorous reports identify additional implications, both positive and negative, from implementing the nutrient reduction practices. A few of these are repeated here:

**Possible benefits**

- Planting cover crops decreases erosion and loss of surface runoff contaminants, increases wildlife habitat and organic matter in soil. It also is possible to harvest forage from cover crops, increasing forage supplies on the farm.
- Increased organic matter in soils improves soil structure and supports increased soil fertility, soil water holding capacity and drought resistance, plus resists erosion and compaction.
- Wetlands can increase the aesthetics of the landscape, increase habitat for Iowa game and waterfowl, and depending on design, could provide hydrologic services through water flow reduction to mitigate downstream flooding.
- Practices that reduce P movement also limit soil erosion and sediment from reaching water bodies.
- Increased use of forages in extended rotations or strategically targeted perennials will increase wildlife habitat and biodiversity and decrease soil erosion, surface runoff, and surface runoff transported P export. It also may support the growth of the beef and dairy industries, and diversify the ecosystem and the economy.

- Practices requiring more equipment or management create job opportunities and expand or develop new industries in the state. For example, more soil sampling and testing, variable-rate technology, installation of bioreactors, terraces, drainage control, vegetative buffers, storage and transport of manure and other emerging technologies would lead to more jobs and more economic development.

**Possible costs**

- Applying liquid swine manure in the spring increases concerns of soil compaction, increases risk of runoff shortly following manure application, and increases risk of rapid movement to tile lines due to frequent wet soil conditions in the spring.
- Reducing nitrogen application rates too much leads to reducing total nitrogen and soil organic matter, thus lowering soil quality over the long term. That also leads to the risk of inadequate nitrogen for corn in high-nitrogen responsive seasons.
- Bioreactors have the concern that in over-designed systems, the denitrifying bacteria can produce methylmercury, which is highly toxic and can bioaccumulate in fish.
- Using controlled drainage to manage the water table at a shallower depth could result in increased surface runoff, which would have implications for soil erosion and transport of other surface runoff contaminants (e.g. phosphorus).
- Monoammonium phosphate (MAP) and diammonium phosphate (DAP) are typically fall applied when it is logistically easy and an effective time for P application. However, the N in the fall-applied MAP and DAP is at a high risk of leaching.
- The practice of reducing soil test P to optimum is positive for P loss and for the economics of crop production for those who don't apply manure. However, from the perspective of the best utilization of Iowa resources, using the P Index and letting soil-test P increase until the P Index is at the upper boundary of the optimum level would allow farmers to utilize the manure N resource without the cost of moving manure to more distant fields.

## Conclusions

Estimating the costs of a change in practice to an individual farmer is a relatively straightforward process. But when enough farmers make a change that impacts the supply and demand, a different set of estimation problems arise. The whole nature of the estimation process changes when a change in practice involves changes beyond the farm gate. Winners and losers must be considered as well as the unintended consequences of the actions.

The Iowa NPS Nutrient Reduction Science Assessment examined alternative scenarios to reduce N and P runoff. Costs to the individual farmer were estimated in the discussion of the scenarios. However, costs beyond the farm gate were not considered. Adoption of the practices is expected to occur over many years. As such, market prices will adjust to changes in supply and demand resulting from practice adoption. Existing crop and livestock sectors will adjust and new markets (cellulosic biofuels) may emerge. The level of initial investment shown in the three scenarios is within range of earlier conservation investments and is possible over an extended time frame.

Not including these costs does not diminish their importance. Their exclusion simply recognizes estimation of these costs is not the central focus of this effort. If one or more of the scenarios is deemed worthy of further consideration, these macro costs may be included.

# Section 2.5
# Nonpoint Source Science Assessment Team Members

## Iowa State University College of Agriculture and Life Sciences
**John Lawrence**, **Science Assessment Team Chair**, Associate Dean, Extension Programs and Outreach
**Matt Helmers**, **Nitrogen Team Chair,** Agricultural and Biosystems Engineering
**Tom Isenhart**, **Phosphorus Team Chair,** Natural Resource Ecology and Management
**Jim Baker**, Agricultural and Biosystems Engineering
**Mike Castellano**, Agronomy
**Reid Christianson**, Agricultural and Biosystems Engineering
**Bill Crumpton**, Ecology, Evolution and Organismal Biology
**Rick Cruse**, Agronomy
**Mike Duffy**, Economics
**Phil Gassman**, Center for Agricultural and Rural Development
**Antonio Mallarino,** Agronomy
**John Sawyer**, Agronomy
**Dave Webber**, Agricultural and Biosystems Engineering

## Iowa Department of Agriculture and Land Stewardship
**Dean Lemke**
**Shawn Richmond**

## Iowa Department of Natural Resources
**Keith Schilling**
**Calvin Wolter**

## USDA Agricultural Research Service
**David James**
**Dan Jaynes**
**John Kovar**
**Mark Tomer**

## U.S. Environmental Protection Agency
**Katie Flahive**

## USDA Natural Resources Conservation Service
**Eric Hurley**

**Susan Thompson**, Strategy and Science Assessment Editor,
Iowa State University College of Agriculture and Life Sciences