Science Assessment of Cropland Practices for Minnesota's Nutrient Reduction Strategy: Part 1 Nitrogen

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Chapter 1: Introduction

Objective and Goals

The specific, assessable objective of this work was to develop nitrogen (N) loss reduction efficiencies appropriate for Minnesota for a variety of conservation practices. This was intended to support efforts related to the 2025 revision of the Minnesota Nutrient Reduction Strategy. Some phosphorus (P) loss benefits and tradeoffs due to these practices were also considered.

The recommended reduction efficiencies needed to be developed in a transparent way, with fully described methods so the results were reproduceable. The data selection criteria needed to be justifiable to produce defensible and realistic recommended efficiencies. It was also important to include a measure of variability (i.e., standard deviation) for the recommended values to plainly reflect scientific uncertainty in the performance of these conservation practices.

The broader goal of this work is to increase knowledge of how conservation practices perform in Minnesota, which is intended to **help accelerate adoption of these practices**. The ultimate aim of these efforts was to create positive action toward **cleaner water in Minnesota and downstream.**

Methods

Several excellent conservation practice reviews have been performed for Minnesota (Kjaersgaard and Pease, 2021; Lenhart et al., 2017; Mulla and Whetter, 2020), but this assessment had the specific goal of assigning an appropriate numeric efficiency for each practice. This quantitative review involved:

1. Sourcing and reviewing studies that contained direct measures of nutrient loss from a conservation practice and an appropriate control,
2. Extracting those paired data into a database, and
3. Analyzing the resulting extracted data and recommending an overall nutrient removal efficiency for each practice.

The approach here aimed to be consistent with the processes used for science assessments in other Hypoxia Task Force states (IDALS, 2024; IDOA, 2015; IL EPA, 2023).

Nearly 800 studies were identified over the course of this review, although some were not ingested due to the timeframe (e.g., "*have but not reviewed*"; Figure 1a). Most of the data sources were reviewed between August 2023 and February 2024 (Figure 1b). Initially studies were identified through online keyword searches and via colleagues. For example, Dr. Fabian Fernandez (UMN) shared a variety of articles and theses, and Dr. Jane Frankenberger (Purdue University) shared the reference lists used to inform Indiana's science assessment. The preliminary science assessment performed by faculty at the University of Minnesota (shared by Brad Carlson) helped identify a number of non-peer reviewed Minnesota studies that would have otherwise been overlooked.

The review built upon itself as articles were identified by tracing back references that were cited as containing nutrient data. The Journal of Environmental Quality provided the most studies (e.g., Figure 6) so, in an attempt to capture the newest studies which wouldn't have been cited yet, the table of contents of the most recent Journal of Environmental Quality issues were reviewed for possible studies. Google Scholar profiles of several prolific Minnesotan researchers were also combed for possible relevant studies (e.g., J. Baker, G. Feyereisen, J. Moncrief, D. Mulla, G. Randall, C. Rosen, M. Russelle).

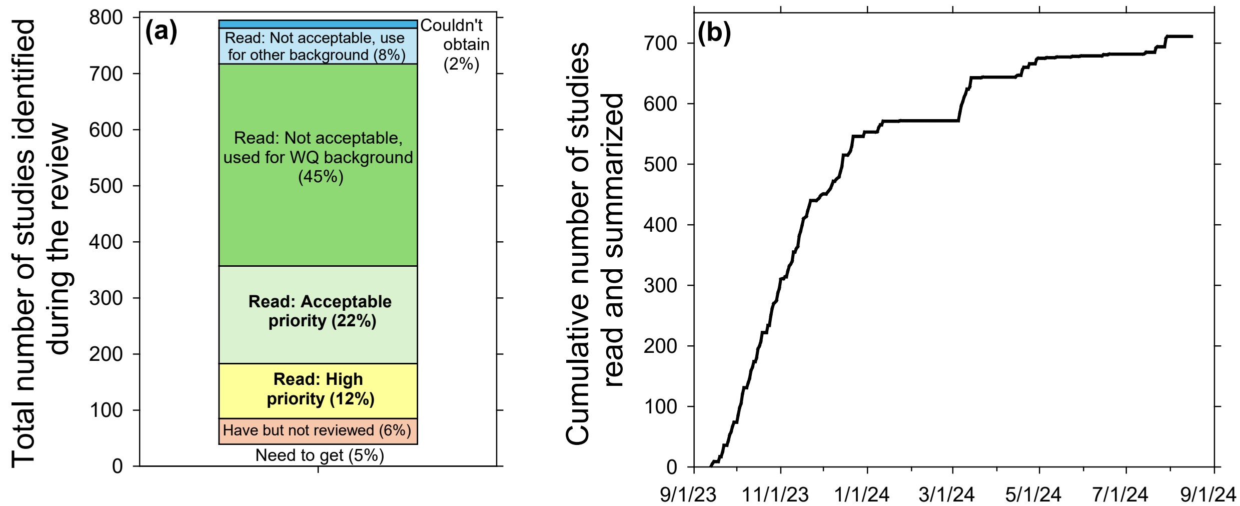

Figure 1. Description of the 796 studies identified in this analysis (a) and the timeline of their review (b). Each information source was tracked by its stage in the review process (e.g., "Need to get" versus having been read). "High priority" and "Acceptable priority" related to the clarity of data reporting and ease of data extraction.

The nearly 800 studies sourced included practices assessed in the four chapters here as well as several types of practices that were not ultimately evaluated (Figure 2a vs. b). Working from the original list of nearly 70 practices provided by the MPCA, a number of studies were sourced and reviewed (Figure 2b) before it was determined this report would only focus on nitrogen. These un-assessed practices were more P- than N-focused (i.e., P management; Reduced tillage; In-field and in-channel sediment practices; Manure and grazing practices; Irrigation practices). These practices are important for the state of Minnesota, and thus, should be reviewed for nutrient strategy efforts at some point.

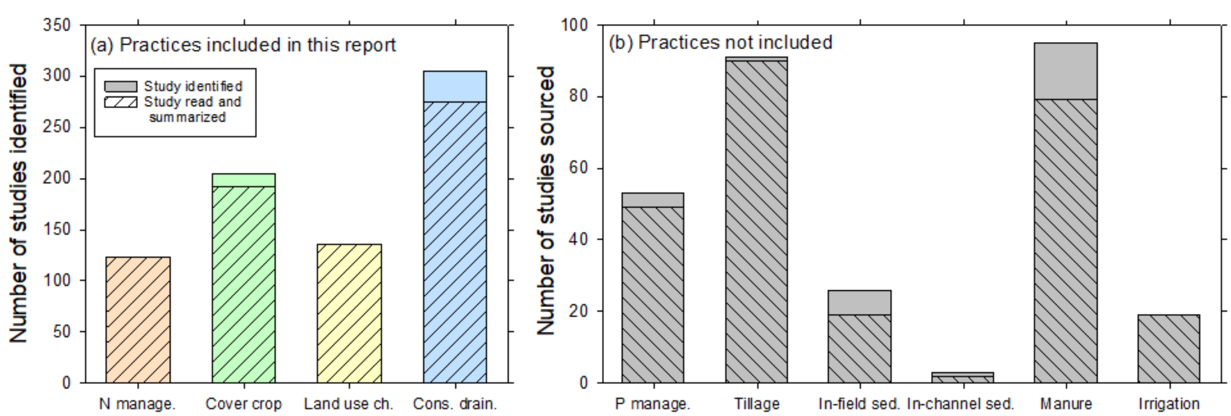

Figure 2. Number of studies identified and reviewed that were included in this report (a) and not included (b).

Approximately 710 of the total 800 studies were read and summarized. In the main body of this report, the unread conservation drainage studies mostly pertained to the practice of wetlands, which was not assessed here (Figure 2a, top part of blue bar). However, this practice was later comprehensively evaluated and detailed in supplementary information, which summarizes results from three studies (109 site-years) on constructed wetlands. The unread cover crop studies were mainly studies from Europe. Several studies evaluated more than one practice and were thus classified under multiple categories. For example, a University of Minnesota thesis by Wayment (2021) studied N leaching under a Kura clover living mulch, a cereal rye cover crop, and varying N fertilizer rates.

Approximately 270 studies contained extractable water quality data (34% of total studies, Figure 1a). These were internally categorized as "*High priority*" or "*Acceptable priority*". The former indicated the study had clear direct measures of N and/or P losses from both a control and a treatment. The latter determination indicated that nutrient losses were reported but perhaps the methods deviated from the ideal criteria (e.g., use of suction lysimeters) or the way the data were reported complicated data extraction (e.g., only one multi-year average was reported). When specific values were not reported in the text or a table, data were extracted from figures using a plot digitizer (https://plotdigitizer.com/). This was done for approximately 50 figures from 30 unique studies.

This quantitative review was not intended to be a rigorous statistical analysis, nor did it follow meta-analytical methods. This approach was developed to align as closely as possible with direct measures of water quality as reported in original studies to maintain transparency and simplicity. There was often a lack of significant differences between treatments in the reviewed studies, but the data were nevertheless extracted. **The aim of this work was not to demonstrate statistical significance, but rather to quantify tangible differences in nutrient loss due to a practice, if those differences existed.**

Data extraction criteria

Four major criteria were used for the extracted data.

1. Field or plot studies, not modeling studies

This selection criterion was to ensure the resulting values aligned as closely as possible with direct measurements of nutrient loss observed in the field. Modelling studies were reviewed (e.g., RZWQM: Malone et al., 2014; Malone et al., 2023; SWAT: Her et al., 2017; Wilson et al., 2014; DRAINMOD-NII: Luo et al., 2010; Wilson et al., 2020; EPIC: Rejesus and Hornbaker, 1999), but with the primary intent of sourcing original monitoring data that had not been reported elsewhere. Exclusion of modeling results was consistent with other state's science assessment processes.

2. Studies with a control versus a conservation practice treatment

A clearly defined and consistent control as well as a clearly defined and consistent treatment were necessary to assess the effectiveness of a given practice across the available studies. Both the control and the treatment needed to be present in the same location and same year to allow calculation of the loss reduction for that specific paired site-year (Equation 1). Equation 2 shows an example of how the nutrient loss reduction efficiency is calculated.

$$Equation\ 1.\ Percent\ reduction\ (\%) = \frac{(control\ loss - treatment\ loss)}{control\ loss} \times 100$$

Equation 2. Example for a 20% N loss reduction

$$20\%\ N\ loss\ reduction = \frac{\left(no\ cover\ crop\ loss\ of\ 25\ {}^{kg\ N}/_{ha} - cover\ crop\ loss\ of\ 20\ {}^{kg\ N}/_{ha}\right)}{no\ cover\ crop\ loss\ of\ 25\ {}^{kg\ N}/_{ha}} \times 100$$

A "*paired site-year*" consisted of the annual N loss from the control and the annual N loss from the treatment in that given year (Figure 3). Individual annual losses were extracted from the studies rather than the overall multi-year means that studies often report. That is, in the example shown in Figure 3, the overall study mean of 16% might have been used in discussion but was not extracted into the database. Compiling individual site-years rather than study means increased the sample sizes for these analyses (i.e., increased the power), and better facilitated a reflection of variability across all the study years. In cases where one multi-year mean was reported, that value was duplicated in the database for the given number of years.

Most generally the control was a 2-y corn and soybean rotation (Figure 3). Applying this 2-y rotation effect to the standard equation resulted in Equation 3. However, use of a 2-y corn and soybean rotation was not possible for all the practices depending upon data availability. Thus, some practices simply used a "*corn grain*" control which may have been a mix of corn phase site-years and continuous corn site-years. The controls are transparently listed for each practice in an accompanying table of each section.

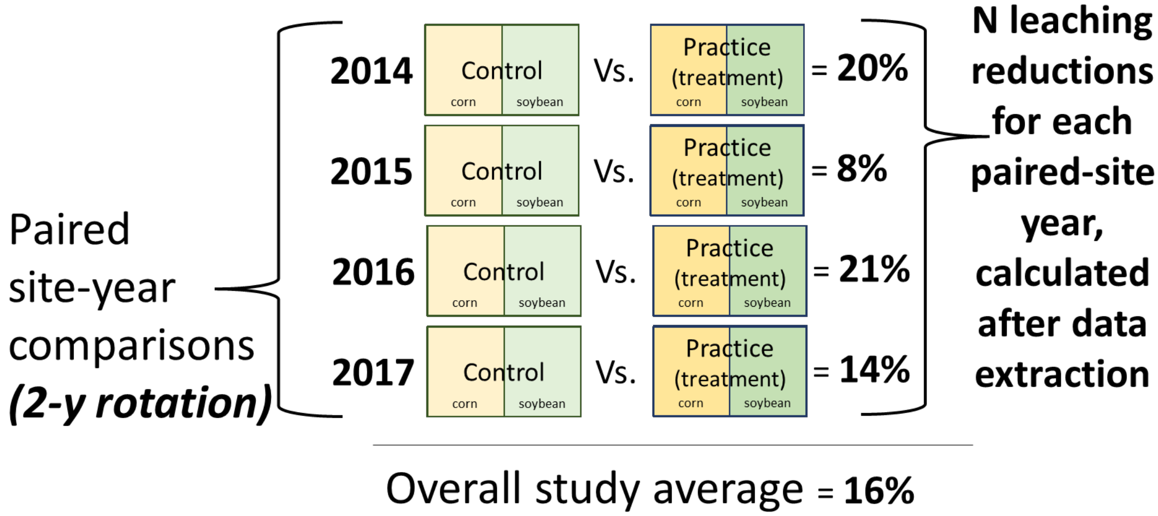

Figure 3. Example of data extraction for a corn-soybean control. Each calculated "paired site-year" N leaching reduction in the database depended on the extraction of four N loss values (corn and soybean in a given year for the control and corn and soybean in the given year for the treatment). Note the individual years were extracted from studies rather than the overall study mean (16% in this example).

$$Equation\ 3.\ Percent\ reduction\ (\%) = \frac{(2\ year\ average\ CS\ loss\ from\ the\ control - 2\ year\ average\ CS\ loss\ with\ the\ treatment)}{2\ year\ average\ CS\ loss\ from\ the\ control} \times 100$$

The practice of intermediate wheatgrass was a good example of an inconsistently defined control. Half of the extracted water quality site-years used corn (or corn-soybean) as a control whereas the other half used annual wheat (Table 30). The control used in each study was appropriate for the purposes of that given study, but this created challenges for this large-scale quantitative review.

Determination of an appropriate control also varied for the edge-of-field practice types (saturated buffers and bioreactors). In those cases, the upstream nutrient loss from the field was used as the control. These studies satisfy the criterion because the control and treatment were both present in the same location and year but were simply assessed upstream versus downstream of the practice.

Specifically defining each conservation practice treatment was just as important as defining the control. Several studies assessed the use of a variety of conservation practices performed together in a field or small watershed. These types of studies produced confounded effects where the impact of one specific conservation practice "*treatment*" could not be identified. Another example of the importance of a consistently defined treatment was for the combined practice of "*N timing with a rate reduction*" which was a practice that was included in the original Minnesota Nutrient Reduction Strategy. Across the field studies reviewed, the application timings and magnitudes of rate reductions were not consistent enough to quantify a single representative impact for this practice (Table 11).

The definition of a "*treatment*" was also nuanced in the assessment of the practice of drainage water recycling. Several studies recycled drainage water to a practice treatment termed "*controlled drainage-subirrigation*" (compared to a free drainage control) whereas others used a reservoir nutrient balance approach. The limitations and rationale for that approach were fully described in section "*Drainage water recycling*".

3. Studies reporting annual nutrient loss values

This criterion entails two important components: 1) that values used in this assessment be at the *annual* time-step and 2) that nutrient *losses*, not just concentrations, are the metric of interest. The analysis of the N fertilizer management practices was an important exception to the second part of this criterion because annual flow-weighted N concentrations were used rather than annual N losses (discussed below).

Firstly, annual assessments of nutrient removal performance are the convention for nutrient strategy efforts because annual studies provide a comprehensive view of how practices perform over all seasons. This conveniently aligns with the concept of a cropping year and common agricultural management operations. It's also useful to note the size of the Gulf of Mexico hypoxic zone is assessed annually. Although it is the May-June nutrient loadings in the Mississippi River that drive the size of the hypoxic zone (Scavia et al., 2003), reporting of annual nutrient losses is most common across conservation practice studies.

A number of rainfall simulation studies were reviewed (e.g., Grabber and Jokela, 2013; Kaspar et al., 2001; Kleinman et al., 2005; Kovar et al., 2011), and while these studies provided important insight into the event-based performance of conservation practices, they were only appropriate to use for supporting information in the text rather than in the analysis. Extrapolating data from shorter-term studies would have required assumptions that were deemed not justifiable. For example, consider the hypothetical case where a conservation practice was monitored only during the growing season, and those data resulted in a N removal efficiency of 73% for that season (Table 1). Assuming the control and treatment would contribute the same additional load for the remainder of the year results in quite different efficiencies from the original 73% (i.e., Table 1 Scenarios 1 and 2: 55 and 37%). It is also possible that a given practice may continue to provide a benefit for the remainder of the year, but this would also result in a different annual removal efficiency versus the growing season (Scenario 3: 60%).

Table 1. Example of how different N removal efficiencies can result due to assumptions made to try to account for an unmonitored portion of a year. In this example, hypothetical monitoring was done for the growing season only.

	Reported N loss over the monitored growing season	Scenario #1		Scenario #2		Scenario #3	
		Assuming both have the same N loss for the rest of the year	Hypothetical *annual* N loss	Assuming both have the same N loss for the rest of the year	Hypothetical *annual* N loss	Assuming the practice continues to provide a treatment benefit over the rest of the year	Hypothetical *annual* N loss
No practice (control)	15	+ 5 kg N/ha	20	+ 15 kg N/ha	30	+ 15 kg N/ha	30
Practice (treatment)	4	+ 5 kg N/ha	9	+ 15 kg N/ha	19	+ 8 kg N/ha	12
N reduction efficiency (%)	73%		55%		37%		60%

The need for annual nutrient loss data can be an issue given monitoring equipment is often winterized in cold climates. This could mean that monitoring of freeze-thaw cycles in late winter could be missed. Nevertheless, an attempt was made to use studies that reported values as annual given the caveat that experimental designs and conditions were beyond the control of this review. The lack of late winter/early spring and snowmelt monitoring is an especially relevant gap in this Minnesota-focused review.

The second part of this criterion is that nutrient losses needed to be assessed. That said, both nutrient losses and annual concentrations (and, annual loads, in some cases) were extracted into the conservation practice databases to allow deeper assessment of practice performance.

A nutrient loss is the area-normalized product of the concentration of a nutrient in water and the volume of that runoff or drainage. Conservation practices work by reducing the concentration, the volume, or both of those factors (Christianson et al., 2016). For example, reducing the N application rate reduces the concentration of nitrate in drainage water but does not change the drainage discharge volume. Conversely, the practice of controlled drainage reduces the discharge volume but does not change the nitrate concentration in that water. Thus, in a comparison of numerous practices like this, it is necessary to assess the impact on nutrient loss for each practice because the practices work differently.

The N fertilizer management practices deviated from this criterion. The impacts of these practices on water quality tended to be small in relative terms (e.g., 4-10%, Table 4) and the assessment of annual flow-weighted N concentrations rather than annual N loss reduced noise in the data. These practices wouldn't be expected to substantially impact drainage discharge volume, so this was considered an acceptable approach (see additional details in Chapter 3).

Suction lysimeters

A number of studies in this review assessed N leaching using a common method called suction cup lysimeters, and some of these provided important Minnesota-specific results (Jungers et al., 2019; Kuehner et al., 2020; Wayment, 2021). This method involves installing a porous cup at the end of a PVC pipe at a specified depth in the soil and then applying suction to draw soil pore water for collection and analysis. There are many types of lysimeters for assessing nutrient leaching (e.g., ion exchange or resin lysimeters, large drainage lysimeters, pan lysimeters; Singh et al., 2018), but suction lysimeter studies have generally been excluded from the science assessments in other states for several reasons.

The most important reason for excluding these studies is that suction lysimeters only allow direct assessment of nutrient concentrations. Numerical modeling, often water balance modeling, is used to calculate the leachate volume to estimate nutrient loss (Singh et al., 2018). Thus, nutrient losses reported in suction lysimeter studies are not direct measures. Modeling results are not used in these science assessments because nutrient losses calculated from direct measurements of both concentrations and discharge volumes are the convention.

Another consideration is that suction lysimeters provide localized point estimates of soil water nutrient concentrations. Kuehner et al. (2020) cautioned that nitrate concentrations from suction lysimeters a few feet apart could notably differ. Replicated tile drainage plots also have limitations (e.g., discharge may differ across "*identical*" plots), but this method provides a benefit in that drainage nutrient concentrations represent an integrated composite across the plot area. Using "*large numbers*" of suction cup samplers can overcome this limitation (Singh et al., 2018) but often only one or two lysimeters are deployed per plot (albeit plots are generally also replicated).

Use of suction lysimeters may present several additional issues such as appropriate calculation of ET for a water balance when a crop coefficient may not yet be fully developed for a newer cropping practice or the possibility of the suction actually altering the soil pore hydrology. Nevertheless, suction lysimeters may be the only way to (easily) assess leaching in sandy soils where replicated drainage plots wouldn't be practical. Further, studies from Minnesota's central outwash sands region where nitrate leaching to groundwater is extremely important needed to be represented in this Minnesota-focused review. Thus, N losses reported in suction lysimeter studies were included here (e.g., Wayment, 2021) albeit with some wariness about this mixing of methods. All suction lysimeter studies needed to have the ceramic cup placed at least 90 cm deep to represent leaching below the root zone.

The broad review here included studies that used nearly all the various types of lysimeters (suction, wick, pan, resin, zero and equilibrium tension, drainage monolith). Replicated tile drainage plots were the most common method used to assess N leaching, but the prevalence and variety of alternative methods raised questions about the equivalence of reported N loss values as well as resulting practice efficiencies (%) calculated using different methods. A comparison of N leaching methods across soil types would be useful for this science assessment and others.

4. Studies performed in Minnesota or similar climatic and cropping conditions

Study location might seem the most obvious selection criterion for this Minnesota-focused review, but it was often one of the last filters applied because of the relatively few studies overall that reported direct measures of annual nutrient loss. Lenhart et al. (2017) similarly noted in the Ag BMP Handbook there was simply an insufficient number of studies performed in Minnesota alone for a thorough review of conservation practices. The science assessments for each of the Midwestern states tend to use a similar set of studies due to these data limitations. In the current effort, it was generally preferable to use an annual runoff study from, for example, Missouri, rather than conclude there were insufficient data to develop a removal efficiency for a practice. Thus, the locations of the reviewed studies spanned the US and globe (Figure 4).

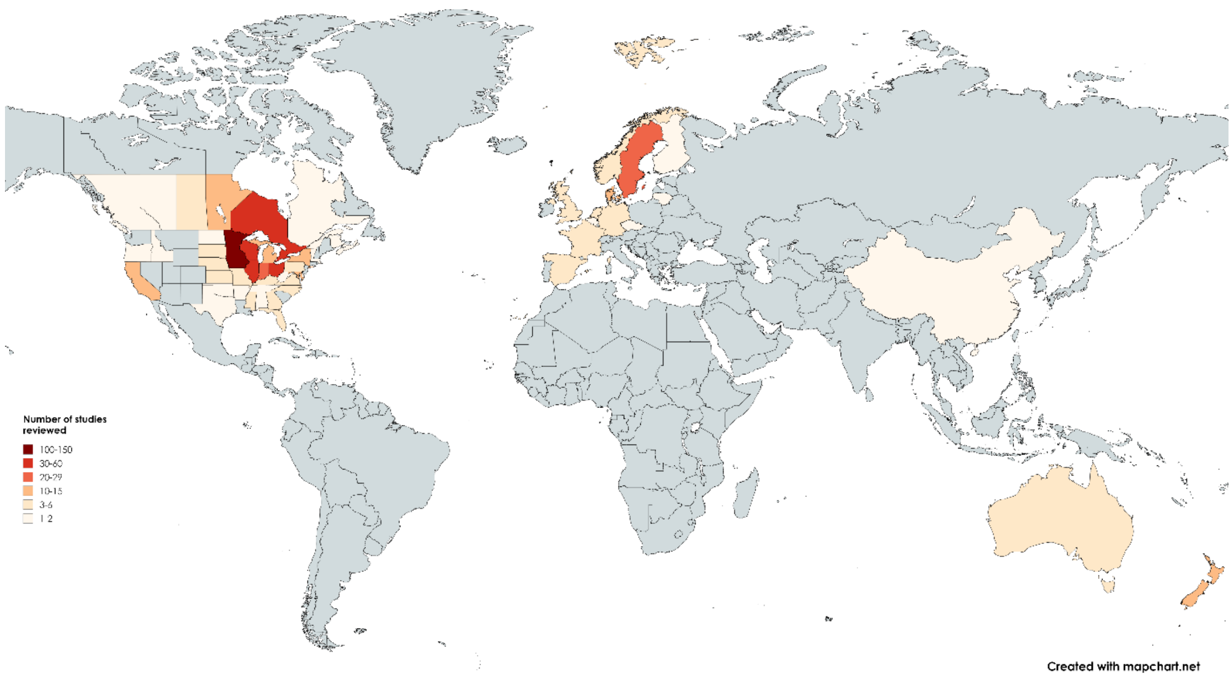

Figure 4. Map illustrating locations of the more than 700 reviewed studies. Not including 31 studies that were considered "Midwestern" (including more than one state) and 54 studies that were general reviews.

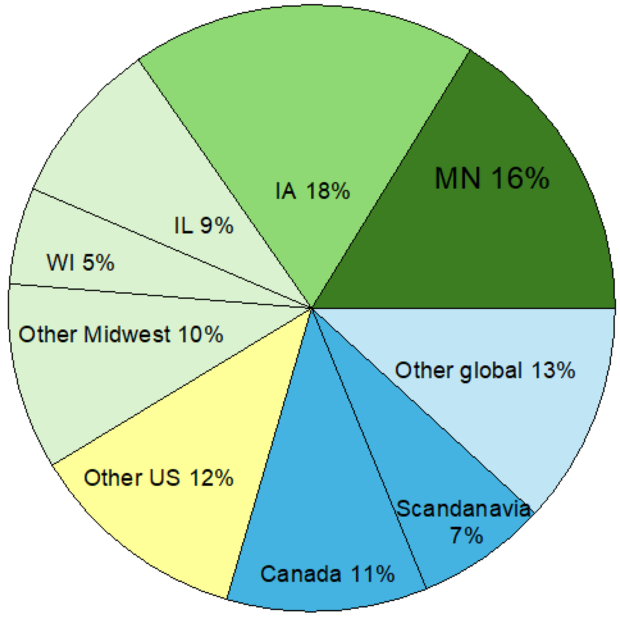

Figure 5. Distribution of the locations of the 270 studies that contained extracted water quality nutrient data (across all practice types, all nutrient forms and pathways).

The cold climate focus of this review meant studies from the Canadian prairies and Scandinavia, where snowmelt hydrology and nutrient dynamics may be similar to Minnesota, were very relevant (Figure 5). The lack of conservation practices studies performed in cold climates that include snowmelt periods is a well-known data gap (Mulla and Whetter, 2020). This is especially important for dissolved nutrients (especially P) in snowmelt runoff, which is not as erosive as rainfall runoff (Cade-Menun et al., 2013; Ginting et al., 2000; Mulla and Whetter, 2020; Ulén, 2003). Short growing seasons (for both cash crops and conservation covers) and limited infiltration during significant periods of the year when soils are frozen can lead to strong seasonality in nutrient loss forms and pathways (Kjaersgaard and Pease, 2021; Liu et al., 2019).

The focus of this report was the portion of Minnesota that falls within the Mississippi River Basin, consistent with the scope of the Hypoxia Task Force which is the driver of the nutrient strategies. There were insufficient data to make separate recommendations specifically for the Red River Valley but many of the data gaps equally applied to that portion of Minnesota.

Additional preferred review criteria

Studies that had undergone peer-review were preferred. This was consistent with the process for updating recommended practices in the Illinois Nutrient Loss Reduction Strategy where peer-reviewed studies are given the highest priority (IL EPA, 2023). Extracting data from peer-reviewed studies tended to be easier than from gray literature (state agency reports, web references) because the experimental design and reporting of methods and results were generally clearer and more complete. Nevertheless, a number of reports and other types of gray literature were reviewed (Figure 6a, gray slices). The Journal of Environmental Quality merits a special note because this journal provided many highly relevant studies; nearly a third of studies containing data appropriate for extraction were published in this journal (Figure 6b).

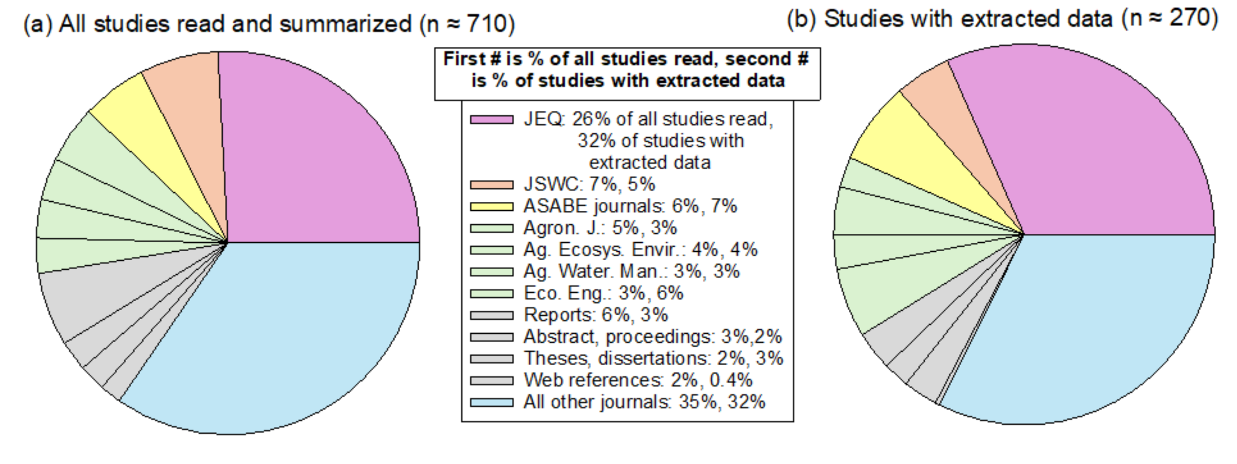

Figure 6. Data sources for the 710 studies that were reviewed (a) and for the 270 studies that contained extractable data (b).

This review also had a special focus on newer studies to ensure the most up-to-date data were compiled. That aim was easily achieved due to the growing availability of field studies that assess nutrient losses, which is perhaps in response to gaps identified in early Midwestern nutrient strategy science assessments (Figure 7). Half of the studies that provided extracted data were published during the ten years since the original Minnesota Nutrient Reduction Strategy was released (135 of the 270 studies were published since 2014; Figure 7). Studies were sourced for this review through spring 2024, but it's worth noting that a study published in early 2024 likely included data no newer than from the 2022 field season.

Finally, while there was an emphasis on newer studies, it would be errant to exclude or neglect many "*classic*" studies that are now more than two decades old (e.g., Jaynes et al., 2001; Randall et al., 1997; Randall et al., 2003; Strock et al., 2004). These seminal works have significantly underpinned understanding of nutrient loss in Minnesota and the upper Midwest. These "*classics*" are the anchor of this report.

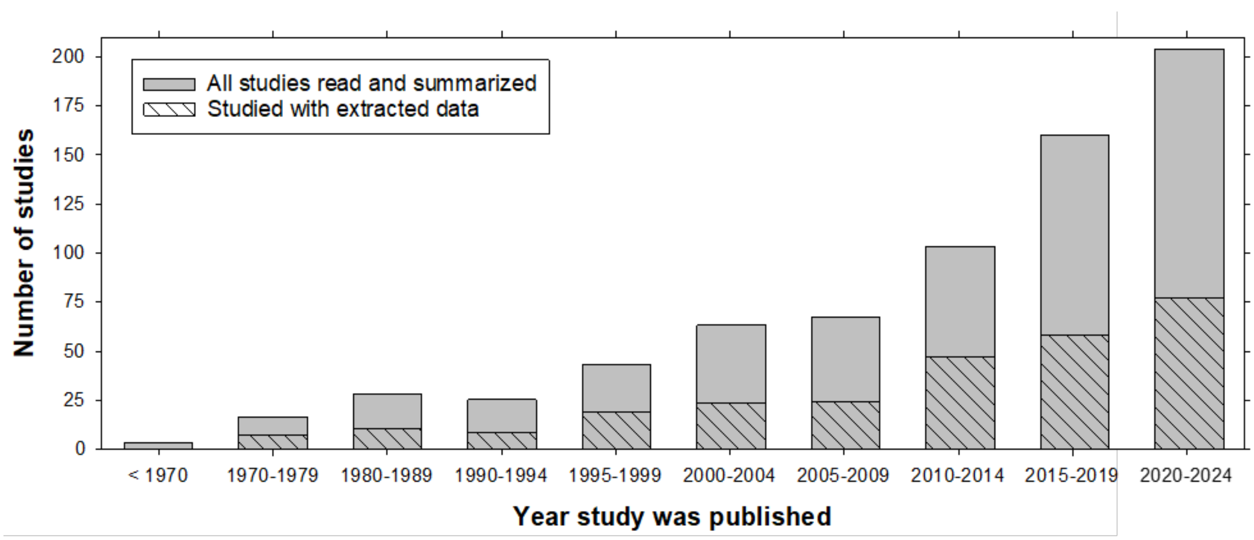

Figure 7. Distribution of the publication dates for all the studies reviewed as well as for just the studies that provided extracted data. Note the 1970's and 1980's are represented with one bar each whereas the more recent bars show 5-y increments.

Presentation and interpretation of results

Box and whisker plots were used throughout to convey the results. Generally, these figures have two panels (Figure 8). The left panel shows the ranges of nutrient losses from both the control and the treatment. The right panel shows the range of resulting nutrient reduction efficiencies from the paired site-years.

The mean is the dashed line and the median (or, 50th percentile) is the solid line in these box and whisker plots. The mean ± standard deviation shown in the right panel is the final recommended value that is also presented in the summary chapter overview table (Table 4) and in the summary table at the beginning of each chapter. Standard deviation was a simple and useful metric to illustrate the variability of the results. Note, however, that the standard deviation is not explicitly shown in a box and whisker plot.

Some of the P assessments in particular were highly variable (e.g., Figure 13). Median values were often added in white text in the right panel for the P assessments because those data ranges tended to be more skewed than the N assessments.

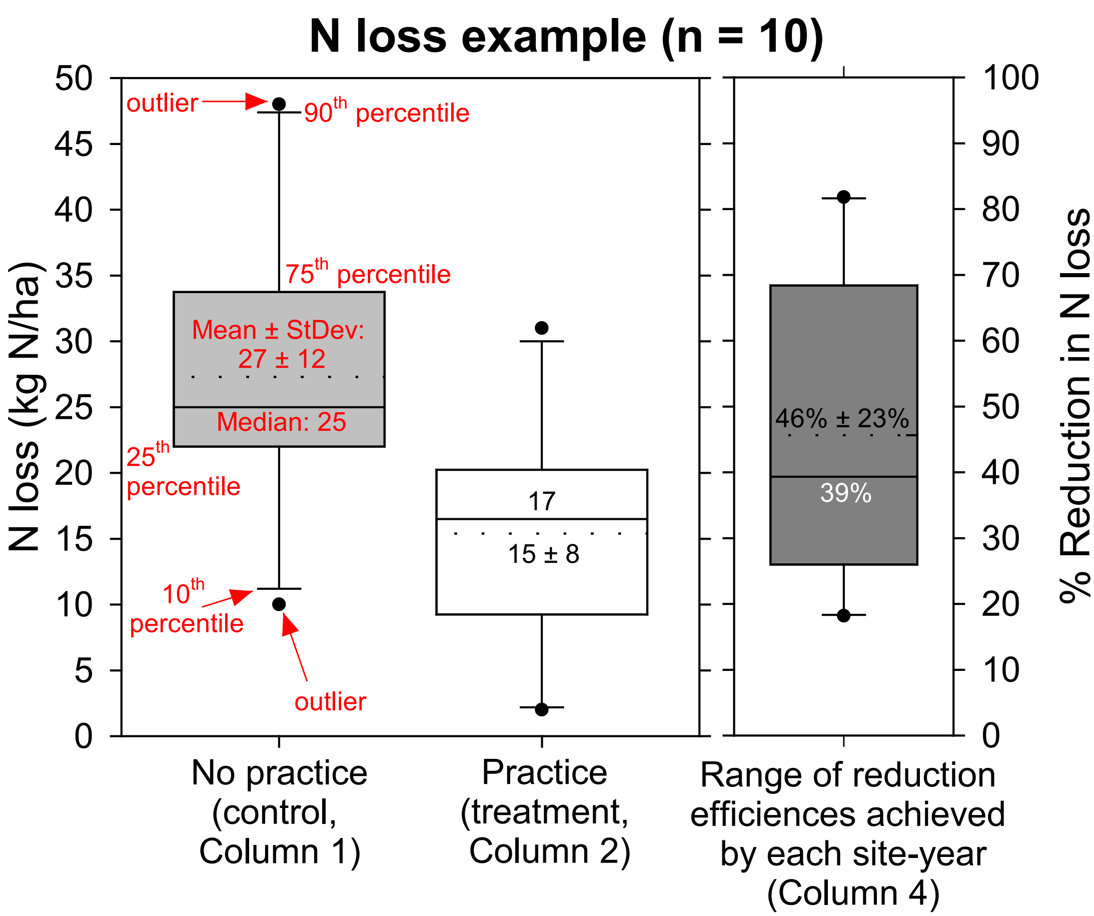

Figure 8. Example of the box and whisker plots used for results presentation. Red text describes the plot details: The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range (25th and 75th percentiles); whiskers are the 10th and 90th percentiles; dots are outliers. The data for this example were sourced from noted columns in Table 2.

The example data in Table 2, which corresponds with the results in Figure 8, help illustrate the process of developing a nutrient reduction efficiency. The three bars in the figure each have the same number of site-years (10, in this case), but the site-years are really only paired in the furthest right bar which shows the range of reduction efficiencies achieved by each site-year. The overall recommended mean reduction efficiency was based on that set of paired data. For example, the mean and standard deviation in this hypothetical dataset showed this practice resulted in a 46 ± 23% N loss reduction (median: 39%; Figure 8 right panel).

This method results in a different recommended value than if the mean N loss from the control is compared with the mean N loss from the treatment. That is, comparing mean N losses of 27 and 15 kg N/ha resulted in a 44% N reduction efficiency in these example data (i.e., (27-15)/27 = 0.44; Table 2). Simply comparing those two means (or, medians for that matter) is not the best method because the power of the data is reduced from the original sample size of ten and it eliminates the ability to develop a standard deviation.

Table 2. Example N loss data and corresponding N loss reduction efficiencies for a hypothetical practice. These data were used to make the example in Figure 8. Column 4 is calculated for each paired site-year using Equation 1.

	Column 1 No practice (control)	**Column 2 Practice (treatment)**	**Column 3 Difference in N loss due to the practice**	**Column 4 Relative difference in N loss**
	------------------------------ N loss (kg N/ha)		**-------------------------------**	**%**
Paired site-year 1	25	20	5	20%
Paired site-year 2	22	18	4	18%
Paired site-year 3	10	2	8	80%
Paired site-year 4	42	21	21	50%
Paired site-year 5	31	11	20	65%
Paired site-year 6	22	4	18	82%
Paired site-year 7	48	31	17	35%
Paired site-year 8	22	14	8	36%
Paired site-year 9	25	18	7	28%
Paired site-year 10	26	15	11	42%
Mean ± Stdev	27 ± 11	15 ± 8	11.9 ± 6.5	**46 ± 23%**
Median	25	17	9.5	**39%**
Difference in mean losses ((27-15)/15):	**44%**			
Difference in median losses ((25-17)/25):	**32%**			

Quick tips for interpretation of the box and whisker figures include:

- **Do the ranges of the light gray (control) and white (treatment) boxes in the left panel overlap a lot or just a little?** These boxes represent the interquartile range (25th to 75th percentiles) and thus represent the "*bulk*" of the data for each treatment. If there is little overlap in the two boxes in the left panel, there is likely greater certainty that the practice consistently provided an impact (either positive or negative).
- **Does the practice consistently decrease both the mean and the median** comparing the light gray (control) and white (treatment) boxes in the left panel? A consistent trend for both of those metrics helps demonstrate the "*dependability*" of the impact.
- **Do the nutrient reduction efficiency mean and median in the right panel have a consistent sign?** For example, if the mean is negative but the median is positive, the data were variable (possibly non-normal) and it was more difficult to assign a meaningful impact.

Terms and common abbreviations

The term **"*water quality*" used throughout is narrowly defined here as referring to nitrogen (N) and/or phosphorus (P)** in water. The true definition of "*water quality*", that is, "*the chemical, physical, and biological characteristics of water*", is much broader than this specific context. Note, this definition in itself doesn't have a negative connotation but rather refers to direct measures that describe water. Assessment and discussion of "*water quality*" is a function of any given project's specific scope and goal. For example, a report by BWSR (2015) discussed the water quality benefit of Minnesota's Buffer Law exclusively in terms of aquatic life (i.e., no water chemistry measures). The chemical and physical status of any water body must be "*right*" for appropriate aquatic life (the biology) to be present and healthy. Nevertheless, the primary scope for this review was to support efforts around the Hypoxia Task Force which focuses on reducing loss and transport of N and P in the Mississippi River Basin.

The term BMP (best management practice) is specifically avoided in this review. The preferred term is "*conservation practice*" which is more consistent with terminology used by the federal premier conservation agency in the US, the NRCS (e.g., "*conservation practice standard*"). The term BMP is misleading because a conservation practice is only "*best*" when it is spatially and temporally appropriate for a site (Book et al., 2024; Yuan et al., 2022).

This review attempted to be a "*jargon free zone*" so use of acronyms and abbreviations was minimized. However, several water quality terms used throughout were:

- <u>Load</u>: Mass of a nutrient flowing through a specified point during a specified time. For example, "*100 kilograms of N flowed from that tile outlet least year*." Loading concepts are discussed in context of the edge-of-field practices (saturated buffers, bioreactors, wetlands) but the general convention for this review was nutrient loss.
- <u>Loss</u>: Nutrient load normalized by area, for example "*20 kilograms per hectare*". Annual nutrient losses are the convention for this science assessment.
- <u>FWMC</u>: flow-weighted mean concentration. The concentration of a nutrient in a water sample is the mass of that nutrient in a defined volume of water. This is typically reported as milligram per liter (mg/L) for nitrate-nitrogen and often as microgram per liter (μg/L) for phosphorus. Converting a set of water sample concentrations collected over a year into an "*annual flow-weighted mean concentration*" involves weighting each sample by the time and flow rate associated with that particular sample. Those resulting values are summed over the year and normalized by the total flow that occurred during the year. An annual flow-weighted concentration provides an annual metric like load or loss but can be less "*noisy*" than load or loss because it is normalized by flow volume rather than being a direct product of flow volume.

Terms used to describe nutrients species included:

- <u>DRP</u>: Dissolved Reactive Phosphorus. Dissolved nutrients represent the sample being filtered, generally using a 45 μm filter. Many studies included as "*dissolved P*" here reported the results as DRP, although others included:
  - <u>OrthoP</u>: orthophosphate
  - <u>PO_4</u>: phosphate (PO_4^{3-})

    - SRP: soluble reactive phosphorus
    - TDP: total dissolved phosphorus
- NO_3-N: Nitrate-nitrogen; Many tile drainage studies use the terminology of "*N loss*", although technically, nitrate-nitrogen is most often what is analyzed in the laboratory. The inclusion of the "*-N*" means that only the nitrogen portion of the nitrate molecule is reported which is the standard convention for studies in the USA. Other parts of the world report nitrate as the entire ion rather than as just the nitrogen portion. This is apparent when comparing the US EPA and World Health Organization drinking water standards for nitrate (10 mg NO_3-N/L versus 50 mg NO_3/L, respectively; USEPA, 2024; WHO, 2022). Based on the molecular weights, multiplying values reported as "*NO_3-N*" by a value of 4.43 converts to "*NO_3*".
- TN: Total Nitrogen; Most tile drainage studies in the US Midwest report nitrate in lieu of total nitrogen because nitrate tends to comprise the vast majority of the total nitrogen in these waters. There were very few total N site-year reported or extracted and these data were generally not assessed here.
- TP: Total Phosphorus

Cropping system and general abbreviations include:
- CPS: Conservation Practice Standard; These are documents developed by the USDA NRCS that describe the context, application, design, and management of each conservation practice.
- CS: Corn-soybean rotation
- CC: Continuous corn
- HTF: Mississippi River/Gulf of Mexico Hypoxia Task Force
- MRTN: Maximum Return To Nitrogen; the nitrogen application rate where the economic net return to nitrogen application is greatest (Sawyer et al., 2006).
- MN NRS: Minnesota Nutrient Reduction Strategy, often referring to the original strategy document (MPCA, 2013) rather than the 2025 revision

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Chapter 2: Results summary

"Calling all practices"

The economist Thomas Sowell wrote: "*There are no solutions. There are only trade-offs*" (Sowell, 1987). While this sentiment holds some truth in agricultural contexts, it is essential to recognize that conservation practices offer substantial benefits alongside tradeoffs, especially when evaluated across multiple ecosystem services. In agriculture, there are tradeoffs, but they do not equate to an even balance; rather, the benefits often far exceed the limitations (Garrett et al., 2021). By focusing on the cumulative advantages of diverse practices, we can set realistic, achievable water quality goals. This calibration of expectations highlights that all available conservation practices are necessary, and no single group of practices alone will suffice to meet our water quality objectives (Feyereisen et al., 2022; McLellan et al., 2015). **Here, we are "*calling all practices*"** as well as calling all scientists, conservation professionals, landowners, and policymakers. Bold thinking beyond traditional conservation paradigms and courageous actions, even if they are small, are welcome at this table.

The main take-aways from this report are:

- **All conservation practices are needed** to make significant headway toward water quality goals in Minnesota. **No one practice will work on every acre, but every acre needs at least one practice.**
- **In agriculture, conservation practices come with tradeoffs, but the benefits often outweigh these limitations**. All practices, including popular ones, have some tradeoffs; however, when applied thoughtfully, they provide essential ecosystem services and cumulative benefits.
- **Upgraded and new research infrastructure is recommended** in Minnesota to address the most pressing nutrient data gaps for these conservation practices. Additionally, collaboration with other Midwest land- and non-land-grant institutions is essential to develop a regional research approach.
  See the beginning of each individual chapter for specific highlight bullets by practice type.

This report is best viewed from the perspective that:

- The recommended nutrient loss efficiencies developed for each practice are presented within the context of the entire suite of practices. This comprehensive review provides the benefit of assessing each practice, for both N and P, using a consistent approach.
- Nothing in this report is new. This was a review, thus by definition, these data and many of the findings have been previously reported.

Chapter/practice type color coding:

N fertilizer management practices	Cover cropping	Land use change perennials	Conservation drainage practices

Table 3. Interpretive summary of the benefits and tradeoffs of the conservation practices reviewed here. See each individual chapter for many additional details and nuances.

	Nitrate leaching	Other nutrient tradeoffs	Agronomic tradeoffs	Adoption tradeoffs	Data limitation tradeoffs
Chapter 3: N fertilizer management practices	These practices (as assessed) have the lowest N leaching reduction efficiencies, which may be contrary to common perception.	---	Some of these practices provide a small corn yield benefit.	These practices are widely applicable across the ≈ 8 million corn acres in Minnesota, but there may be limited ability to further adopt these practices if many acres already follow best guidance. These practices should be considered baseline practice and advancing toward water quality goals requires moving beyond these practices.	The biggest data gaps for these practices involve practice stacking (e.g., timing X rate reductions). The impacts of N application rates well above the MRTN (> 125%) and nuances of manure management were not assessed here.
Chapter 4: Cover cropping	N leaching reduction was highly variable. This benefit depends on biomass which tends to be low in Minnesota.	Cover crops reduced nitrate and total P losses in surface runoff but increased P leaching losses.	A cereal rye cover crop resulted in a small corn yield penalty in this dataset.	This practice is already widely promoted for a number of benefits. Nearly 40% N leaching reduction is possible with cover cropping when following short-season cash crops (e.g. sweet corn, peas, corn silage, wheat, edible beans) in a cold climate. By contrast, reductions average around 20% in a cereal rye corn-soybean rotation and 17% in corn-corn. Interseeding may offer additional opportunities to enhance these efficiencies.	More measures of N leaching developed on drainage plots across the state would help refine the important relationship between cover crop biomass and N loss reduction.
Chapter 5: Land use change perennials	This group of practices was the most effective for N leaching reduction.	These practices effectively reduced nutrients in runoff (note, limited data), but tend to increase P leaching.	Land use changes for most of these practices resulted in a 100% reduction in corn grain yield during the years corn was not grown. However, yield reductions when corn is continuously grown are considered separately.	These practices require some of the largest changes from the "*status quo*". Commercialization and market development for some of the newer practices (newer crops) may take years despite nitrate loss occurring today.	Direct measures of water quality are needed for the newer practices in this group. There was surprisingly little surface runoff water quality data and none of those studies were from Minnesota which is an important gap in terms of dissolved nutrients in snowmelt.
Chapter 6: Conservation drainage	These practices are effective for reducing tile drainage N losses with relatively high certainty.	---	No large changes in in-field management are necessary but only a few practices provide a yield benefit.	These practices have high capital costs and unique barriers that mean alternative conservation delivery mechanisms are needed to streamline their implementation. These practices are only applicable in tile drained areas, albeit Minnesota has more than 8 million drained acres.	There were limited data for some of these practices in Minnesota, particularly outside the southern of the state and in colder, drier regions like Red River Valley.

Table 4. Summary of recommended N and P reduction efficiencies developed in this review. "---" means the pathway was not reviewed (or was not relevant) for a given practice. † indicates the median rather than mean was used as the recommended value. ‡ indicates values from the original NRS report (MPCA, 2013).

	------------------ Leaching --------------				----------- Surface runoff -----------		
	Nitrate	**Dissolved P**	**Total P**	**‡Nitrate original NRS**	**Nitrate**	**Dissolved P**	**Total P**
Chapter 3: Nitrogen fertilizer management practices							
N rate reduction to the Maximum Return To Nitrogen (MRTN) rate							
Corn-soybean rotation – 10% rate reduction	7%	---	---	16%	---	---	---
Corn-soybean rotation – 25% rate reduction	18%	---	---		---	---	---
Continuous corn – 10% rate reduction	9%	---	---	---	---	---	---
Continuous corn – 25% rate reduction	21%	---	---	---	---	---	---
N timing modification							
100% fall to 100% spring pre-plant	5 ± 11%	---	---	---	---	---	---
100% spring preplant to a spring split	4 ± 12%	---	---	---	---	---	---
Timing modification toward spring and sidedress plus a rate reduction	Insufficient data	---	---	26%	---	---	---
Nitrification inhibitor	10 ± 10%	---	---	14%	---	---	---
Chapter 4: Winter cover cropping §							
Cover cropping in general	---	-45 ± 86%	-13 ± 43%	---	16 ± 76%	-2% †	10 ± 41%
Cover crop (with establishment success)‡	---	---	---	51%	---	---	---
Cereal rye in a corn-soybean rotation	20 ± 38%	---	---	---	---	---	---
Cereal rye in continuous corn	17 ± 33%	---	---	---	---	---	---
Oat cover crop in a corn-soybean rotation	Insufficient data	---	---	---	---	---	---
Cover crops following short season crops in a cold climate (not undersown)	39 ± 26%	---	---	---	---	---	---
Chapter 5: Land use change perennials							
Land use change / perennials in general	---	---	-94 ± 134%	---	52 ± 40%	21 ± 25% limited data	46 ± 26% limited data
Extended rotation (including perennial)	41 ± 21%	-108 ± 155%	---	---	---	---	---
In rotation: Alfalfa	63 ± 41%	---	---	---	---	---	---
In rotation: Small grain (oat)	60 ± 12%	---	---	---	---	---	---
Living covers							
Kura clover	49 ± 38%	---	---	---	---	---	---
Winter oilseed relay crops	Insufficient data	---	---	---	---	---	---
Intermediate wheatgrass	Insufficient data	---	---	---	---	---	---
Land use changes							
Conversion to prairie	94 ± 9%	-55 ± 119%	---	83%	---	---	---
Prairie strips	---	---	---	95%	66 ± 5% limited data	---	81 ± 14% limited data
Conversion to pasture	74 ± 13%	---	---	95%	---	---	---
Conversion to bioenergy crops	61 ± 54%	3% †	---	95%	---	---	---
Chapter 6: Conservation drainage practices							
Controlled drainage	45 ± 27%	30 ± 40%	30 ± 29%	38%	---	---	---
Saturated buffers	43 ± 26%	Insufficient data	Insufficient data	---	---	---	---
Denitrifying bioreactors	30 ± 21%	1% † Extrapolated	Insufficient data	13%	---	---	---
Shallow drainage	41 ± 24%	Insufficient data	Insufficient data	---	---	---	---
Gravel / blind inlets	-60 ± 93% limited data	43 ± 29%	41 ± 31%	---	---	---	---
Drainage water recycling	51 ± 21% limited data	-1 ± 80% limited data	22 ± 57% limited data	---	---	---	---
Constructed Wetlands¥	42 ± 7%			50%			

§ High N leaching reductions (e.g., ≥ 39%) are achievable under favorable conditions. However, performance is highly variable and dependent on cover crop biomass, with low biomass (< 0.5 Mg/ha) associated with a risk of increased N leaching (Chapter 4).
¥Constructed Wetlands section was added as supplementary information at the end of the report (Page 186-188).

An attempt was made to quantify more than 50 nutrient and loss pathway combinations across approximately 20 individual practices. Recommendations were made for 44 combinations with data limitations precluding development of nine additional values (Table 4). Recommended efficiencies from Table 4 are visually represented in **Figure 9 (N leaching)**, **Figure 10 (P leaching)**, and **Figure 11 (surface runoff)**. A summary of the **corn grain yield impacts** of these practices is shown in Figure 12.

Disclaimer: These recommended values were developed for the singular purpose of assigning one efficiency value per practice per nutrient loss pathway for the state of Minnesota. Data availability meant the N leaching values most generally pertained to tile drained areas and the overall scope of the nutrient strategies most directly aligns with the portion of Minnesota that is within the Mississippi River Basin. **Nutrient management decisions for individual fields (e.g., in the irrigated sands) should continue to align with the recommended agronomic best management practices (BMPs) established and published by the University of Minnesota and the Minnesota Department of Agriculture (MDA). These BMPs also support compliance with Minnesota's Groundwater Protection Rule.** For more information, visit the MDA's website: https://www.mda.state.mn.us/nfr.

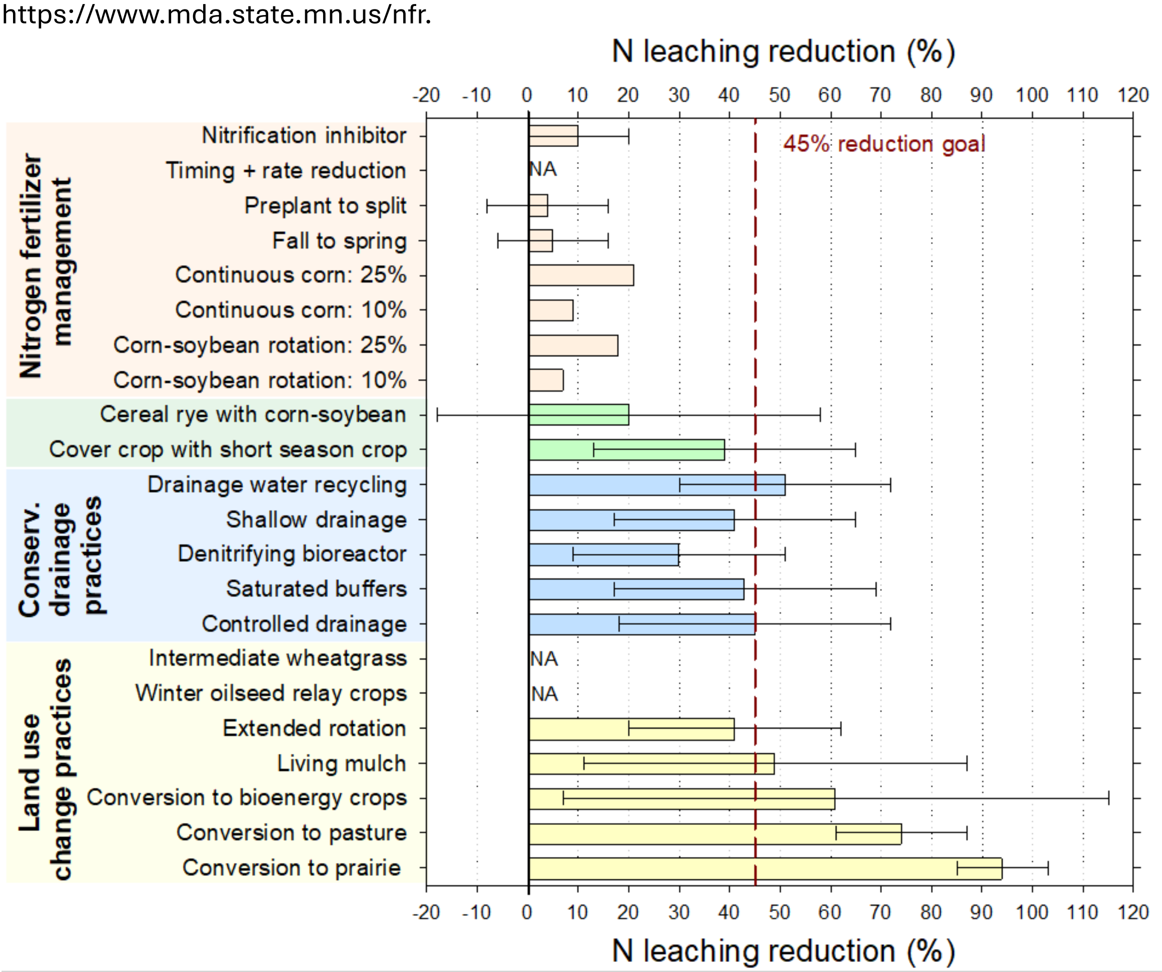

*Figure 9. Visual representation of recommended **N reduction efficiencies** for **leachate or tile drainage** developed in this review (from Table 4). The bar represents the mean and the whiskers show plus and minus one standard deviation. The method used to develop efficiency values for the practice of N rate reduction did not facilitate development of associated standard deviations.*

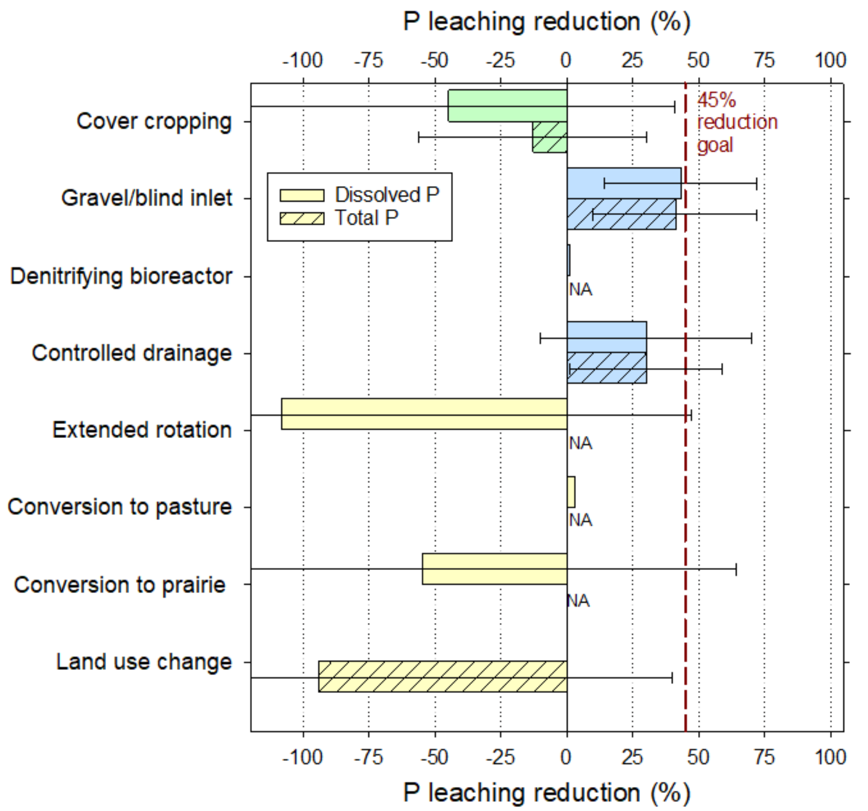

*Figure 10. Visual representation of recommended **phosphorus (P) reduction efficiencies** for **leachate or tile drainage** developed in this review. Dissolved P is shown with open bars and total P is shown with hatched bars. The bar represents the mean and the whiskers show plus and minus one standard deviation although the whisker edges are not shown to best illustrate bar magnitudes. Values taken from Table 4. Note, many of the practices reviewed were intended for N loss reduction rather than P loss reduction.*

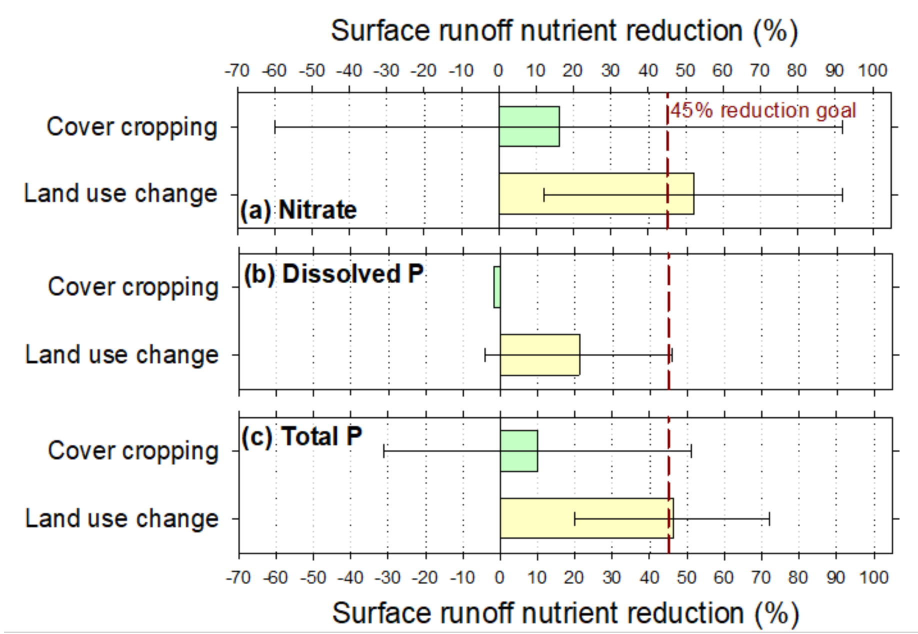

*Figure 11. Visual representation of recommended **nitrate** (a), **dissolved phosphorus** (b) and **total phosphorus** (c) reduction efficiencies for **surface runoff** developed in this review. The bar represents the mean and the whiskers show plus and minus one standard deviation. Values taken from Table 4. Note, many of the practices reviewed were intended for N loss reduction rather than P loss reduction.*

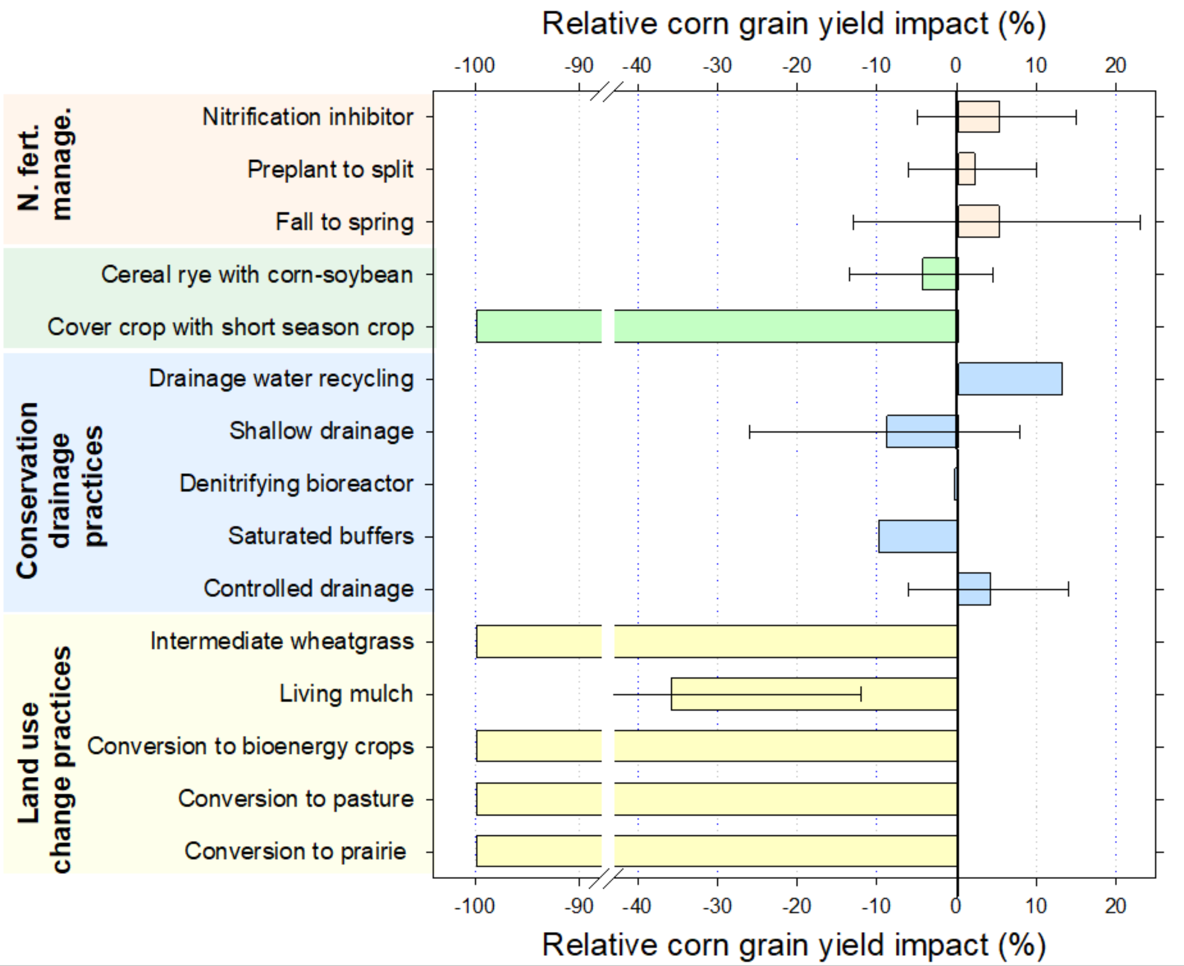

Figure 12. Visual representation of the average impact on ***corn grain yield*** *for each of the practices assessed (i.e., not corn silage or sweet corn). A 100% corn grain yield penalty was assumed for five practices that converted land away from corn grain production. It is important to note this 100% corn yield penalty for these five practices does not necessarily equate to an economic or profitability penalty if the crop grown is* profitable. ***Practices with a 100% yield penalty reflect those where corn is removed from the rotation. This does not indicate an agronomic yield impact for cases where corn continues to be grown alongside conservation practices.*** *Corn yield impacts of the N rate reduction practices were too variable to include here. The yield reduction for saturated buffers and bioreactors was the relative amount of land required for those edge-of-field practices; no in-field yield reduction is expected for those practices.*

Coefficient of variation for nutrient reduction efficiencies

Practices that provide high nutrient loss reductions are desirable, but the ability to reliably provide that benefit on a consistent basis is also important. "*Reliability*" or "*certainty*" of a practice was assessed here using coefficients of variation. The coefficient of variation is a simple statistical measure of the dispersion of data points around a mean and is calculated as:

$$Equation\ 4.\ Coefficient\ of\ Variation = \left|\frac{Nutrient\ loss\ reduction\ efficiency\ standard\ deviation}{Nutrient\ loss\ reduction\ efficiency\ mean}\right|$$

Here, the absolute value was used to normalize for the positive (nutrient loss decrease) and negative (nutrient loss increase) means.

Relatively lower coefficients of variation are preferable because they indicate less variability in a dataset. Here, less variability around a given practice's mean translates into greater certainty that that practice will provide the mean benefit. A coefficient of variation greater than 1.0 indicates the magnitude of the standard deviation exceeded the mean. If the standard deviation exceeded the mean, this was visually illustrated by the whisker error bars crossing zero in Figure 9, for example.

The coefficients of variation for the nutrient removal efficiencies developed here tended to exceed 1.0 more for the P leaching and surface runoff recommendations compared to the N leaching recommendations (albeit the sample sizes differed; Figure 13). For example, the N leaching practices provided relatively high certainty with only four of the fifteen practices having a coefficient of variation ≥1.0 (Figure 13a, the top four). Conversely, four of the five practices assessed for dissolved P leaching had coefficients of variation exceeding 1.0 (Figure 13b) and three of the five surface runoff coefficients exceeded 1.0 (across all nutrients; Figure 13d-f).

Lenhart et al. (2017) similarly noted that treatment of dissolved P in water is highly variable. They assessed performance by averaging across relevant practices for a given nutrient and reported coefficients of variation of 1.98, 1.01, and 0.32 for practices that removed dissolved P, nitrate, and sediment, respectively. Indeed, practices that address sediment (and associated particulate nutrients) have been well researched over the past century and thus have relatively high performance certainty (Lenhart et al., 2016). "*Invisible*" dissolved nutrients represent a new challenge.

Kaspar et al. (2008) noted "*high*" confidence in the ability of cover crops to reduce N leaching, but this was contextualized with the caveat "*...when reasonable establishment and growth occurs*." Thus, for this Minnesota-focused review, the relatively high coefficient of variation for N leaching reduction provided by the practice of a cereal rye cover crop may make sense (Figure 13a: 1.9). However, it is all relative because Ruffatti et al. (2019) reported cover cropping provided more consistent N leaching reduction than moving N applications from the fall to the spring. That finding was confirmed here given the "*fall to spring*" coefficient of variation exceeded that of cover cropping (Figure 13a: 2.2 > 1.9). The N fertilizer management practices had relatively high coefficients of variation due to their low mean N reduction efficiencies (i.e., small denominator in Equation 4). The "*extremely high*" confidence in perennials to reduce N leaching reported by Kaspar et al. (2008) was supported here with coefficients of variation all less than 0.9 (even as low as 0.10 for prairie conversion).

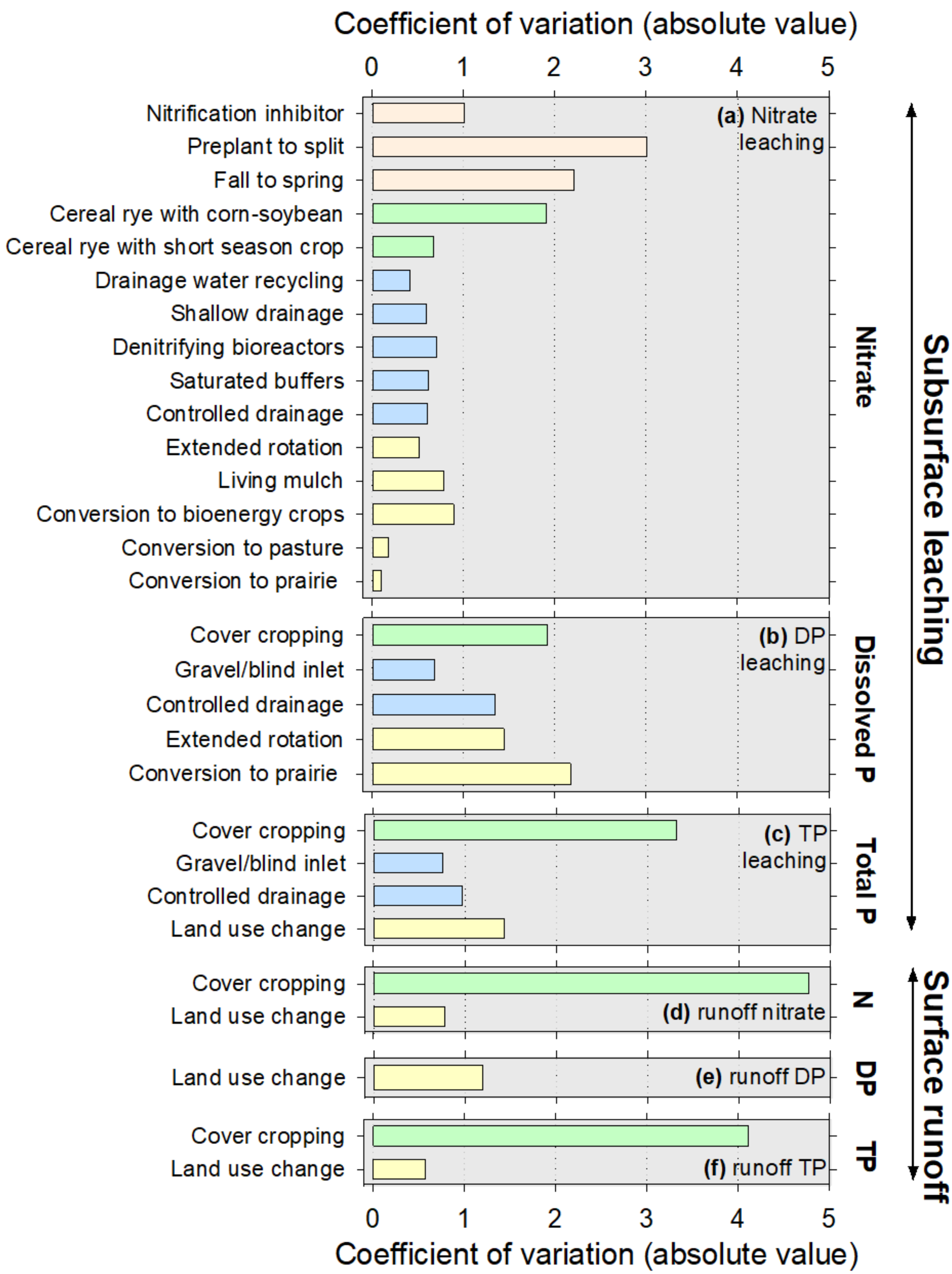

Figure 13. Coefficient of variation (absolute value) for annual reduction efficiencies for nutrients in leachate or tile drainage (a - c) and for surface runoff (d - f). Smaller coefficients of variation are preferable, with for example, values less than 1.0 indicating greater certainty for a practice.

Suggested future research

This Minnesota-focused science assessment was sometimes limited by the general availability of nutrient loss data for these conservation practices (i.e., low total site-years; Table 5, first set of columns) and/or applicability of available research to Minnesota's climate (Table 5, second set of columns). Three major recommendations for research infrastructure in Minnesota would address these knowledge gaps:

1. Upgrade existing and construct new **sets of replicated tile drainage plots** across the state.
2. Establish a set of long-term replicated **surface runoff plots** instrumented for annual data collection, including collection of snowmelt runoff.
3. Launch a monitoring and demonstration **network of field-scale conservation drainage practices** that is based on a robust experimental design.

One of the caveats of this analysis was the necessity of averaging across N losses developed using different leaching methods (e.g., see section about "*Suction lysimeters*"). Improved drainage plot infrastructure would address many of the limitations identified throughout this assessment:

- Recommended N reduction efficiencies **could not be developed for intermediate wheatgrass and relay cropping winter oilseeds** (e.g., Table 5, n = 3 for both) which are practices that are heavily emphasized in the state. These practices likely impact both N concentration and leachate/drainage volume, thus monitoring these practice's water quality impact using direct measures of both is an urgent need.
- The N leaching reduction provided by **cereal rye cover cropping had one of the highest coefficients of variation**. Although there were many site-years for this popular conservation practice (n = 60), only 17% were from Minnesota (Table 5) and all of those data were based on suction lysimeter methods. More consistent measures of N leaching developed on coordinated drainage plots across the state would help refine the important relationship between cover crop biomass and N loss reduction.
- The combined practice of N timing modification X N rate reduction had the lowest number of suitable site-years and number site-years from Minnesota (Table 5: 0 and 0%, respectively). Increased replicated drainage plot infrastructure would allow **testing of increased numbers of treatments, including stacked practices like this that better reflect the nuances of reality**.
- These N-focused practices had many tradeoffs with P, especially P in leachate. However, **older drainage research infrastructure in the state may not be able to accommodate monitoring needs** of these newer environmental challenges. For example, a once per week water sampling frequency and the bucket-and-stopwatch method for flow measurement result in high uncertainty when used to calculate annual nutrient loads, especially for P (Williams et al., 2015). Development of sophisticated drainage research infrastructure in other states has outpaced Minnesota (Clevenger et al., 2024; Spangler, 2024).

Table 5. The number of site-years used to create each recommended value (first set of columns) and the relative percentage of these site-years that were from Minnesota (second set of columns). ***The color of each cell is based on a gradient ranging from: Green****: highest count of site-years or most site-years from Minnesota;* ***Red****: lowest count of site-years or fewest site-years from Minnesota.*

	# of site-years used						% of site-years from Minnesota					
	Leaching or drainage			Surface runoff			Leaching or drainage			Surface runoff		
	NO_3	DP	TP	NO_3	DP	TP	NO_3	DP	TP	NO_3	DP	TP
Chapter 3: Nitrogen fertilizer management practices												
N rate reduction to the Maximum Return To Nitrogen (MRTN) rate												
Corn-soybean rotation	151						30%					
Continuous corn	101						43%					
N timing												
100% Fall to 100% Spring pre-plant	15						60%					
100% spring preplant to a spring split	21						19%					
Timing modification toward spring and sidedress plus a rate reduction	0						0%					
Nitrification inhibitor	15						80%					
Chapter 4: Winter cover cropping												
Cover cropping in general		30	8	20	28	26		0%	0%	0%	0%	0%
Cereal rye in a corn-soybean rotation	60						17%					
Cereal rye in continuous corn	19						53%					
Oat cover crop in a corn-soybean rotation	4						0%					
Cover crops following short season crops in a cold climate (not undersown)	24						13%					
Chapter 5: Land use change perennials												
Land use change / perennials in general			21	14	7	8			14%	0%	0%	0%
Extended rotation (including perennial)	17	14					18%	21%				
In rotation: Alfalfa	32						13%					
In rotation: Small grain (oat)	11						0%					
Living covers												
Kura clover	17						76%					
Winter oilseed relay crops	3						0%					
Intermediate wheatgrass	3						100%					
Land use changes												
Conversion to prairie	32	20					13%	0%				
Prairie strips				5		5				0%		0%
Conversion to pasture	17						0%					
Conversion to bioenergy crops	23	17					0%	0%				
Chapter 6: Conservation drainage												
Controlled drainage	38						50%					
Saturated buffers	42						14%					
Denitrifying bioreactors	57	29					7%	7%				
Shallow drainage	20						25%					
Gravel / blind inlets	5	17	14				20%	24%	7%			
Drainage water recycling	26						0%					

One of the most obvious gaps illustrated in Table 5 related to surface runoff (i.e., many of the red and orange cells in the table). Despite the widely purported benefits of perennials for reducing sediment losses in surface runoff, fewer than 15 site-years were available to document the impact of these practices on nutrients in runoff. The impact of snowmelt on annual nutrient loss reduction provided by vegetative practices is a clear data gap. This is especially important in Minnesota given the wide success of the state's Buffer Law, the proximity of these buffers to the freshwater stream network, and the cautions from Canada about limited effectiveness of vegetative buffers for P in runoff (Kieta et al., 2018; Vanrobaeys et al., 2019).

Monitoring data from Discovery Farms programs has highlighted snowmelt runoff as a critical period for nutrient transport, with frozen soil conditions exacerbating nutrient losses (Formo et al., 2018; Formo et al., 2020). While side-by-side comparisons between BMP and conventional practices are not available, temporal trends from long-term monitoring indicate that BMPs can effectively reduce nitrate losses. For example, cover crops have been shown to lower median nitrate concentrations in tile drainage, and crop rotation significantly influences nitrate losses, with corn and soybean rotations generally leading to higher concentrations than small grains or alfalfa (Radatz, 2021). Additionally, legacy N from past fertilizer applications can contribute to nitrate losses, even in unfertilized fields, underscoring the importance of sustained nutrient management (Rosen et al., 2019). These findings suggest that BMPs, though variable in performance, can still play a critical role in mitigating nutrient losses over time. New surface runoff monitoring infrastructure, ideally using a replicated plot design, would further quantify these benefits and address key data gaps related to both nutrient loss and tradeoffs.

The third research recommendation helps address unique barriers of the relatively newer conservation drainage practices. Focus groups in the state over a decade ago called for increased opportunities to see these practices at work (Lewandowski, 2010), but their adoption across the state's 8.25 million tile drained acres has nevertheless been slow. An intentional and well-designed network of saturated buffer, bioreactor, alternative inlet, and drainage water recycling projects across the state would serve the dual purposes of: 1) addressing data gaps for these practices in Minnesota (e.g., generally < 15% of site-years; Table 5) as well as 2) demonstrating effective treatment options to landowners and the public for the "*invisible*" pollutant nitrate. These networked sites should be leveraged as gateway hubs to catalyze "*Batch and Build*"-type programs in Minnesota. Such alternative delivery mechanisms will accelerate implementation of these dependable practices (e.g., low variation, Figure 13), streamline the process for farmers, and reduce total public expenditure for water quality.

This summary of data gaps is not just a blanket call for more research, however. In fact, this review identified several areas where water quality data are likely sufficient, at least in terms of our understanding of N leaching reduction. For example, the practice of controlled drainage is often assessed at 40-46% N loss reduction (45% here; Frankenberger et al., 2023; IDALS, 2024) and the coefficient of variation reflected high certainty for this practice (0.60, Figure 13). The practices of N rate reduction and timing modification are additional examples where there is a relatively large number of site-years and there has been relative consistency across assessments (e.g., 1-10%; IDALS, 2024). That is not to say there are no longer relevant research questions for these practices, but additional investment to quantify these practice's N loss reduction wouldn't notably change the efficiencies recommended here.

It is also useful to note that several recommended reduction efficiencies felt a little optimistic but were made nevertheless because those values were supported by the existing data. The recommendation of a 10% N leaching reduction provided by nitrification inhibitors was higher than the new value used in the Iowa Nutrient Reduction Strategy of only 5% (IDALS, 2024). The higher value recommended here was more consistent with the higher values from original work performed on this practice in Minnesota (i.e., 10-18% by Randall and Vetsch (2005) and Randall et al. (2003)). The recommended 30% N loss reduction for bioreactors also seemed high, but in contrast to nitrification inhibitors, this may have been impacted by the lack of data collected in Minnesota. The 41% N loss reduction recommended for the practice of shallow drainage was disproportionately influenced (increased) by one study. Finally, there was large uncertainty around the 20% mean N leaching reduction recommended for a cereal rye cover crop given the Minnesota-specific site-years averaged only 5% reduction. The 13% N loss reduction reported by Strock et al. (2004) two decades ago may be close to what a long-term impact would be for cover cropping in Minnesota.

The nutrient removal effectiveness that results when practices are "*stacked,*" or are used in combination to treat a given acre (Christianson et al., 2018), specifically was not addressed here. The scope of the state-based science assessments, thus far, has been to develop treatment efficiencies for individual practices because that reflects the state of the available science. However, in reality, practices are often used in highly nuanced combinations that vary from field to field and year to year. Many of the in-field and edge-of-field practices assessed here can be easily stacked to increase benefits to water. In-field vegetative practices like cover or relay cropping that are widely promoted in the state perform better with sufficient drainage (Kladivko, 2020), so pairing such practices with conservation drainage practices may be a natural fit. Excitingly, the practice of drainage water recycling could facilitate new opportunities for some of the in-field vegetative practices by reducing water limitations (Baker et al., 2012; Reinhart et al., 2019).

Stacking practices on a given acre cannot be quantified by simply adding practice efficiencies (e.g., following the example provided by Feyereisen et al., 2022). The resulting nutrient removal effectiveness, as well as the possibility for synergies or tradeoffs between practices, needs to be further evaluated especially in cold climates (Li et al., 2011). Nevertheless, this current science assessment illustrated that, more fundamentally, understanding of the performance of some individual practices still needs to be refined. The suggested research infrastructure will help advance science in the state and region to address these limitations toward meeting today's water quality goals and society's future agro-environmental challenges. This science assessment hopes to create a collaborative space for scientists, conservation partners, landowners and policymakers to increasingly engage in conversations about and action on water quality in Minnesota where we are "*calling all practices*."

Chapter 2 References

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Chapter 3: Nitrogen fertilizer management

Associated NRCS practice codes: CPS #590 Nutrient management

Definition: The practice of nutrient management is the use of appropriate nutrient application rates, placements, timings, and products to reduce the environmental impacts of crop production (Dinnes et al., 2002; MN NRCS, 2022). This review was for nitrogen (N) only and focused heavily on tile-drainage studies. Manure studies were not included here.

Values in the original Minnesota Nutrient Reduction Strategy:

- Recommended rates (i.e., corn after soybeans and proper manure crediting): 16% N leaching reduction
- Spring and sidedress applications with 30-lb N rate reduction: 26% N leaching reduction
- Nitrification inhibitors: 14% N leaching reduction

Table 6. Recommended N leaching reductions for nitrogen management practices

		---------------- Leaching ------------			-------- Surface runoff --------		
		Nitrate	Dissolved P	Total P	Nitrate	Dissolved P	Total P
N rate reduction to the Maximum Return To Nitrogen (MRTN) rate							
	Corn-soybean rotation – 10% rate reduction	7%	---	---	---	---	---
	Corn-soybean rotation – 25% rate reduction	18%	---	---	---	---	---
	Continuous corn – 10% rate reduction	9%	---	---	---	---	---
	Continuous corn – 25% rate reduction	21%	---	---	---	---	---
N timing modification							
	100% Fall to 100% Spring pre-plant	5 ± 11%	---	---	---	---	---
	100% spring preplant to a spring split	4 ± 12%	---	---	---	---	---
	Timing modification toward spring and sidedress plus a rate reduction	Insufficient data					
Nitrification inhibitor		10 ± 10%	---	---	---	---	---

Key Highlights:

- N management practices provide only moderate N leaching reductions (e.g., 4-10%).
- There may be limited opportunity to make large water quality gains by improving N management further because survey data indicated general consistency with recommended application rates and timings (although survey responses may underestimate actual application rates).
- The Maximum Return To Nitrogen (MRTN) approach well balances agronomic and environmental considerations. Unless N is being over-applied, there are limits to the amount of N leaching reduction achievable by reducing N rates.
- Society's perception of the effectiveness of N management practices may exceed water quality benefits that have been documented using empirical measurement in the field.
- Additional research to better document the N leaching benefit of stacked practices (e.g., timing modification X rate reduction) would be useful.

Nitrogen fertilizer management background and database overview

Nitrogen (N) leaching in agricultural systems is not caused by any single factor but is rather the combined effect of soil properties, crop management, and climate (Dinnes et al., 2002). The noncontrollable factors of precipitation and mineralization exert an immensely important influence on these N losses (Randall and Mulla, 2001). In tile drained landscapes, drainage discharge, which is driven by precipitation, is often the most important factor (Baker et al., 1975; Bakhsh et al., 2002; Bjorneberg et al., 1996; Hanrahan et al., 2019; Jaynes, 2015; O'Brien et al., 2022; Oquist et al., 2007). However, the controllable factors of cropping system and nutrient management also influence annual N leaching (Randall and Mulla, 2001). Understanding these drivers and their relative impact on N loss underpins the ability to productively discuss the benefits and limitations of conservation practices, especially the N fertilizer management practices.

It is useful to recognize at the beginning of this chapter that fine tuning N management will help reduce N leaching, but this approach alone is insufficient to meet water quality goals (Dinnes et al., 2002; Kaspar et al., 2007; Menegaz, 2021). The reduction of N leaching provided by N management practices can be variable, thus these practices will need to be paired with others (Preza-Fontes et al., 2021; Ruffatti et al., 2019; Waring et al., 2022). Limitations in the ability of N management practices to achieve water quality goals have been known for decades and have been eloquently summarized many times:

- "*In summary, policy makers must keep in mind the influence of climate, soil, and cropping system when designing environmental policy.* ***A crop production system on highly productive soils where biological influences can be significant will have difficulty attaining consistently a goal of <10 mg NO_3-N/L in subsurface drainage water, even though prudent N fertilization is followed.***" (Randall et al., 1997)
- "***The problem is not simply one of N fertilizer use, but of a corn–soybean production system created by artificial soil drainage and intensive tillage***. *...Thus,* ***NO_3 concentrations exceeding the MCL appear endemic to artificially drained soils cropped to corn within the Midwest.***" (Jaynes et al., 2001)
- "*'Fine-tuning' in-field management practices relative to rate, method, timing and form/additives of N applications has the potential to decrease NO_3-N concentrations and therefore leaching losses with subsurface drainage. ...However, given the large N needs for corn and the close relationship between yield and available N,* ***it is unlikely in most cases that N application rates can be adjusted downward more than 10 to 15% without significant economic loss of production***." (Baker, 2001)

For this review, a total of 122 nitrogen management studies were reviewed. Of these, 20 studies contained water quality data which was extracted to compile over 400 N leaching site-years. These site-years were nearly all from the US Midwest (Figure 14b) and spanned a variety of N management practices (Figure 14c). The majority of the studies were performed on replicated tile drainage plots (Figure 14a). These site-years were further sorted during analysis, so values presented in Figure 14 represent the total available data pool. Only N leaching was studied (i.e., not surface runoff) and the predominant form of N was nitrate. Manure was specifically excluded from this analysis. This review did not focus on or make separate recommendations for irrigated sandy soils although N management in those situations is distinct (Kaiser et al., 2023; Lamb et al., 2015).

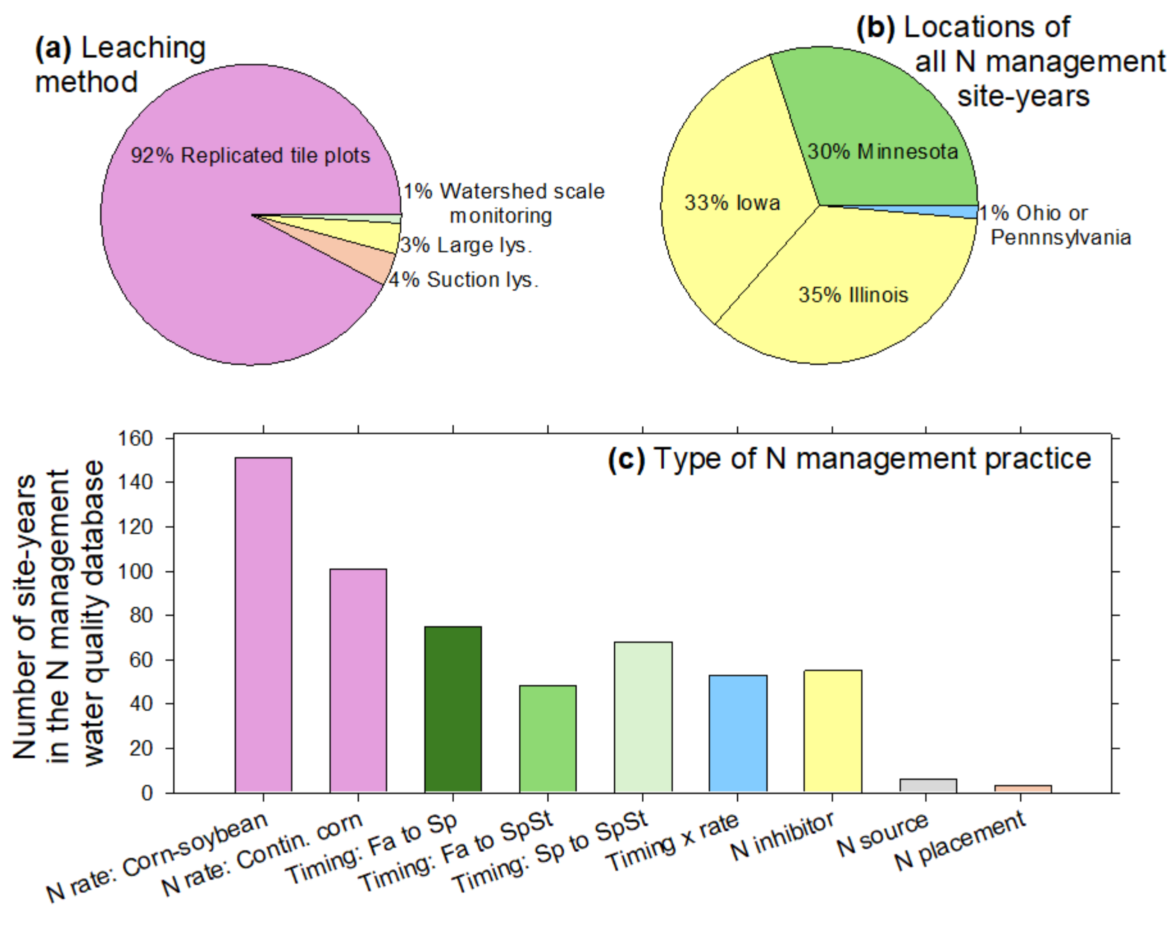

Figure 14. Description of the 452 nitrogen management practice site-years compiled into a water quality database showing: (a) prevalence of monitoring methods for nutrient leaching; (b) site-year locations; and (c) the site-year count for each type of N management practice. Fa, fall; Sp, spring; St, split; lys., lysimeter.

This analysis of N fertilizer management practices was performed using annual flow-weighted mean N concentrations (FWMC) rather than annual N losses. This was a significant methodology difference from the other practices in this report. Annual flow-weighted mean N concentration is often a more responsive parameter than annual N loss (Danalatos et al., 2022; Jaynes et al., 1999; Lawlor et al., 2008; Pittelkow et al., 2017). This is because N loss is a function of flow and its variability, whereas flow-weighted mean concentrations are essentially normalized by flow (i.e., literally, weighted by flow).

Many studies of N management practices have concluded there were no significant differences in N losses between the treatments whereas there were differences in flow-weighted N concentrations (Bakhsh et al., 2005; Helmers et al., 2012; Kanwar et al., 2005; Waring et al., 2022). This lack of difference between treatment-specific N losses is due to the effects of precipitation and variation in drainage discharge (Jaynes, 2015; Lawlor et al., 2008; Randall and Vetsch, 2005b; Vetsch et al., 2019). A large impact on drainage discharge was not expected from any of these N management practices (e.g., compared to a land use change perennial). Therefore, assessing the N leaching reduction provided by these practices using annual flow-weighted concentrations rather than loss values is a defensible and robust approach.

Nitrogen rate reduction to the MRTN

The representative nutrient loss reduction values developed for the practices in this science assessment generally used the relative differences in measured water quality between a control (non-recommended practice) versus a treatment (recommended practice) (see Equation 1). For the practice of N rate reduction, the treatment (or, recommended practice), was the use of the Maximum Return To Nitrogen rate (MRTN; Table 7). Setting the recommended practice for N rate as the MRTN was consistent with the approach to N management for corn established by land grant universities across the US Midwest (Kaiser et al., 2023). Default 0.10 prices ratios were used in this MRTN assessment (nitrogen price: $0.40/lb; corn price: $4.00/bu; Kaiser et al., 2024). The “control” in this assessment was a rate higher than the MRTN (defined below).

Table 7. Recommended Maximum Return To Nitrogen (MRTN) application rates and profitable ranges for Minnesota, Illinois, and Iowa for corn in a corn-soybean rotation and continuous corn. The application rates are shown in both lbs/acre and their converted values in kg/ha. The profitable ranges correspond to the kg/ha values. These values developed using a default 0.10 price ratio in the online MRTN calculator (Kaiser et al., 2024).

	Corn in a corn-soybean rotation				Continuous corn			
	------ MRTN -----		Profitable range		------ MRTN -----		Profitable range	
	lb N/ac	------------ kg N/ha ---------			lb N/ac	------------ kg N/ha ----------		
Minnesota	142	**159**	148	174	173	**194**	178	212
Illinois - Central	181	203	188	219	200	224	210	239
Iowa - Main	147	165	150	178	190	213	198	230
Rate 10% above the Minnesota MRTN	156	**175**	---	---	190	**213**	---	---
Rate 25% above the Minnesota MRTN	178	**199**	---	---	216	**242**	---	---

However, this type of categorical comparison (control vs. treatment) presented a challenge for the continuous variable of N rate. One of the first problems encountered with this approach was that there were few site-years extracted from literature reporting N leaching using a rate that was close to Minnesota’s MRTN (Figure 15, pink bar). This was because not only does the MRTN vary by state (Table 7), but it also varies over time, and this influenced rates used in water quality studies across states and over time.

The second complicating factor was that even if a study monitored N leaching under a rate that was relatively similar to the MRTN, that study also needed to have measured N leaching under a rate that was higher than the MRTN and this higher rate needed to be relatively consistent across studies. In the compiled N rate database, 19 site-years from four studies reported N leaching under rates of 151-167 kg N/ha in a corn-soybean rotation (i.e., within 5% of the MRTN; Figure 15a). However, the “higher” rate used across those studies was too inconsistent to allow comparison. For example, Hernandez-Ramirez et al. (2011) studied N leaching under rates of 157 kg N/ha versus a higher rate of 180 kg N/ha whereas Pittelkow et al. (2017) studied rates of 156 versus 234 kg N/ha. Those high rates represented 15 versus 50% increases over the low rate which matters in this context because the water quality benefit of the practice of N rate reduction is a function of the initial and final rates. Thus, a method that differed from the standard approach of comparing site-years from a “control” and a “treatment” was necessary for the continuous variable of N rate reduction.

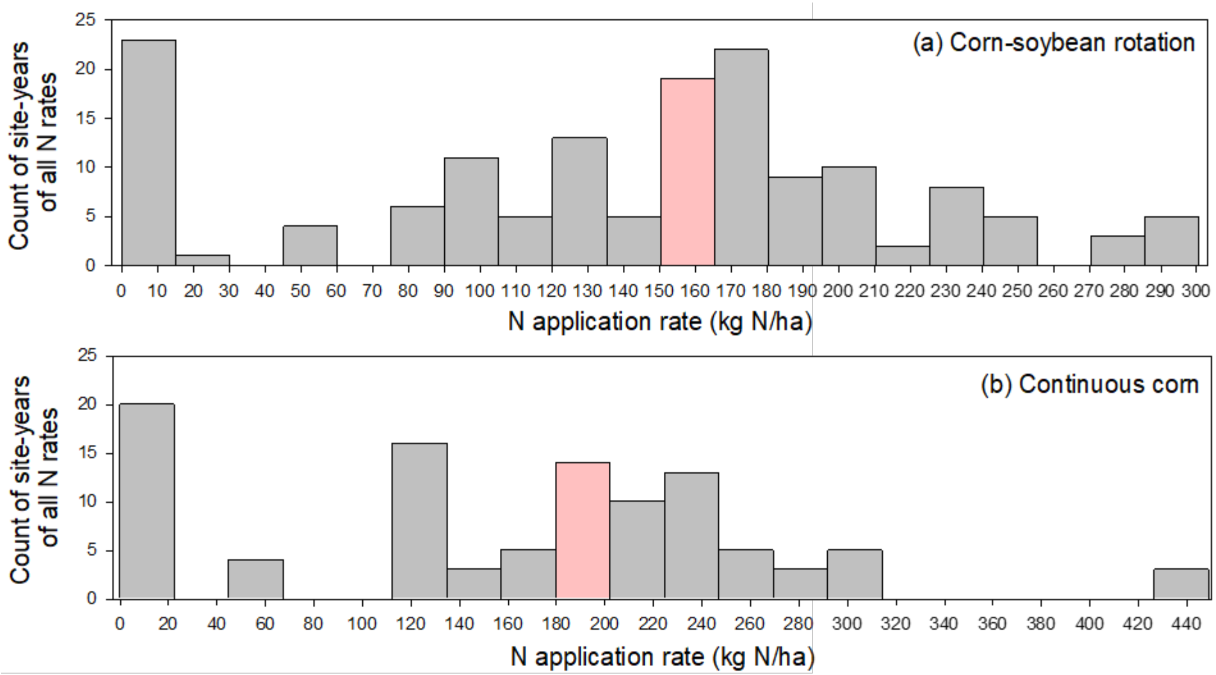

Figure 15. Histogram of site-years for corn-soybean rotation (a) and continuous corn (b) compiled in the N rate/N leaching database. The pink bars highlight the number of site-years close to the Maximum Return To Nitrogen (MRTN) rate for Minnesota (corn-soybean rotation 159 kg N/ha; continuous corn 194 kg N/ha).

The ultimate approach followed a process similar to that reported by (Lawlor et al., 2008) where N application rate and annual flow-weighted mean nitrate concentrations were correlated using an exponential growth curve (Figure 16 and Figure 17). Two N application rates of 10% and 25% above the MRTN were used for comparison with the MRTN for both corn in rotation and continuous corn (Table 7). The rate that was 10% above the MRTN was just higher than the profitable range reported for Minnesota (using the default 0.10 price ratio, Kaiser et al., 2024). Thus, these “high” rates exceeded the recommended range for agronomic production. Annual flow-weighted nitrate concentrations extracted from studies listed Table 8 in were used for this analysis.

Note that defining this practice of “N rate reduction” as 110% versus 100% of the MRTN, for example, meant that the actual reduction in rate was slightly less than 10%. The MRTN for corn in rotation in Minnesota was 159 kg N/ha, and 110% of that was 175 kg N/ha (1.1 * 159). However, a reduction from 175 to 159 kg N/ha is only a reduction of 9.1% in rate ((175-159)/175 = 9.1%). Likewise, reducing from a rate 25% higher than the MRTN results in only a 20% rate reduction from the high rate ((199-159)/199 = 20%). Nevertheless, full transparency of this description and presentation of the corresponding regressions means a N leaching reduction for any combination of N rates could be calculated by an interested reader.

For a 2-y average corn-soybean rotation, a reduction from 110% of the MRTN to the MRTN (i.e., 175 to 159 kg N/ha) resulted in a reduction of the flow-weighted mean nitrate concentration of 7% ((14.6 – 13.6)/14.6; black versus blue lines in Figure 16 panel b). Likewise, a reduction from 125% of the MRTN to the MRTN resulted in a N leaching reduction of 18% ((16.6 – 13.6)/16.6; black versus green lines in Figure 16 panel b). These results show the relative reductions in flow-weighted concentrations were slightly less than the relative reductions in N application rate. This makes sense as it would not be expected that a given benefit to N leaching would be greater than a given rate reduction. The original regression developed by Lawlor et al. (2008) in Iowa showed a similar relationship (Figure 16a).

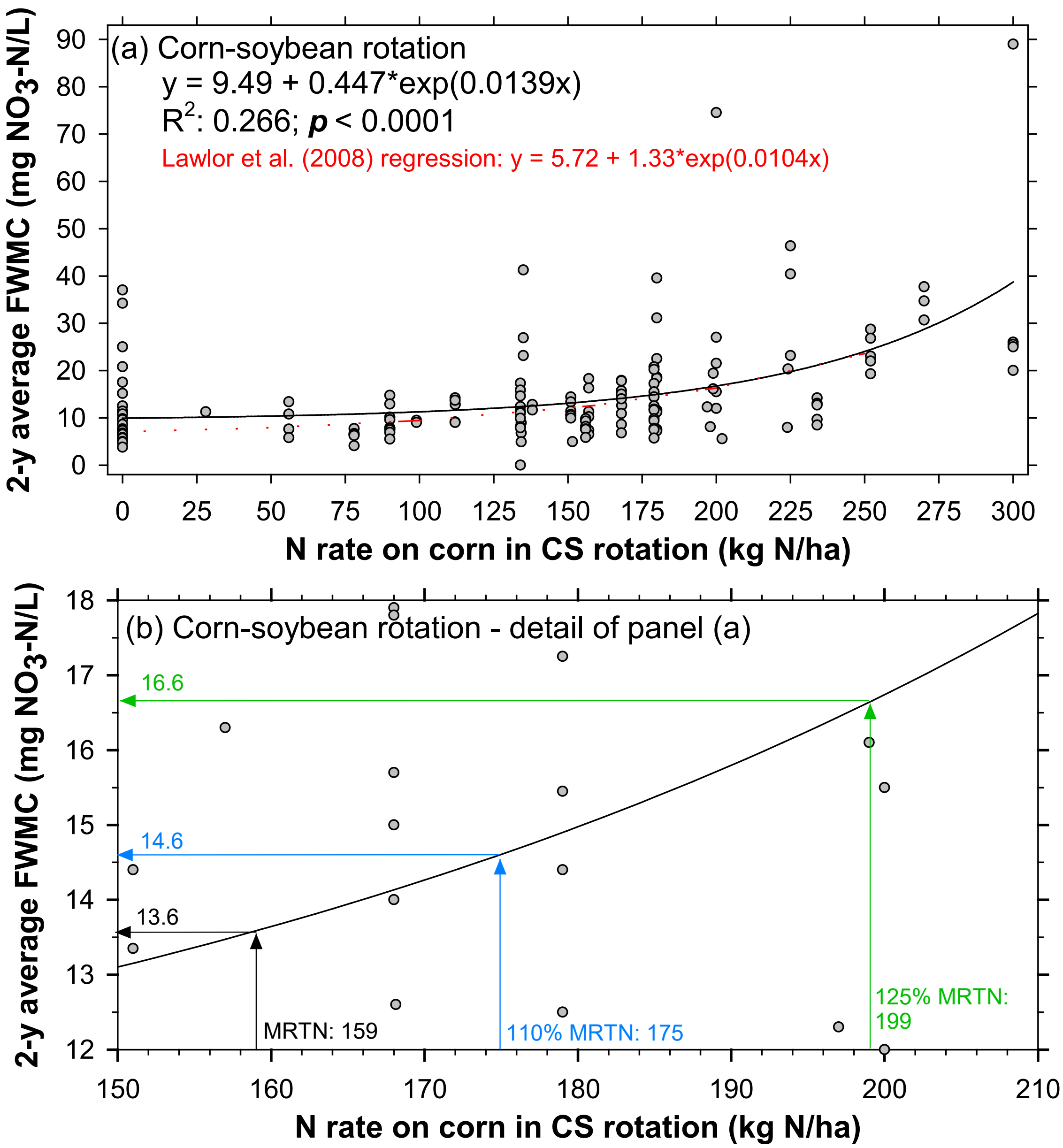

Figure 16. Exponential relationship between nitrogen (N) application rate for corn in a corn-soybean (CS) rotation and 2-year average flow-weighted mean nitrate concentration (FWMC, panel a) with detail around the Maximum Return to N (MRTN) rate in panel b. n = 151 *site-years each representing a 2-y rotation average. Shown with the original regression from Lawlor et al. (2008) in Iowa. Data were sourced from studies listed in Table 8.*

Following a similar procedure for continuous corn, a reduction from 110% of the MRTN to the MRTN (i.e., 213 to 194 kg N/ha) resulted in a reduction of the flow-weighted mean nitrate concentration of 9% ((22.2-20.2)/22.2; black versus blue lines in Figure 17 panel b). Likewise, a reduction from 125% of the MRTN to the MRTN resulted in a reduction of the flow-weighted mean nitrate concentration of 21% ((25.5 – 20.2)/25.5; black versus green lines in Figure 17 panel b).

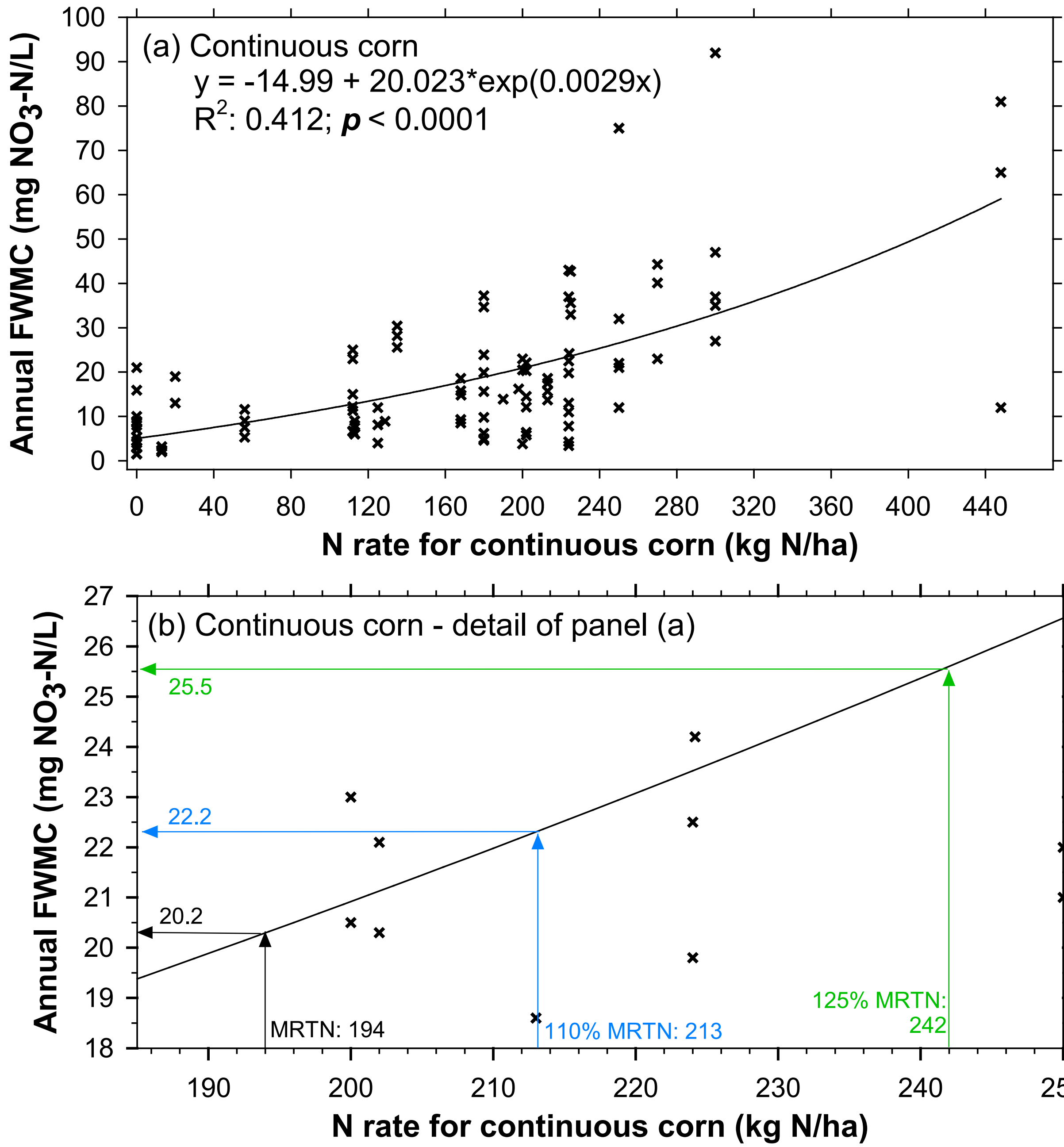

Figure 17. Exponential relationship between nitrogen (N) application rate for continuous corn and annual flow-weighted mean nitrate concentration (FWMC, panel a) with detail around the Maximum Return to N (MRTN) rate in panel b. n = *101 site-years. Data were sourced from studies listed in Table 8.*

In summary, this simple method showed that reducing N rates from **110 and 125% of the MRTN to the MRTN for a corn-soybean rotation reduced N leaching by 7 and 18%, respectively** (16 and 40 kg N/ha reductions in N rate). Those same relative reductions in N rate for continuous corn provided **9 and 21% reduction in leaching**, respectively (19 and 48 kg N/ha rate reductions).

Table 8. References for the N rate assessments in Figure 16 and Figure 17.

Corn-soybean rotation (2-y average)		**Continuous corn**	
Study	**Location**	**Study**	**Location**
(Vetsch et al., 2019)	Minnesota	(Gast et al., 1978)	Minnesota
(Radatz, 2021)	Minnesota & Wisconsin (Methods and site-years varied)	(Radatz, 2021)	Minnesota & Wisconsin (Methods and site-years varied)
(Struffert, 2016)	Minnesota (suction lysimeters)	(Struffert, 2016)	Minnesota (suction lysimeters)
(Wayment, 2021)	Minnesota (suction lysimeters)	(Wayment, 2021)	Minnesota (suction lysimeters)
(Hernandez-Ramirez et al., 2011)	Indiana	(Hernandez-Ramirez et al., 2011)	Indiana
(Ruffatti et al., 2019)	Illinois	(Preza-Fontes et al., 2021)	Illinois
(Pittelkow et al., 2017)	Illinois	(Toth and Fox, 1998)	Pennsylvania (zero tension pan lysimeters)
(Gentry et al., 2024)	Illinois	(Bolton et al., 1970)	Ontario (one multi-year mean reported)
(Waring et al., 2022)	Iowa	(Baker and Timmons, 1994)	Iowa (non-weighing lysimeters)
(Jaynes and Colvin, 2006)	Iowa	(Helmers et al., 2012)	Iowa
(Lawlor et al., 2008)	Iowa		
(O'Brien et al., 2022)	Iowa		

The original Minnesota Nutrient Reduction Strategy reported a N leaching reduction of 16% for the practice of “recommended rates” (MPCA, 2013). That single value was further refined here using the approach of a moderate rate reduction (10% above the MRTN) and more substantial rate reduction (25%). The Minnesota Watershed Pollutant Load Calculator combines the practices of improved rates and timing with a default value of 13% N leaching reduction for both tile drainage and groundwater (MPCA, 2022). The Iowa Nutrient Reduction Strategy used a lower N leaching reduction of 10% for reduction of N application rates to their MRTN (IDALS, 2014).

Leaching reduction values that trend lower (e.g., 7%, Table 1) for relatively small reductions in N rate may be the most appropriate for widespread use in Minnesota. For example, based on statewide surveys in 2010, 2012, and 2014, average N application rates were 161-166 kg N/ha for corn in rotation and 171-180 kg N/ha for continuous corn (MPCA, 2020). These reported rates were within 5% of the MRTN for corn in rotation and lower than the MRTN for continuous corn. More recently, a survey by the Minnesota Department of Agriculture reported the statewide average N application rates for corn in rotation and continuous corn were both 173 kg N/ha (MDA, 2023). This was higher than the 159 kg N/ha MRTN for corn in a corn-soybean rotation but was not beyond the bounds of the profitable range (i.e., 148-174 kg N/ha; Table 7). While farmer surveys may underreport true N application rates, these surveys nevertheless indicated there may not be much opportunity for the practice of rate reduction, because at least on average, N rates across the state have not been that much higher than the MRTN. Thus, lower N leaching reductions on the order of 5-10% rather than 15-20% may be more appropriate for widespread modelling across the state.

Nitrogen application rates that exceed the recommended optimum (e.g., exceeded the profitable range in Table 7) tend to increase N leaching without significant improvement to corn yield (Angle et al., 1993; Wayment, 2021). This effect on water quality was illustrated in the current dataset because exponential growth regressions provided better fits than linear regressions, especially for the corn-soybean rotation (Figure 16 and Figure 17, linear R^2: 0.16 and 0.38, respectively). **This exponential trend means that each unit of additional N applied at rates much higher than the MRTN disproportionately harmed water quality more than additional units applied at rates right around the MRTN. While the MRTN approach is not explicitly based on empirical measurements of water quality, these data help support its use in efforts balancing agronomic and environmental goals.**

However, Figure 16b and Figure 17b both illustrated that application of the MRTN, on average, did not result in annual nitrate concentrations lower than the drinking water standard of 10 mg NO_3-N/L (USEPA, 2024). This finding is not new. Multiple studies dating back several decades have concluded achieving tile drainage nitrate concentrations below the drinking water standard will be difficult in corn-based cropping systems, even at reasonable N application rates (Jaynes et al., 2001; Randall et al., 1997).

Tradeoffs and limitations of N rate reduction for water quality improvement

Binning the available data by N rate helped illustrate the relative impacts of rate reduction on corn yield compared with N loss, although as described above (e.g., Figure 15), binning the N rate data in this way was insufficient to develop recommended N leaching reduction values. When N rates for a corn-soybean rotation were grouped into bins, increasing the rate from the second highest bin to the highest bin (i.e., means of 174 to 233 kg N/ha applied; Figure 18a) increased corn yield by 17% but increased N loss by a relatively larger 25% (Figure 18b and d, based on mean values). At the other end of the scale, reducing N rates below recommended values can decrease corn yield relatively more than the benefit provided to N leaching (Gentry et al., 2024; Vetsch et al., 2019; Wayment, 2021). In these data, moving from a moderate application rate bin to a low bin (134 to 98 kg N/ha) reduced corn yield by 15% and actually increased N loss by 29%.

Combining these agronomic and environmental considerations, the yield-scaled N losses increased from the moderate rate bins to the highest rate bin. That is, the average yield-scaled N loss for the 153, 174, and 233 kg N/ha bins were 2.4, 3.5, and 4.0 kg N lost/Mg grain produced (Figure 18e). As more N was applied, more N was lost per unit of grain produced. Yield-scaled N losses were also high at the low end of N rates, with the 65 and 98 kg N/ha rate bins resulting in mean yield-scaled N losses of 5.1 and 4.1 kg N/Mg grain, respectively. This may be due to poor plant development and inefficient N uptake at low rates, where underdeveloped root systems limit nutrient absorption and increase N loss risk. However, further assessment should be conducted to confirm this assumption. Thus, moderate N application rates around the MRTN (e.g., 134-171 kg N/ha) optimized yield-scaled N losses, albeit this binning analysis was limited by the distribution of available N rate and water quality data.

These data bins also illustrated the variability (or, lack thereof) of N loss across N rates. For example, the mean N loss for both the 98 and 174 kg N/ha application rate bins was approximately 33 kg N/ha (Figure 18d). This provided additional visual justification for the use of flow-weighted mean concentrations in this analysis rather than annual N losses as were used for the other types of conservation practices (e.g., Figure 18c versus d). Many studies report the strong correlation between N application rate to corn and N leaching (Baker and Johnson, 1981; Helmers et al., 2012; Jaynes and Colvin, 2006; Kanwar et al., 1996; Vetsch et al., 2019; Zwerman et al., 1972). These data reiterate that while N application rate is a strong driver of N loss, it is not the only driver, and it often may not be the most important driver (Randall and Mulla, 2001).

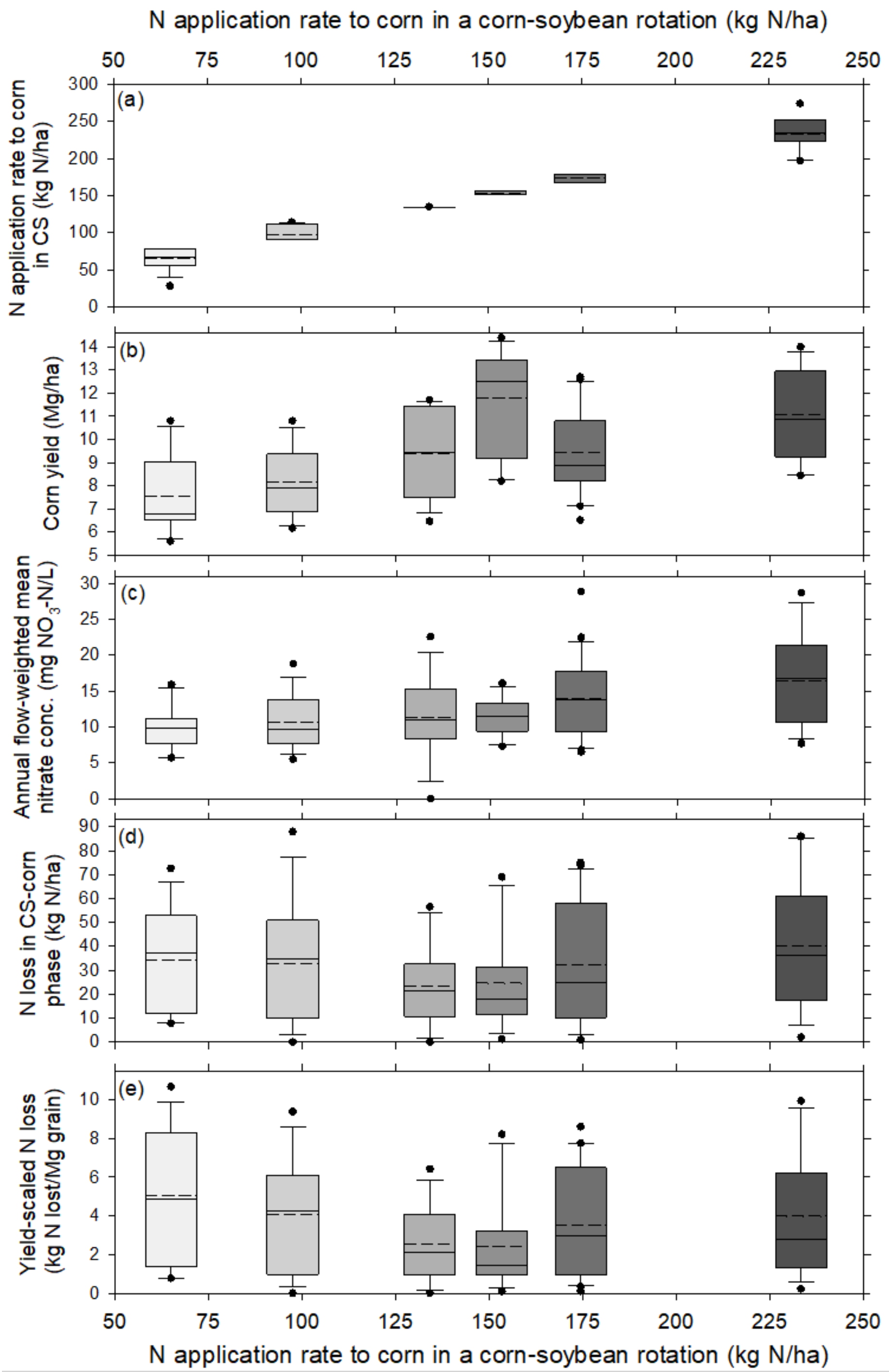

Figure 18. Nitrogen application rates to corn in a corn-soybean rotation (a) and associated ranges of corn yield (b), annual flow-weighted nitrate concentration (c), nitrate loss (d), and yield-scale nitrate loss (e) when binned by N rates. Bins were created to reflect 11-23 site-years per bin. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 8.

0 N check plots

Zero application rate check plots used in many water quality studies provide interesting insight into the nuances of N management and N leaching. Namely, N is still leached when a treatment of 0 kg N/ha is included , and these leachate nitrate concentrations can exceed the drinking water standard (Smiciklas and Moore, 2008; Struffert et al., 2016). This phenomenon can occur due to the organic matter mineralization, which releases nitrate, even in absence of external N inputs. Studies have shown that this process can contribute to significant nitrate leaching under annual cropping systems, particularly during periods of low plant uptake, such as early spring or late fall (Gupta et al., 2004; Helmers et al., 2012; Ricks, 2019). Figure 16a and Figure 17a document the wide range of nitrate concentrations that occur when no N fertilizer is applied.

This N rate database allowed comparison of studies that used both a conventional N rate on corn in a corn-soybean rotation and a 0 N check plot (replicated tile drainage plots only; Lawlor et al., 2008; O'Brien et al., 2022; Pittelkow et al., 2017; Ruffatti et al., 2019; Waring et al., 2022). These results confirmed that a treatment of 0 N to corn does not result in a flow-weighted nitrate concentration of 0 mg NO_3-N/L nor does it result in a N loss of 0 kg N/ha (Figure 19b and c). In fact, across these 0 N check plots, the average annual flow-weighted mean concentration was 6.6 ± 2.4 mg NO_3-N/L and N loss was 21 ± 18 kg N/ha (Figure 19). Of course, these water quality metrics were lower than when N fertilizer was used in the paired site-years (10.6 ± 2.9 mg NO_3-N/L, 32 ± 22 kg N/ha; Figure 19). But these metrics were not zero because crop residues and soils naturally contain organic nitrogen that undergoes mineralization annually (Gentry et al., 2024; O'Brien et al., 2022; Randall and Mulla, 2001).

Yield-scaled N losses averaged 3.3 ± 2.7 and 4.2 ± 3.7 for conventional N management and 0 N fertilizer site-years, respectively (Figure 19e). Severe restriction of N rates when growing corn resulted in more units of N being lost per unit of grain compared to when fertilizer was applied. Linking agronomic and environmental goals in this way illustrates the tradeoff that must be considered by those who propose to “*rate reduce*” our way to clean water.

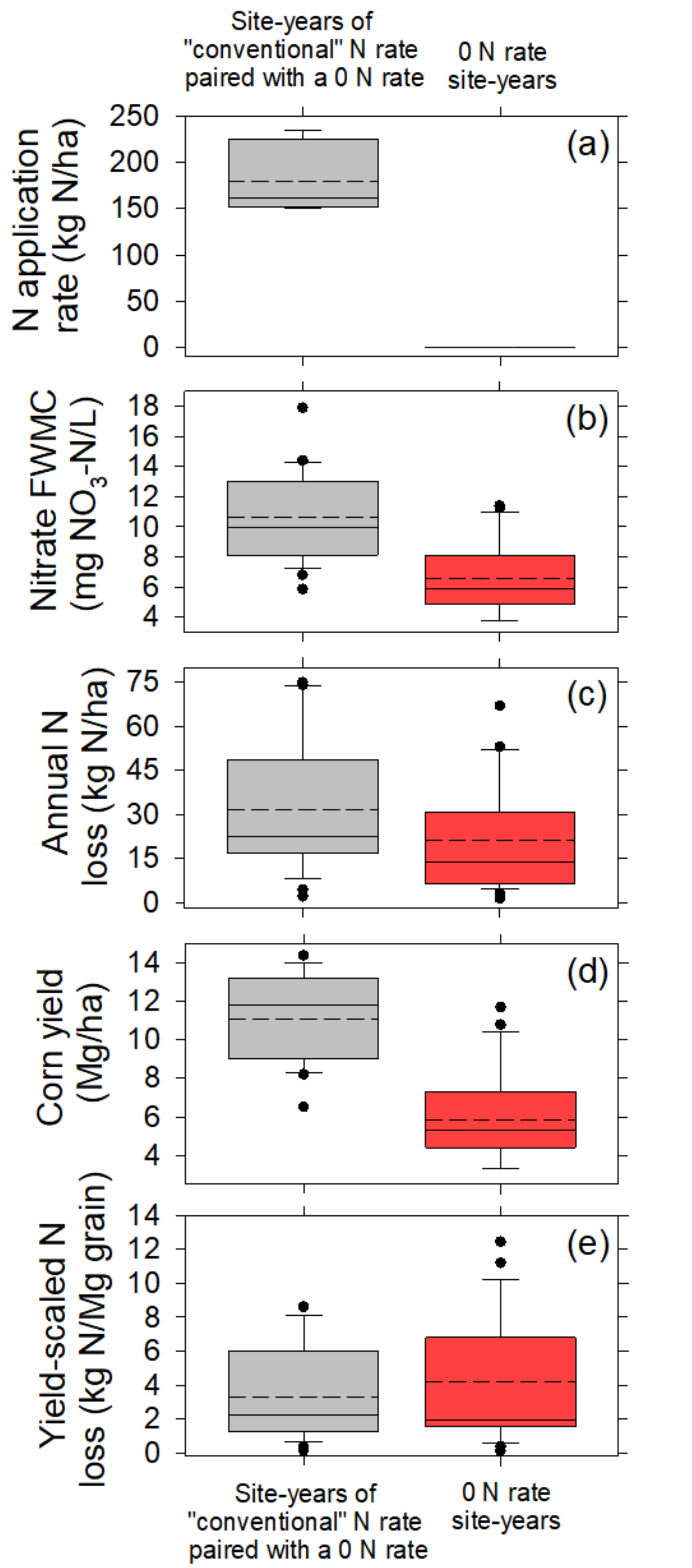

Figure 19. Nitrate leaching (b, c) and corn yield impacts (d, e) of a 0 N application rate to corn in a corn-soybean rotation. Each panel shows site-years where a conventional N application rate and a 0 N check were both performed at the same location and year (replicated tile plots only; n = 25 for all bars). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from: Lawlor et al., 2008; O'Brien et al., 2022; Pittelkow et al., 2017; Ruffatti et al., 2019; Waring et al., 2022.

Nitrogen application timing

In this review, Welch et al. (1971) provided the earliest insight into the benefits of synchronizing the timing of N application with the timing of N uptake for corn production. Later, Randall et al. (2003b) and Randall and Vetsch (2005b) reported the water quality benefits of this synchronization. These seminal studies at Waseca, Minnesota documented N leaching reductions of 10-17% by moving N applications away from the fall. Smiciklas and Moore (2008) demonstrated this effect in Illinois where weekly nitrate concentrations were more than 40% lower from spring versus fall-applied N treatments. The water quality benefits of moving N application from the fall to the spring continue to be documented especially in locations with winter drainage (Gentry et al., 2024; Pittelkow et al., 2017). Crop yield benefits (either real or perceived) may help increase adoption of this timing practice (Pittelkow et al., 2017; Randall and Vetsch, 2005a; Randall et al., 2003a; Welch et al., 1971).

Within the growing season, splitting N applications can be better than a single spring N application for corn yield (Bakhsh et al., 2002; Quinn et al., 2023; Randall et al., 2003a; Welch et al., 1971) and for water quality (Randall et al., 2003b). Bakhsh et al. (2002) reported as much as a 25% reduction in nitrate losses due to a split versus a single preplant N application in Iowa. However, there are also studies that note a moderate or no benefit due to splitting N applications (Gentry et al., 2024; Jaynes and Colvin, 2006; Pittelkow et al., 2017). There is additional nuance for the practice of in-season split N applications in Minnesota because this is the recommended practice for irrigated sandy soils (Lamb et al., 2015; Rubin et al., 2016; Spackman et al., 2019; Struffert et al., 2016). This review primarily focused on tile drained areas.

Moving from fall to spring (single applications)

Applying a single spring N application rather than a single fall N application reduced N leaching by an average of 5 ± 11% in studies performed in Minnesota and Iowa (median: 5%, n = 15, Figure 20 left panels). Reductions in flow-weighted mean nitrate concentrations due to this timing practice were higher when assessed across studies from Illinois (21 ± 10%; Figure 20 right panels). Eastern and southern Midwestern states have a greater likelihood of winter drainage than Minnesota (Christianson et al., 2016). It follows that the practice of moving N application from the fall to the spring could provide a relatively greater benefit in Illinois compared to Minnesota. Thus, data from Illinois were excluded from the recommended representative value (Table 9).

There was no synonymous practice reported in the original Minnesota Nutrient Reduction Strategy for modification of N timing alone as assessed here (see section "*Modified N rate and timing combination*"). The Iowa Nutrient Reduction Strategy originally recommended a 6 ± 25% N leaching reduction for moving from fall to spring pre-plant N application (IDALS, 2014). This was recently reviewed, and the standard deviation was revised (6 ± 16%; IDALS, 2024).

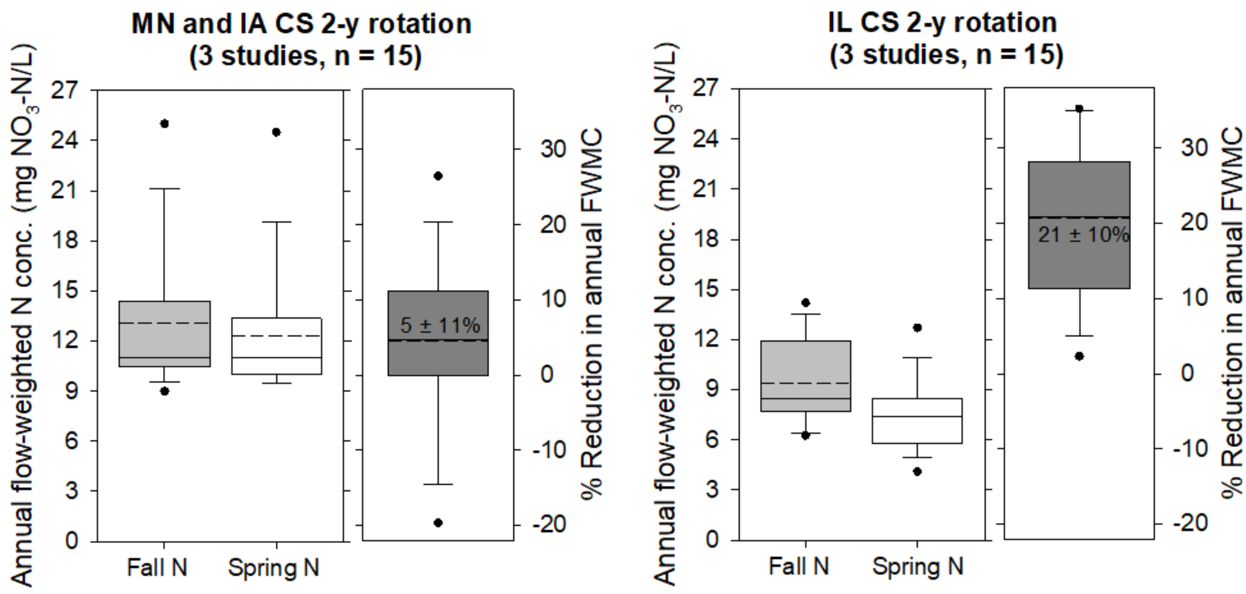

Figure 20. Range of annual flow-weighted mean nitrate concentrations (FWMC) plus associated reductions due to the application of N in the spring versus the fall in a corn-soybean rotation. Left panels show data from studies performed in Minnesota and Iowa; right panels show data from Illinois studies. One site-year represents a 2-y corn-soybean rotation. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 9.

Table 9. Studies reporting both fall and spring N application timings (as single applications) and annual nitrate leaching. One site-year represents a 2-y corn-soybean (CS) rotation.

Study	**# site-years (2-y CS avg)**	**Location**	**Method and notes**
Randall et al. (2003b)	4	Minnesota	Replicated tile plots; yield data from companion paper: Randall et al. (2003a)
Randall and Vetsch (2005b)	5	Minnesota	Replicated tile plots; yield data from companion paper: Randall and Vetsch (2005a)
Waring et al. (2022)	6	Iowa	Replicated tile plots; only study used that was not from Waseca, MN; applied nitrification inhibitor to fall treatment
Ruffatti et al. (2019) *	2	Illinois	Replicated tile plots; the “dominant portion” of the total N rate was applied in the fall or spring; Applied nitrification inhibitor to fall treatment
Pittelkow et al. (2017) *	9	Illinois	Replicated tile plots;
Gentry et al. (2024) *	4	Illinois	Replicated tile plots; applied nitrification inhibitor to fall treatment

* not used in recommended average for Minnesota

On average, there was a corn yield benefit of 5 ± 18% due to N application in the spring rather than the fall (Figure 21 left panels; median benefit: 0.3%). This benefit averaged 0.4 ± 1.3 Mg/ha. Spring application reduced yield-scaled nitrate losses compared to fall application, with an improvement of 14 ± 37% (Figure 21 right panels). These yield-scaled N losses were based on corn grain yield and the N loss only in the corn phase (i.e., not the 2-y average as shown in Figure 20).

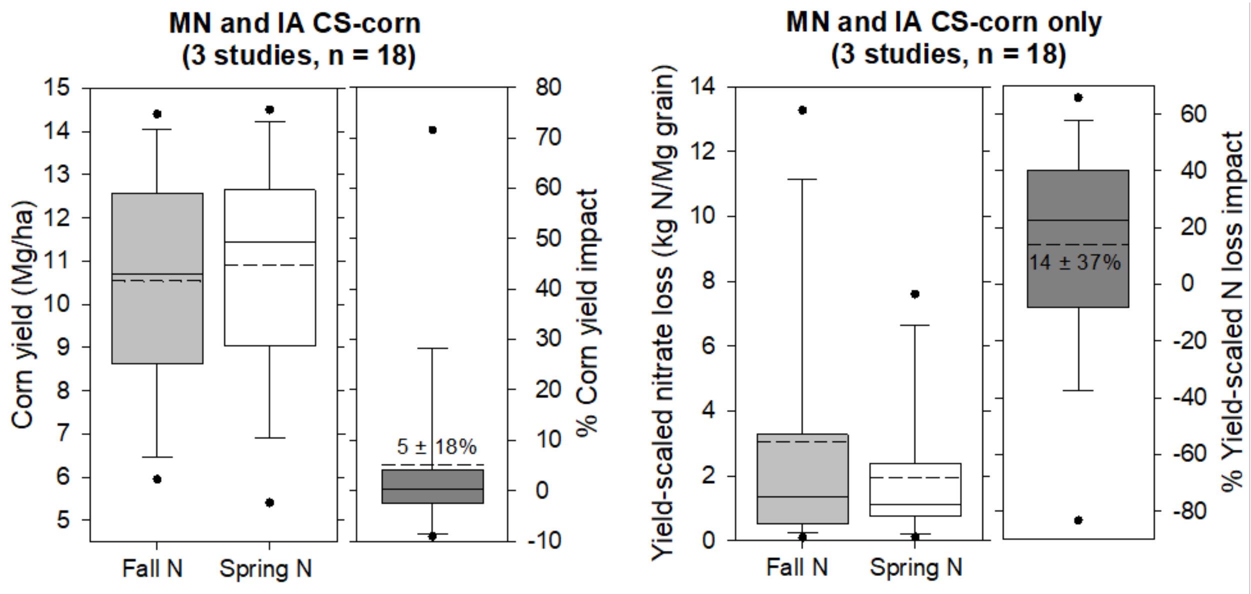

Figure 21. Ranges of corn yield impact (left) and yield-scaled N losses (right) due to single fall N application versus a single spring N application. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. "CS" refers to corn-soybean rotation and yield-scaled N losses were calculated only using the corn phase. Data were sourced from studies listed in Table 9.

A survey performed by the Minnesota Department of Agriculture for the 2019 crop year reported the majority of N applied in Minnesota is as urea (67%) with anhydrous ammonia making up an additional 22% (MDA, 2023). Only 6% of this urea was applied in the fall, whereas roughly half of the anhydrous ammonia was fall-applied (51%). Back of the envelope calculations indicate that at least three quarters of Minnesota's N applications occurred in the spring (0.67 * 0.94 + 0.22 * 0.49 = 74%). In other words, much of the state's N for corn in 2019 was applied at generally recommended timings. This would only be expected to have increased since 2019 when the Groundwater Protection Rule in Minnesota came into effect; this rule restricts fall N application in certain parts of the state (MDA, 2024).

In summary, the annual flow-weighted nitrate concentration, corn yield, and yield-scaled N loss data illustrated multiple benefits of moving N applications from the fall to the spring. However, these benefits were all relatively moderate. Approximately 75% of the N applied for corn in a corn-soybean rotation is already applied in the spring in Minnesota, so the additional benefit of increased uptake of this best practice may be small except in vulnerable areas. Additionally, more research in Minnesota is needed to determine water quality effects of using non-ammoniated P sources in the fall or moving the application of ammoniated P sources to the spring.

Moving from spring pre-plant to spring split

Splitting the N application over the season rather than applying a single pre-plant application reduced the annual flow-weighted mean nitrate concentration by an average of 4 ± 12% (n = 21; Figure 22). The median reduction was notably small (1%, Figure 22). There was no difference between states in the performance of this practice, so studies from Minnesota, Iowa, and Illinois were used to develop this recommended value (Table 10). Menegaz (2021) studied the benefits of a split N application in a continuous corn system, thus those data were extracted into the database but not used in the recommended average for a corn-soybean rotation.

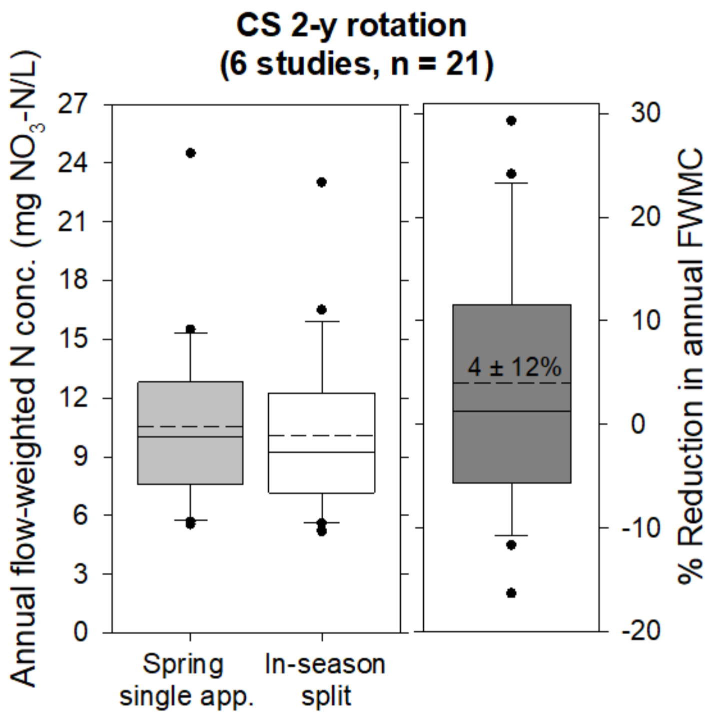

Figure 22. Range of annual flow-weighted mean nitrate concentrations (FWMC) plus associated reductions due to a single spring N application versus within spring split applications. One site-year represents a 2-y corn-soybean rotation. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 10.

Table 10. Studies reporting both single spring and spring split N application timings and annual nitrate leaching. One site-year represents a 2-y corn-soybean (CS) rotation.

Study	**# site-years (2-y CS avg)**	**Location**	**Method and notes**
(Randall et al., 2003b)	4	Minnesota	Replicated tile plots; yield data from companion paper: (Randall et al., 2003a)
(Waring et al., 2022)	6	Iowa	Replicated tile plots; pre-plant was anhydrous ammonia vs. split was urea
(Jaynes, 2015)	2	Iowa	Replicated tile plots; single pre-plant vs. single early side-dress at V6
(Jaynes and Colvin, 2006)	2	Iowa	Replicated tile plots; single post-emergence vs. post-emergence + split at V16; both UAN
(Pittelkow et al., 2017)	3	Illinois	Replicated tile plots;
(Gentry et al., 2024)	4	Illinois	Replicated tile plots;
(Menegaz, 2021) *	12	Minnesota	Replicated tile plots; continuous corn; urease inhibitor used with the split; urea and ESN

* not used in recommended FWMC average representative for CS in Minnesota

There was no synonymous practice reported in the original Minnesota Nutrient Reduction Strategy for modification of N timing as assessed here (see section "*Modified N rate and timing combination*"). The Iowa Nutrient Reduction Strategy originally recommended a 7 ± 37% N leaching reduction for side-dressing (IDALS, 2014), but this is being revised down to 1 ± 14% (IDALS, 2024).

The recommended 4% here for spring split applications fell within the new and revised values from Iowa. It was also proportionally aligned with the recommended value here of 5% for the practice of moving N applications from the fall to the spring. A shift from fall to spring is a larger change than adjusting from spring pre-plant to spring split applications so the 4% vs. 5% values make sense relative to each other. The lower reductions may reflect the influence of previous N management, as early-season N losses or limited N availability can reduce the effectiveness of split applications (Kabir et al., 2021)

On average, there was a very small corn yield benefit of 2 ± 8% due to splitting N applications over the season compared to a single pre-plant application (Figure 23 left panels; median benefit: 1.1%). In units of yield, this benefit was extremely small (0.1 ± 0.9 Mg/ha). Within season split N applications reduced yield-scaled nitrate losses compared to a pre-plant application but this benefit was correspondingly small (4 ± 40%, Figure 23 right panels; based on corn phase only).

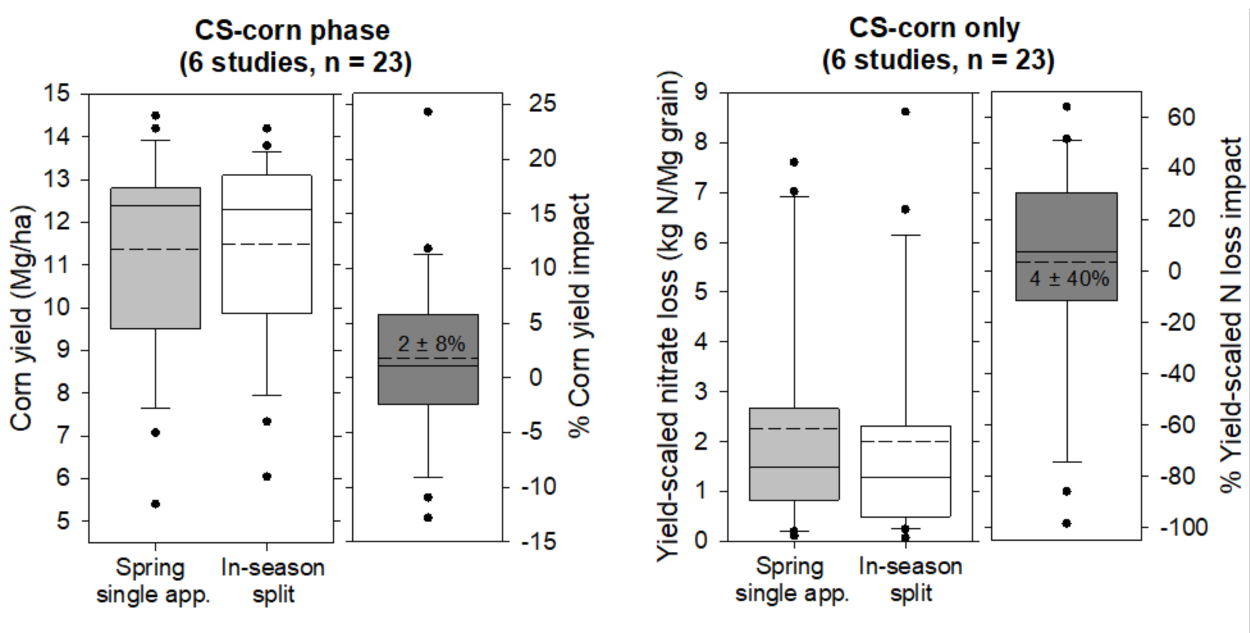

Figure 23. Ranges of corn yield impact and yield-scaled N losses due to a single spring N application versus within spring split applications. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. "CS" refers to corn-soybean rotation and yield-scaled N losses were calculated only using the corn phase. Data were sourced from studies listed in Table 10.

Modified N rate and timing combination

Seven studies in this review contained extractable data for the combined practice of N rate modification crossed with modified N application timing. In general, this practice refers to a rate reduction used in tandem with an application timing that is shifted more in-season. However, the "*control*" (non-recommended practice) and "*treatment*" (recommended practice) were too variable across this dataset to develop a meaningful N leaching reduction recommendation that was representative of a consistent practice (Table 11). For example, some studies used fall N application as the "*control*" timing whereas others used spring pre-plant. In terms of modified N rate, two of the studies (16 site-years) used increased N rates with the modified timing practice as a result of soil nitrate testing that was used to guide the in-season application (Bakhsh et al., 2002; Bjorneberg et al., 1998). Additionally, these complied site-years were for both continuous corn and a corn-soybean rotation, which further complicated the ability to develop a meaningful and representative recommendation.

Table 11. Studies reporting water quality data for the combined practice of N rate modification crossed with modified N application timing. CS, corn-soybean rotation; CC, continuous corn. For CS, one site-year represents a 2-y rotation average. Rate is in kg N/ha.

Study	# site-years	"*Control*" practice		"*Treatment*" (recommended practice)		Location	Method and notes
		Timing	**Rate**	**Timing**	**Rate**		
Wilson et al. (2020a)	2	Single pre-plant	151, 180	In-season split	100, 140	Minnesota	Modelling study but 2 site-years of original data were presented; CS and CC; replicated tile plots
Kanwar et al. (1988)	3	Single early season	175	3-way in-season split	125	Iowa	CC; replicated tile plots
Bjorneberg et al. (1998)	6	Pre-plant	110	Rate based on pre-sidedress soil nitrate test (PSNT)	93-193	Iowa	CS; PSNT was higher than pre-plant in 5 of 6 site-years; replicated tile plots
Bakhsh et al. (2002)	12	Single pre-plant	110	In-season split using late spring nitrate test (LSNT)	93-195	Iowa	CS; LSNT was nearly always higher than pre-plant; replicated tile plots
Jaynes et al. (2004)	4	Fall-applied "farmer" practice	164-188	In-season split using late spring nitrate test (LSNT)	109-177	Iowa	CS; Watershed-scale assessment
Jaynes (2015)	2	Fall	196	Pre-plant	168	Iowa	CS; replicated tile plots
Jaynes (2015)	2	Fall	196	Early side-dress (single application)	168	Iowa	CS; replicated tile plots
Hernandez-Ramirez et al. (2011)	6	Pre-plant	180	Side dress (single application)	157	Indiana	CC; replicated tile plots
Hernandez-Ramirez et al. (2011)	6	Pre-plant	157	Side dress (single application)	135	Indiana	CS; replicated tile plots

Overall, the studies compiled here were too inconsistent to develop a meaningful recommendation for N leaching reduction for the combined practices of modified N rate and timing. The Minnesota Nutrient Reduction Strategy is the only strategy that uses this combined practice; the Iowa and Illinois state strategies present recommended N leaching reductions for N rate and N timing only as individual practices (IDALS, 2014; IDOA, 2015). In the original Minnesota strategy, the practice of "*spring and sidedress applications with 30-lb N rate reduction*" (i.e., 33 kg N/ha rate reduction) is recommended as providing a 26% N leaching reduction (MPCA, 2013).

This previously reported 26% was not consistent with the recommended values developed here for the individual practices of N rate reduction or N timing modification. Based on the relationship for a corn-soybean rotation in Figure 16, applying 33.6 kg N/ha more than the MRTN (or, in other words, a 30 lb N/ac reduction to the MRTN) would result in a flow-weighted mean nitrate concentration of 16.0 mg NO_3-N/L. Thus, the rate reduction of 30 lb N/ha in the strategy's scenario would provide a N leaching reduction of 15% just for the rate reduction component (Figure 16: (16.0 – 13.6)/16.0). The benefit of stacking practices is not additive (Feyereisen et al., 2022), but for the sake of simplicity in this example, the mean of 5% for shifting N application from the fall to the spring (Table 6) could be added to this 15% benefit for the rate reduction. Thus, in this analysis, a 30 lb N/ac rate reduction combined with moving the N application from the fall to the spring would optimistically provide a benefit of 20%. This is not only lower than the value reported in the original strategy but is also notably lower than modeling results from Minnesota indicating that combined rate X timing modification could provide 33-41% N leaching reduction (Wilson et al., 2020a; Wilson et al., 2020b). **Additional empirical water quality research would improve understanding of how the benefits of a combined N rate X timing practice stack.**

Nitrification inhibitor

Use of a nitrification inhibitor with fall-applied N in a corn-soybean rotation reduced N leaching by 10 ± 10% (median: 11%, *n* = 15, Figure 24 left panels). These data heavily reflected studies performed in Minnesota and were only from replicated tile drainage plots with the inhibitor nitrapyrin (Table 12). Only fall use of nitrapyrin was considered in this analysis, which was consistent with assumptions made in Minnesota's NP-BMP tool (Lazarus et al., 2014). Randall and Vetsch (2005b) also studied the use of a nitrification inhibitor with a spring preplant N application, but those data were excluded from the recommended average here (Table 12).

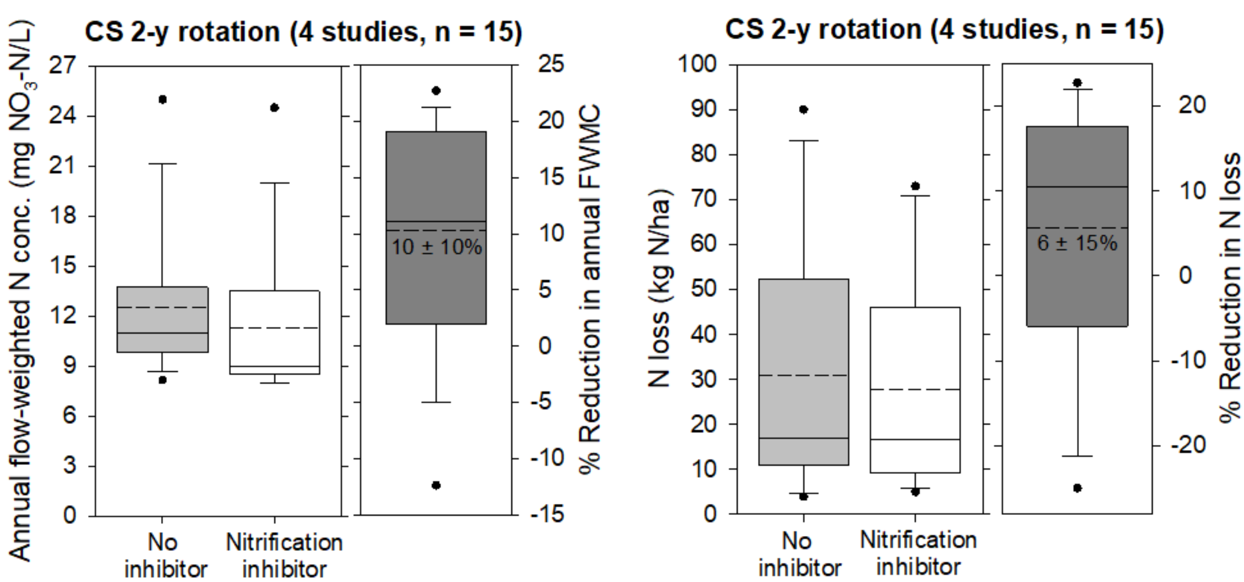

Figure 24. Range of annual flow-weighted mean nitrate concentrations (FWMC) and losses plus associated reductions due to the use of a nitrification inhibitor in the fall in a corn-soybean rotation. One site-year represents a 2-y corn-soybean rotation. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 12.

Table 12. Nitrification inhibitor studies reporting annual nitrate leaching. One site-year represents a 2-y corn-soybean (CS) rotation.

Study	# site-years (2-y CS avg)	Crop rotation	Inhibitor	Location	Method and notes
Randall et al. (2003b)	4	CS rotation	Fall applied nitrapyrin (N-Serve)	Minnesota	Replicated tile plots; yield data from companion paper: Randall et al. (2003a)
Randall and Vetsch (2005b)	5	CS rotation	Fall applied nitrapyrin (N-Serve)	Minnesota	Replicated tile plots; yield data from companion paper: Randall and Vetsch (2005a)
Randall and Vetsch (2005b) *	5	CS rotation	Spring applied nitrapyrin (N-Serve)	Minnesota	Replicated tile plots; yield data from companion paper: Randall and Vetsch (2005a)
Vetsch et al. (2019)	3	CS rotation	Fall applied nitrapyrin (N-Serve)	Minnesota	Replicated tile plots;
Pittelkow et al. (2017)	3	CS rotation	Fall applied nitrapyrin (N-Serve)	Illinois	Replicated tile plots; Only study used that was not from Waseca, MN
Souza et al. (2023) *	2 (Losses only)	CS-corn (irrigated)	DMPSA (2-(N-3,4-dimethyl-1H-pyrazol-1-yl) succinic acid isomeric mixture)	Minnesota	Suction lysimeter at 120 cm
Souza et al. (2023) *	2 (Losses only)	Corn (rainfed)	DMPSA	Minnesota	Suction lysimeter at 120 cm
Owens (1987) *	12	Continuous corn	Nitrapyrin	Ohio	Monolith lysimeters: 1 control, 2 treatments for 6 years; 336 kg N/ha N rate

* not used in recommended FWMC average for fall-applied nitrification inhibitor in a CS rotation

Other studies that reported use of a nitrification inhibitor but were not included in the recommended average included that by Souza et al. (2023) who only reported N losses rather than flow-weighted concentrations (Table 12). Owens (1987) studied nitrification inhibitors using monolith lysimeters but applied a relatively high N application rate (336 kg N/ha). They reported annual N losses over the 6-y study averaging 160 and 117 kg N/ha for the control versus inhibitor treatment which were notable outliers in the current work. Finally, Waring et al. (2022) reported using a nitrification inhibitor, but this was used as a part of their control practice rather than as a treatment. That is, they compared N leaching from under fall applied N with a nitrification inhibitor versus N leaching from spring at-plant and spring split N application timings.

Early studies appear to indicate nitrification inhibitors were more effective for N leaching reduction than later studies. For example, Randall et al. (2003b) and Randall and Vetsch (2005b) reported benefits of 18 and 10% N loss reduction, respectively, due to nitrapyrin used with a late fall ammonia application. More recently, however, neither Pittelkow et al. (2017) nor Vetsch et al. (2019) observed any significant N leaching benefit of this practice. Souza et al. (2023) showed their nitrification inhibitor decreased nitrate leaching in only one of four site-years (mean increase of 21 ± 39% in N leaching; n = 4).

Given these recent studies in particular, a reduction from the 14% used in the original Minnesota Nutrient Reduction Strategy for nitrification inhibitors is justified (MPCA, 2013). The Illinois Nutrient Loss Reduction Strategy and the original Iowa Nutrient Reduction Strategy recommended values of 10% and 9 ± 19% N leaching reduction for nitrification inhibitors, respectively (IDALS, 2014; IDOA, 2015). These values were very consistent with the 10% mean in the current work. However, the Iowa value has recently been revised down to 5 ± 20% N reduction effectiveness (IDALS, 2024). This indicates the recommended 10% value here may be a little generous for nitrification inhibitors, but barring any additional studies, this is reasonable for use in Minnesota.

On average, there was a small corn yield benefit due to the use of a nitrification inhibitor compared to fall N application without an inhibitor (Figure 25 left panels). This mean benefit was 5 ± 10%, although the median indicated only a 2% increase. In units of yield, the benefit averaged 0.4 ± 0.7 Mg/ha. However, the potential for this agronomic benefit was inconsistent across literature and was not always observed especially in the more recent studies (Pittelkow et al., 2017; Souza et al., 2023; Vetsch et al., 2019). Use of a nitrification inhibitor improved yield-scaled nitrate losses in the corn phase by an average of 12 ± 25% (Figure 25 right panels).

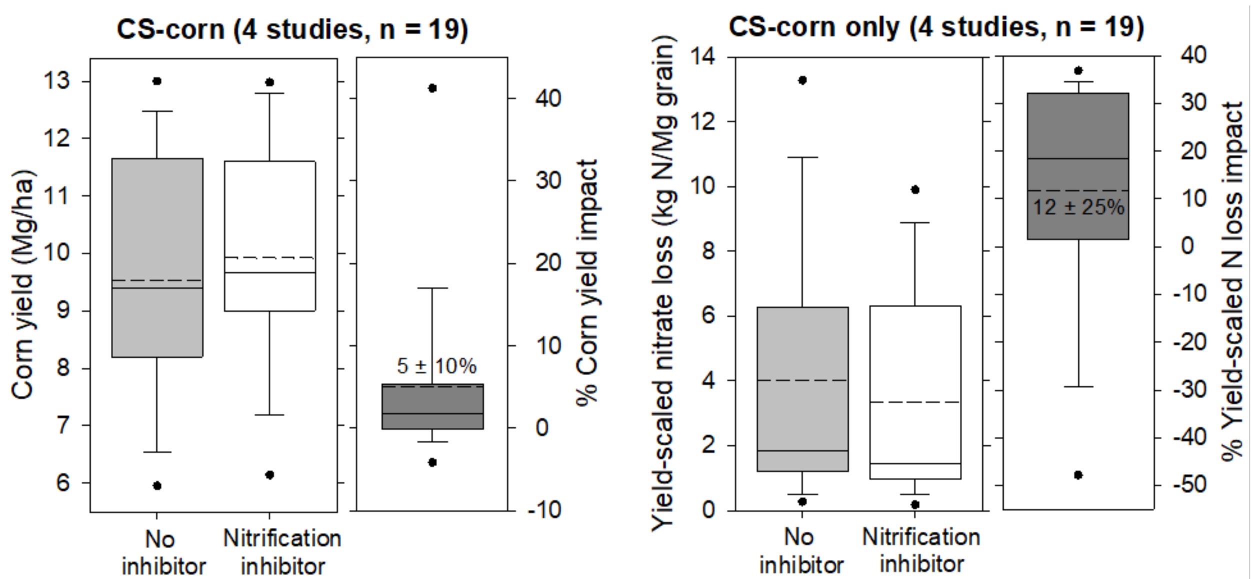

Figure 25. Ranges of corn yield impact and yield-scaled N losses due to a fall-applied nitrification inhibitor. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. "CS" refers to corn-soybean rotation. Data were sourced from studies listed in Table 12.

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Chapter 4: Winter cover cropping

Associated NRCS practice code: CPS #340 Cover crop

Definition: A cover crop is a seasonal living ground cover (grass, legume, forb, etc.) planted for the literal purpose of covering the soil (Kaspar and Singer, 2011; MN NRCS, 2017). A cover crop is planted into or after the main cash crop and is commonly killed before the next cash crop (Hartwig and Ammon, 2002).

Values in original Minnesota Nutrient Reduction Strategy: 51% N leaching reduction but only 20% establishment rate in a full season cropping system (inferred overall rate of 10% N leaching reduction); 51% N leaching reduction when used with a short season cropping system.

Table 13. Recommended nutrient loss reductions for cover cropping in Minnesota.

	------------------ Leaching --------------			----------- Surface runoff -----------		
	Nitrate	**Dissolved P**	**Total P**	**Nitrate**	**Dissolved P**	**Total P**
Cover cropping in general	---	-45 ± 86%	-13 ± 43%	16 ± 76%	-2% †	10 ± 41%
Cereal rye in a corn-soybean rotation	20 ± 38% *	---	---	---	---	---
Cereal rye in continuous corn	17 ± 33%	---	---	---	---	---
Oat cover crop in a corn-soybean rotation	Insufficient data	---	---	---	---	---
Cover crops following short season crops in a cold climate (not undersown)	39 ± 26%	---	---	---	---	---

* Minnesota-only studies: 5 ± 38% (see "*Minnesota-specific*…" section)

† Recommended the median value rather than mean ± standard deviation due to inconsistent impacts of those two metrics (positive or neutral median versus negative mean)

Key highlights:

- A cereal rye cover crop in a corn-soybean rotation reduced N leaching by an average of 20 ± 38% based on studies from Iowa, Illinois, and Minnesota. This average was 5 ± 38% when only Minnesota sites were used, but one study performed on irrigated sands using suction lysimeters heavily (negatively) influenced this result.
- Low cereal rye biomass production (less than approximately 0.5 Mg/ha) presents a risk of increased N leaching. This is especially relevant in Minnesota given a tendency for low biomass production due to the cold climate.
- High N leaching reduction (e.g., 39% or more) is possible for cover crops in cold climates but changes to the primary cropping system (e.g., growing short season primary crops) or cover crop management (e.g., under sowing, interseeding) are necessary to achieve this.
- Cover cropping has a negative impact on dissolved P losses in both above and belowground pathways, but the magnitude of this impact is small.
- It is recommended that cover cropping be studied using replicated tile drainage plots in locations across Minnesota to further refine these values. Additionally, research is needed to assess the efficacy of more diverse seed mixes on nutrient reduction efficiency.

Cover crop review background and database overview

The practice of cover cropping has existed for thousands of years (Meisinger et al., 1991), with the first published studies of this practice dating from roughly a century ago (Christianson et al., 2021). Scientific interest in cover crops has grown exponentially in recent years with, for example, eight reviews on the topic of cover cropping and nitrogen (N) leaching over the past decade (Abdalla et al., 2019; Blanco-Canqui, 2018; Christianson et al., 2021; Daryanto et al., 2018; Nouri et al., 2022; Quemada et al., 2013; Thapa et al., 2018; Valkama et al., 2015). These previous reviews and others often report non-legume cover crops reduce N leaching losses by 50 to 75% (Meisinger et al., 1991; Tonitto et al., 2006). However, the important context of climate and cropping system challenges the application of these findings in Minnesota.

The ability to achieve desirable environmental outcomes with cover cropping is increasingly difficult in cooler climates (Cates et al., 2018; Malone et al., 2014; Miguez and Bollero, 2005). For example, both Strock et al. (2004) and Wilson et al. (2014) reported cover crop establishment in Minnesota would be successful in roughly one out of every four years. The 5-y update to the Minnesota Nutrient Reduction Strategy reported the use of cover crops was increasing at a rate of approximately 40,000 acres per year, but cover crops were currently being done on less than 2% of corn and soybean acres (MPCA, 2020). Despite the many on- and off-farm benefits of cover cropping, landowners face both agronomic and structural barriers for adoption of this practice (Roesch-McNally et al., 2018). This review aimed to further quantify the benefits and limitations of cover cropping to improve water quality in Minnesota.

A total of 186 cover crop studies were reviewed during August 2023 to May 2024. Of these, 50 studies contained water quality data which was extracted to compile 300 cover crop water quality site-years. Each "*site-year*" consisted of measured water quality data from a non-cover crop control paired with water quality data from a cover crop treatment in the same location and year. The 300 site-years spanned both leaching and runoff pathways (Figure 26a and b); all study locations (Figure 26d); all cover crop types (Figure 26e); and multiple cash crop rotations. These site-years were further sorted during analysis, so values presented in Figure 26 represent the total available data pool.

Major trends in research influenced the data available for this systematic review. Firstly, winter cereal rye has been studied much more than any other cover type (Figure 26e). Secondly, nitrate leaching is the most well documented loss pathway studied for cover crops (i.e., 237 for leaching versus < 45 site-years for runoff; Figure 26a and b). Few total nitrogen (TN) site-years were extracted from literature, so N losses in this report focus on nitrate. In terms of location, there were few cover crop studies performed in Minnesota that provided appropriate, extractable water quality data (see section "*Minnesota-specific N leaching data and analysis*"), so data from surrounding states and similar climates were often used.

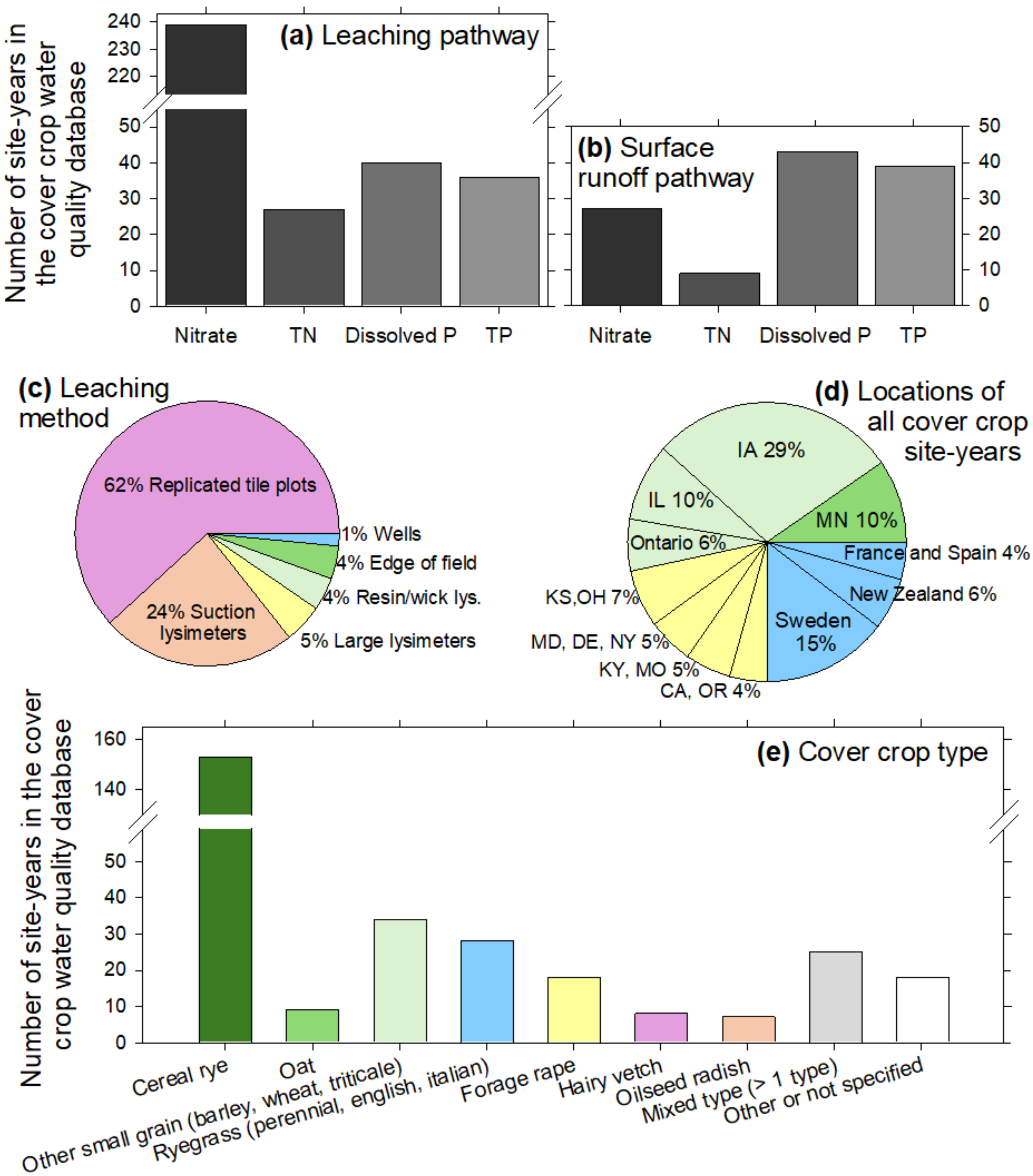


Figure 26. Description of the 300 cover crop site-years compiled into a water quality database showing: (a) and (b) the site-year count for each pathway and nutrient; (c) prevalence of monitoring methods for nutrient leaching; (d) site-year locations; and (e) the site-year count for each cover crop type.

Cereal rye cover crop in a corn grain cropping system

Cereal rye and nitrate leaching

The average nitrate loss reductions for cereal rye in a corn-soybean rotation (2-y average) and in a continuous corn system were 20 ± 38% and 17 ± 33%, respectively (Figure 27, top panels). The respective medians were 27% and 14%. The average annual loss reduced by a cereal rye cover crop for these two corn cropping systems was 8.3 ± 14 and 7.8 ± 14 kg N/ha, respectively (not explicitly shown).

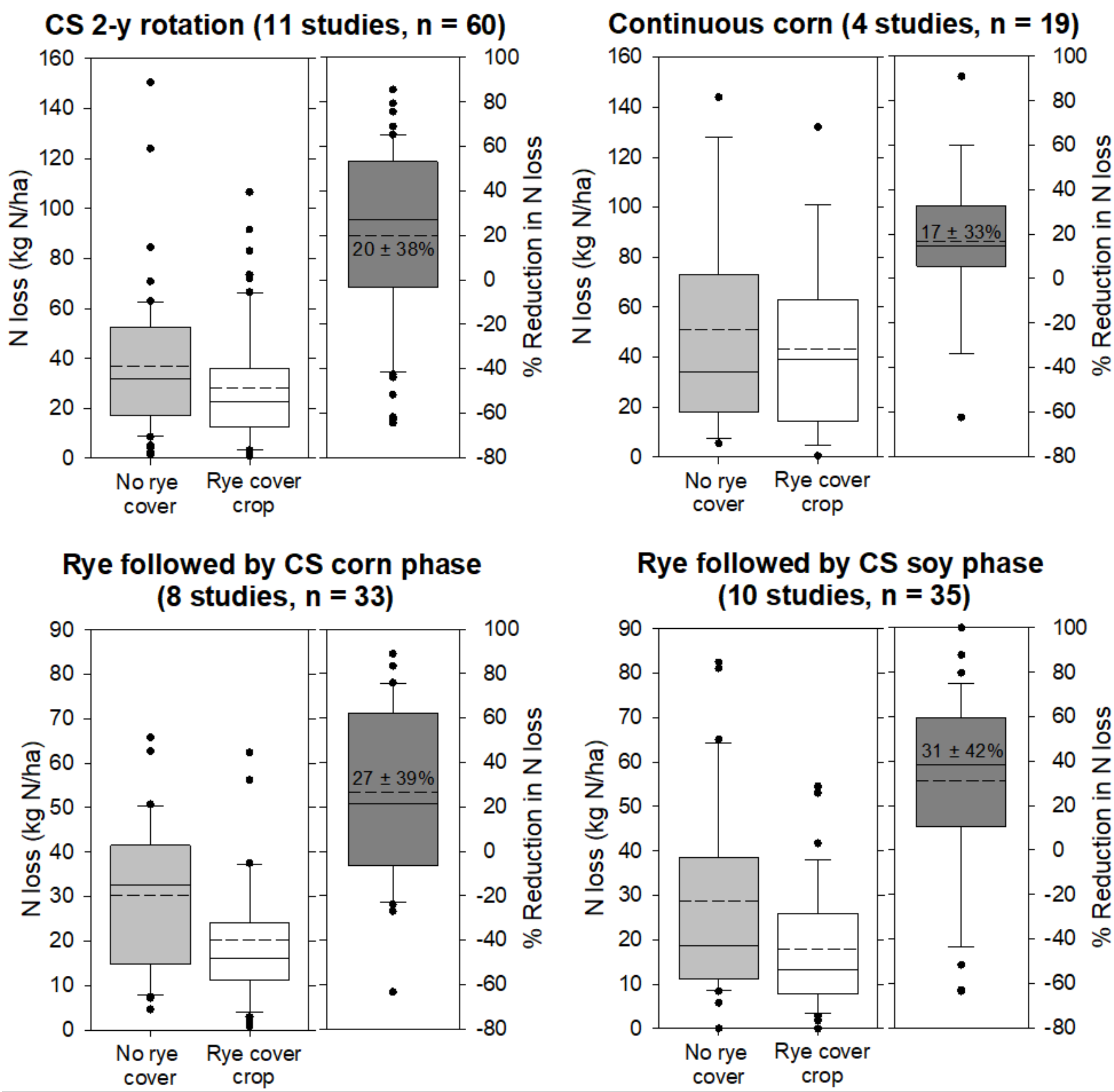


Figure 27. Ranges of nitrate leaching loss and loss reduction due to a cereal rye cover crop in corn grain cropping systems. The 2-y average for a corn-soybean rotation is shown in the upper left panels with individual crop phases shown in the lower panels; continuous corn is shown in the upper right panels. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 14.

Two-thirds of the site-years presented in the above results were from Iowa (Table 14). However, the one study from Minnesota that was included merits special attention. This thesis by Wayment (2021) provided 10 site-years of cereal rye in a corn-soybean rotation and 10 site-years of cereal rye in a continuous corn system. This was the only study included in this dataset that used suction lysimeters and this study was performed on irrigated sands which was a notably different application than the rest of the studies on clay loam or silty clay loam soils. Since this study does not translate well to other regions of the state, such as Red River Valley or southeastern Minnesota, future research is needed to replicate these findings across diverse MN landscapes to refine reduction efficiency values. Nevertheless, this thesis was the only suitable Minnesota study that could be included so that the resulting recommended value would not simply be a duplicate of the recommended value used in Iowa for a cereal rye cover crop.

Table 14. Studies reporting nitrate leaching reduction due to a winter cereal rye cover crop in a corn grain-based cropping system. Note a "site-year" for a corn-soybean rotation was a 2-y rotation average. Studies with "phases reported separately" were used to create the individual phase panels (the lower panels) in Figure 27.

Study	# site-years	Corn system	Location	Method and notes;
Qi et al. (2011)	4	Corn-soybean	Iowa	Replicated tile plots; Corn and soybean phases reported separately
Waring et al. (2020)	10	Corn-soybean	Iowa	Replicated tile plots; Split plot design including corn and soybean together
Waring et al. (2024)	10	Corn-soybean	Iowa	Replicated tile plots; Split plot design including corn and soybean together
Dougherty et al. (2020)	8	Corn-soybean	Iowa	Replicated tile plots; Corn and soybean phases reported separately
Kaspar et al. (2007)	2	Corn-soybean	Iowa	Replicated tile plots; Corn and soybean phases reported separately
Rogovska et al., (2023)	7	Corn-soybean	Iowa	Replicated tile plots; Corn and soybean years reported separately; re-published rye results from (Kaspar et al., 2012)
O'Brien et al. (2022)	2	Corn-soybean	Iowa	Replicated tile plots; Corn and soybean years reported separately
Gentry et al. (2024)	4	Corn-soybean	Illinois	Replicated tile plots; Corn and soybean phases reported separately
Ruffatti et al. (2019)	2	Corn-soybean	Illinois	Replicated tile plots; Corn and soybean phases reported separately
Johnson et al. (2024)	1	Corn-soybean	Illinois	Replicated tile plots; Corn and soybean years reported separately
Wayment (2021)	10	Corn-soybean	Minnesota	Suction lysimeter; Corn and soybean phases averaged (CSb, SbC); Two N application rates
Daigh et al. (2015)	4	Continuous corn	Iowa	Replicated tile plots
Martinez-Feria et al. (2016)	2	Continuous corn	Iowa	Replicated tile plots
Preza-Fontes et al. (2021)	3	Continuous corn	Illinois	Replicated tile plots
Wayment (2021)	10	Continuous corn	Minnesota	Suction lysimeter; Two N application rates

The original Minnesota Nutrient Reduction Strategy reported an N loss reduction of 51% for winter cover cropping paired with an establishment rate following a full season crop of 20% (MPCA, 2013). Thus, it was implied that a cover crop following corn or soybeans (i.e., full season crops) would provide a 10% N leaching reduction over a long term average (51% x 20%). The Minnesota Watershed Pollutant Load Calculator uses a higher default value of 28% N leaching reduction to both groundwater and tile drainage for "*cover crops with corn and soybeans*" (MPCA, 2022). The value used in that MPCA calculator was notably similar to the 31% N leaching reduction recommended for use in Iowa for a cereal rye cover crop (IDALS, 2014). However, a N leaching reduction value used for Iowa would be expected to be higher than a value used to represent the state of Minnesota given the slightly cooler climate of the latter. Thus, the overall N leaching reduction recommended here of 20 ± 38% for a cereal rye cover crop in a corn-soybean cropping system provided a mid-range value between the original Minnesota and Iowa Nutrient Reduction Strategies. **It is highly recommended that cover cropping be studied using replicated tile drainage plots in Minnesota to further refine an appropriate N loss reduction value for this state**.

Minnesota-specific N leaching data and analysis

The overall review of 186 cover crop studies included 23 studies from Minnesota. Of those 23, N leaching losses could be extracted from four studies which yielded 26 Minnesota-specific site-years (Table 15). Four additional studies (of the 23 reviewed) reported cover crop biomass which was extracted and used for the biomass analysis (Table 15; see discussion in section "*Cereal rye biomass and N uptake*").

Table 15. Cover cropping studies performed in Minnesota with nitrate leaching measurement or used as supplemental information for cover crop biomass production. "Conc. only" studies did not report annual losses. "Biomass only" studies did not have associated water quality data appropriate for this review.

Study	# site-years	Cash crop following the cover	Cover	Location in Minnesota	Method and notes;
Strock et al. (2004)	3	CS-soybean	Cereal rye	Lamberton	Replicated tile; Rye followed by soybean treatment only
Krueger et al. (2012)	2	Corn silage	Cereal rye	Morris	Suction lysimeter at 120 cm
Krueger et al. (2013)	1	Corn silage	Cereal rye	West central MN	Edge of field tile sampling
Wayment (2021)	10	CS rotation	Cereal rye	Westport	Suction lysimeter; 5 years each at 2 N rates
Wayment (2021)	10	Continuous corn	Cereal rye	Westport	Suction lysimeter; 5 years each at 2 N rates
Weyers et al. (2019) *	2 (conc. only)	Soybean (previous crop spring wheat)	Cereal rye	Morris	Suction lysimeter at 100 cm
Radatz (2021) *	1 (conc. only)	Corn	Not specified	Not specified	Edge of field tile sampling; averages from a number of site-years
Lenhart (2021) *	3 (conc. only)	CS rotation	Cereal rye (97%)	Martin County	Edge of field sampling; pre- and post- years
Everett et al. (2019) *	20 (biomass only)	Corn, corn silage	Cereal rye	South and central MN	No appropriate water data
Krueger et al. (2011) *	2 (biomass only)	Corn silage	Cereal rye	Morris	No appropriate water data
Liu et al. (2019b) *	4 (biomass only)	Corn, soybean	Cereal rye	Lamberton	No appropriate water data
Wilson et al. (2019) *	6 (biomass only)	Corn	Cereal rye	Rosemount	No appropriate water data

* not used in recommended average for N loss reduction for Minnesota-only studies

Minnesota cover cropping studies that reported N leaching losses resulted in an average reduction of 5 ± 38% due to this practice (Figure 28; Krueger et al., 2013; Krueger et al., 2012; Strock et al., 2004; Wayment, 2021). In terms of order of magnitude, this 5% average was not notably different from the 10% N leaching reduction implied in the original Minesota Nutrient Reduction Strategy for use of cover cropping with full season crops (MPCA, 2013). In other words, considering conservation drainage practices reduced N losses by approximately 30-50% and perennials by >40% in this review, the 5% for cover crops developed here was relatively similar in magnitude to the 10% originally used. The 5% mean for Minnesota studies was also not notably different from modeling performed by Kladivko et al. (2014) that estimated cover cropping in Minnesota's Nicollet and Fillmore counties would achieve roughly 10% N loss reductions at the county-scale.

It is important to clarify that while site-specific studies reported low nitrate reductions, our best average expectation for Minnesota over a muilt-year period remains 20%. This number accounts for years when cover crop germination and establishment are successful and when conditions for N uptake are optimal.

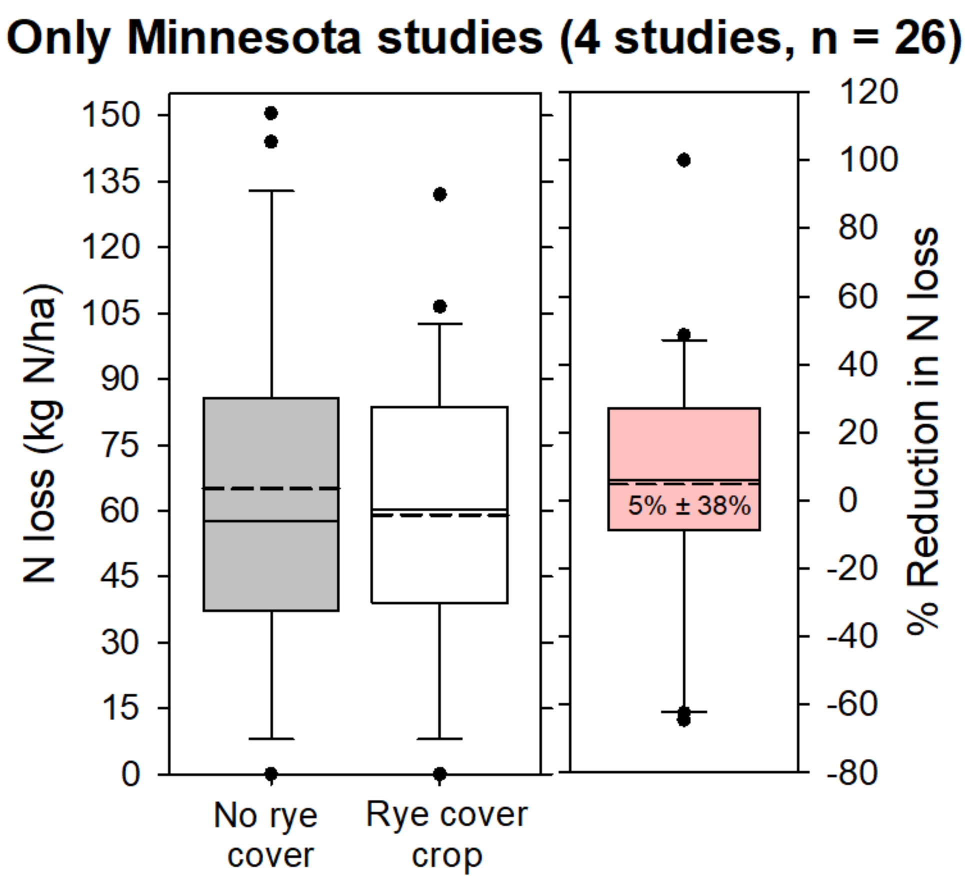


Figure 28. Ranges of nitrate leaching loss and loss reduction due to cover cropping in Minnesota. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 15.

Twenty-two of the 26 site-years used in this Minnesota-only analysis used suction lysimeters, and twenty of those were from an individual thesis which was performed on irrigated sands (discussed above; Wayment, 2021). Thus, the recommendation from above is reiterated here: It is recommended that cover cropping be studied using replicated tile drainage plots in

Minnesota, where direct measures of both N concentration and discharge can be made, to further refine an appropriate N loss reduction value for this state.

Influences on N loss reduction due to a cereal rye cover crop

The benefit provided by a cereal rye cover crop in units of N loss reduced (i.e., kg N/ha reduced) increased as the control N loss increased (Figure 29a). This confirmed earlier findings. Parkin et al. (2006) showed cover crops can take up more N at increased manure application rates and Valkama et al. (2015) showed that N loss reduction (in kg N/ha) provided by cover crops increases at increasing N application rates. This relatively robust linear relationship in the current work provides rationale for the use of an efficiency metric (expressed as a percent) across a range of N losses. That is, the relative N leaching reduction benefit provided by cover cropping does not depend on N loading (Figure 29b; i.e., low R^2) which is why the % metric in this analysis is useful and appropriate.

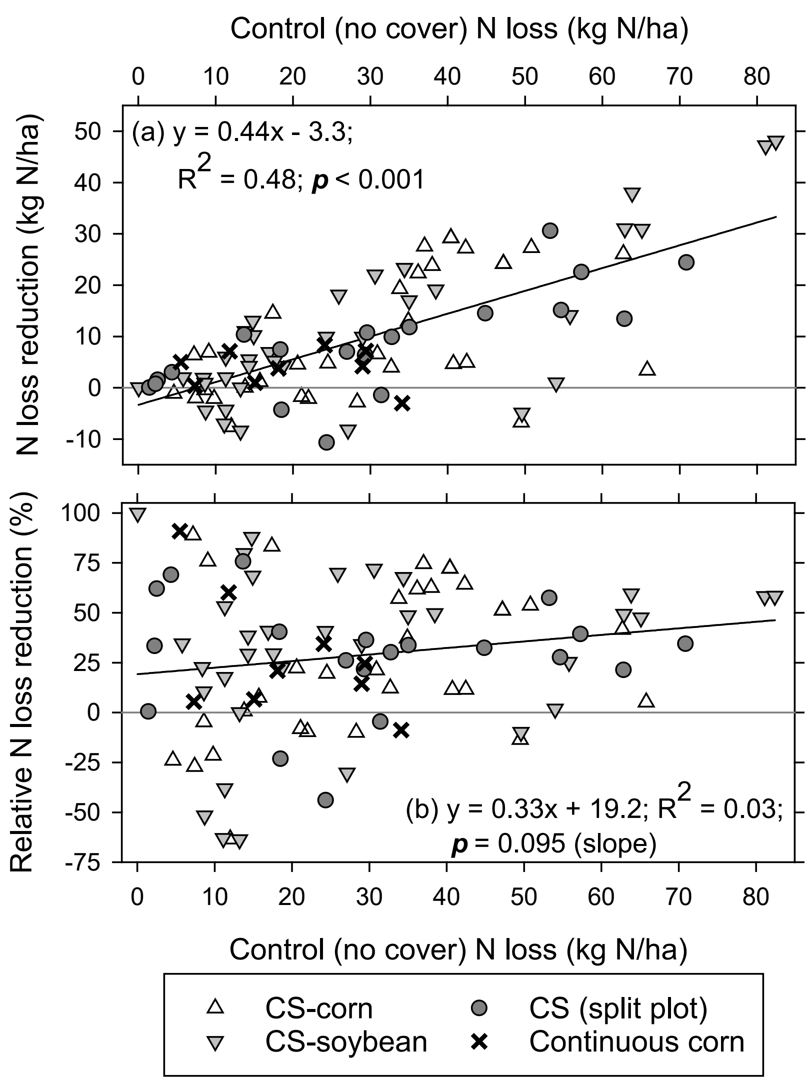


Figure 29. Annual N loss from the control (no cereal rye cover crop) versus the N loss reduction provided by the cereal rye cover crop (kg N/ha reduced, a; % reduction, b). The crop phases shown are the cash crop following the cereal rye cover crop. Data were sourced from studies listed in Table 14.

Nitrate leaching loss reduction provided by cover cropping is commonly reported to be due to reduction in drainage discharge combined with reduction in drainage nitrate concentrations (Christianson et al., 2016). However, many studies in this review did not observe a reduction in drainage discharge due to the cover crop (Aronsson and Torstensson, 1998; Askar et al., 2023; Fraser et al., 2013; Kaspar et al., 2012; Krueger et al., 2012; Logsdon et al., 2002; Nouri et al., 2022). While there were some exceptions (e.g., 11% discharge reduction by Strock et al. (2004)), the overarching result was that there was no difference in discharge between non-covered and cover crop plots (t-test $p = 0.903$, Figure 30a). In contrast, there was a strongly significant reduction in the annual flow-weighted nitrate concentrations due to a cover crop (t-test $p < 0.001$, Figure 30b). Askar et al. (2023) suggested that enhanced soil structure and root-created biopores can facilitate water movement to drainage systems, offsetting potential reductions in discharge.

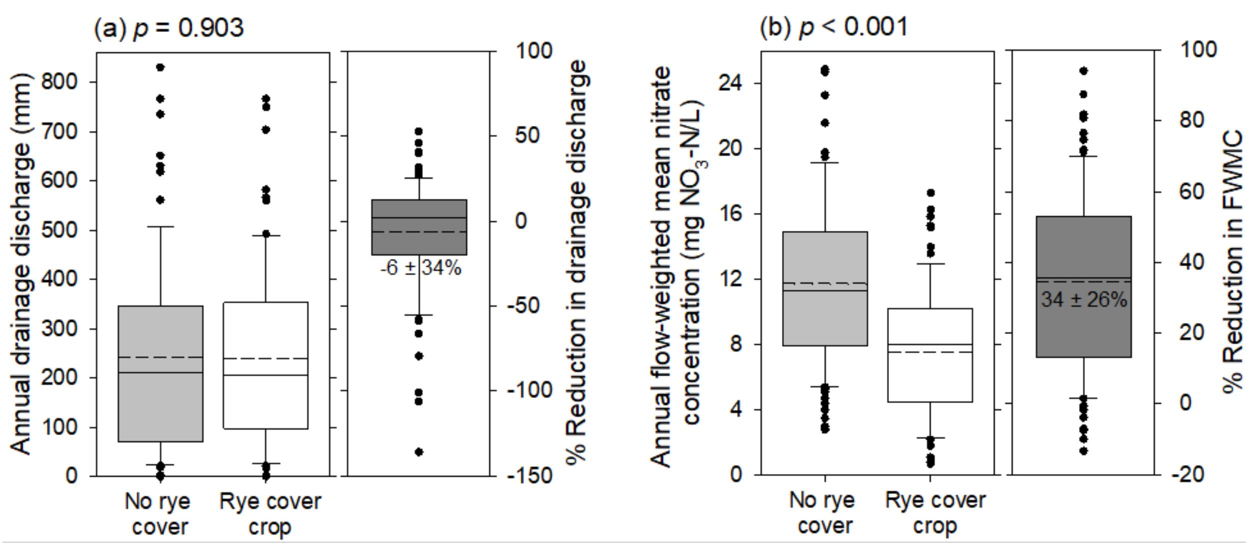


Figure 30. Ranges of annual drainage discharge (a; n = 79) and flow-weighted mean nitrate concentration (b; n = 97) from non-cover control and cereal rye cover crop treatments. Data are from only replicated tile plot studies performed in IA, IL, and MN and represent individual crop phases (i.e., not paired CS 2-y rotations). The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 14.

Cereal rye biomass and N uptake

Cover crop biomass is widely acknowledged to be one of the most important drivers of this practice's water quality benefit (Blanco-Canqui, 2018), thus biomass merits additional assessment in this water quality review. When corn for grain or soybeans were grown with a cereal rye cover crop in Minnesota (see Table 15), the average rye biomass produced was 0.85 ± 0.74 Mg/ha (Figure 31). The median rye biomass from these Minesota site-years was only 0.46 Mg/ha. This low median indicated that most of the time (i.e., the 50^{th} percentile), less than 0.50 Mg/ha would be produced by cereal rye in Minnesota. The mean biomass from the Minnesota studies was 33% lower than in studies from Iowa and Illinois (combining gray bars in Figure 31; 0.85 vs. 1.28 Mg/ha).

Deeper assessment of the Minnesota data confirmed that biomass can be increased when the cash crop is corn for silage rather than grain (Figure 32). Increased cereal rye biomass is also possible when this cover is grown ahead of soybeans versus corn for grain, although the sample size was small in this dataset ($n = 7$, Figure 32). The benefit of more rye biomass before soybeans has been previously observed in Iowa (Waring et al., 2020; Waring et al., 2024). These cash crop effects on rye biomass are relatively well known in Minnesota (e.g., Cates and Lewandowski, 2022) and are associated with management timing (see section *"Cereal rye planting and termination dates"*).

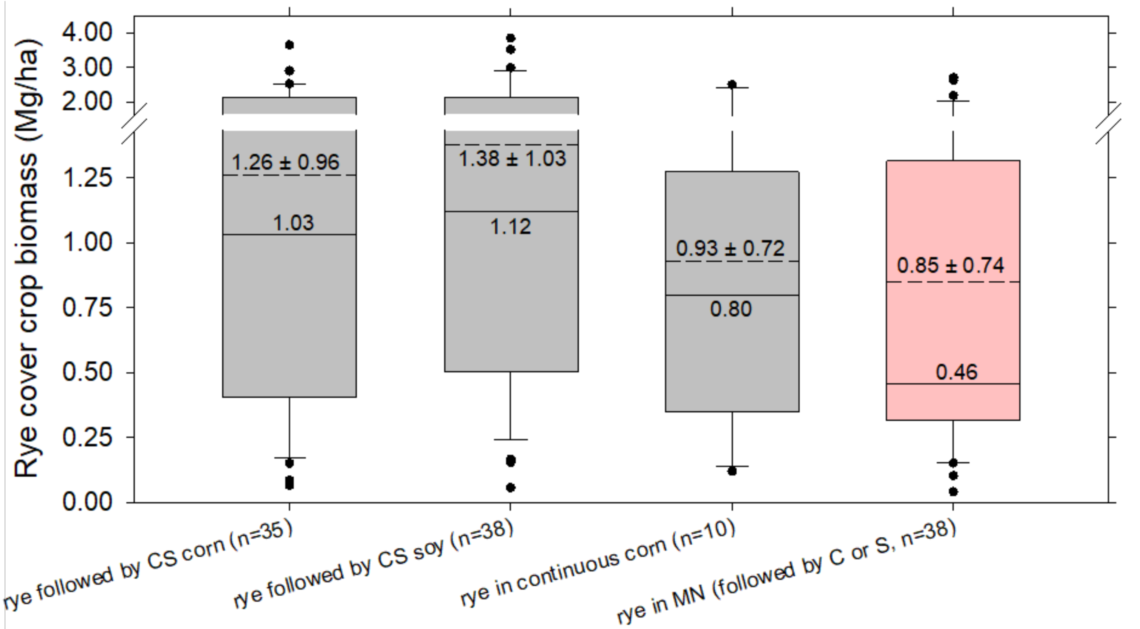


Figure 31. Ranges of winter cereal rye biomass by cash crop rotation phase from replicated tile drainage studies in Iowa and Illinois (gray bars) versus studies in Minnesota (pink bar). Note the y-axis break. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Minnesota data were sourced from studies listed in Table 15.

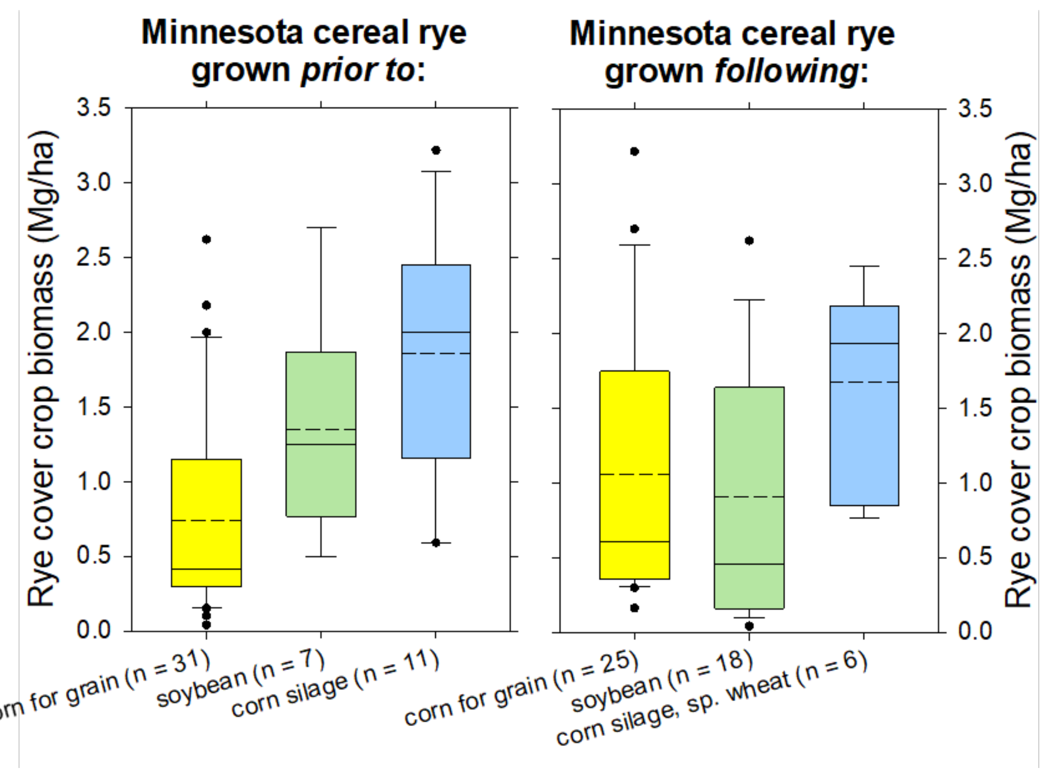
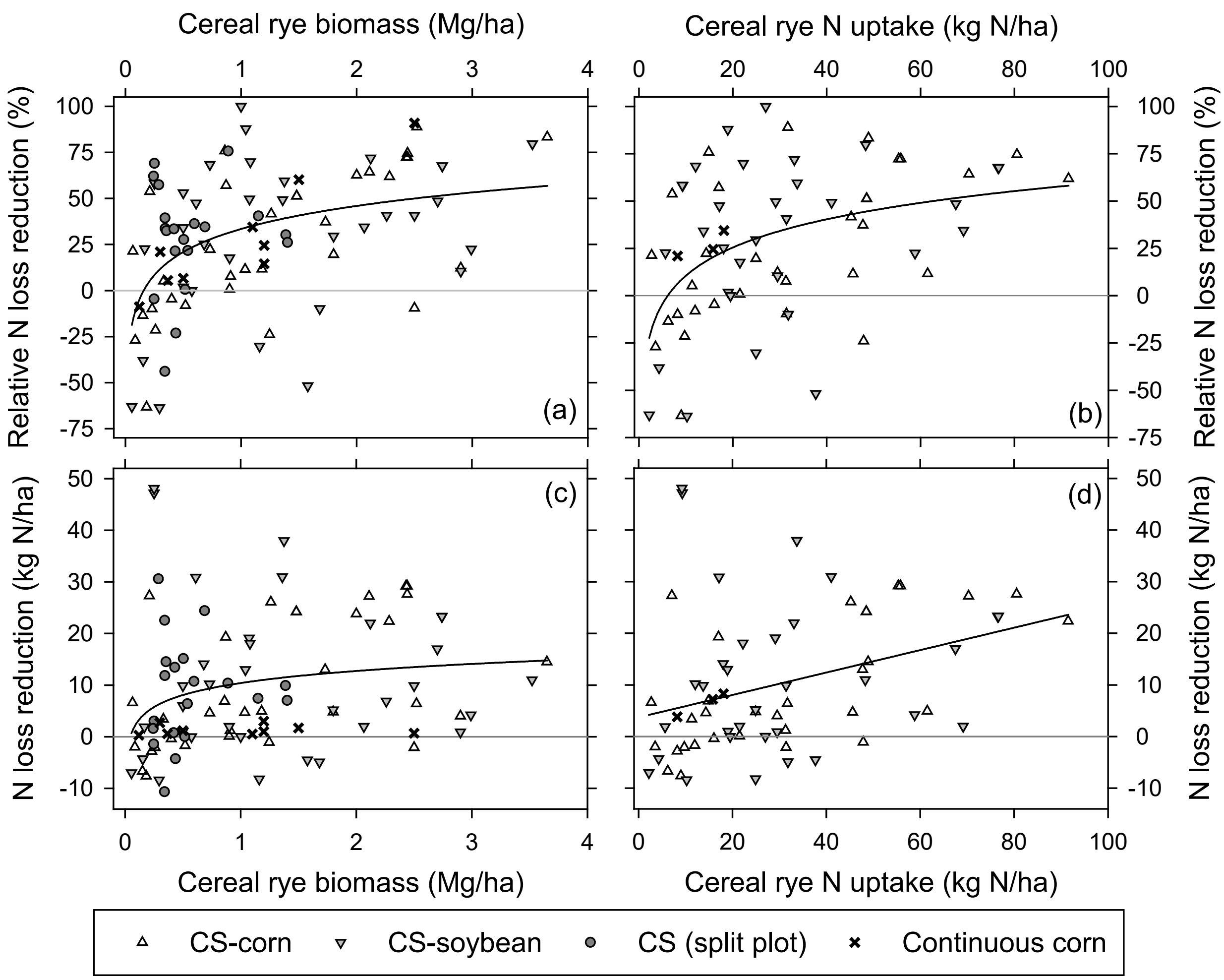


Figure 32. Ranges of cereal rye biomass reported in Minnesota studies sorted by if the cereal rye was grown prior to (left) or following (right) corn for grain, soybeans, or corn for silage and spring wheat. Note the left two bars in the left panel combine into the pink bar shown in Figure 31 above. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers.

Linking cereal rye growth parameters and water quality across this review's multi-state dataset showed biomass was more strongly correlated with relative N reduction efficiency expressed as a percent than it was with reduction in units of N loss (R^2: 0.23 vs. 0.07; Figure 33a and c). Comparison of regression models for these data showed logarithmic relationships provided the strongest fit, with the exception of panel d which was best modeled using a linear relationship. The logarithmic relationship between biomass and N leaching reduction efficiency in panel a was similar to that reported by Christianson et al. (2021) in a recent review. In a meta-analysis, Thapa et al. (2018) modeled the impact of biomass on N leaching reduction efficiency using a quadratic relationship which peaked at approximately 6 Mg/ha. That high level of biomass was predicted to provide nearly 80% N leaching reduction in that earlier work, but such high biomass was not observed in the current dataset.



(a) y = 18.1*ln(x) + 33.4; R^2 = 0.225; ***p*** < 0.0001 (b) y = 21.5*ln(x) - 38.8; R^2 = 0.217; ***p*** < 0.0001

(c) y = 3.36*ln(x) +10.4; R^2 = 0.070; ***p*** < 0.0001 (d) y = 0.217x + 3.71; R^2 = 0.126; ***p*** < 0.0001

Figure 33. Nitrate leaching reduction expressed as a relative percent efficiency (top panels) or as kg N/ha reduced (bottom panels) as a function of cereal rye biomass (left panels) and N uptake (right panels). The crop phases shown are the cash crop following the rye cover crop. Regression equations for all the data in each panel are below the legend. Data are from only replicated tile plot studies performed in IA, IL, and MN. Data were sourced from studies listed in Table 14.

Rogovska et al. (2023) reported a linear reduction of 5.7 kg N/ha leached was achieved for every Mg/ha of rye biomass produced on their replicated plots in Iowa. However, the logarithmic relationship in Figure 33c indicated there was a non-linearity across this larger dataset. More specifically, gains in biomass at the low end of biomass production provided relatively more benefit for water quality. For example, based on the relationship in Figure 33c, the ability to increase biomass from 0.25 to 1.25 Mg/ha would provide an incremental improvement to N leaching reduction of 5.5 kg N/ha for that one unit increase in biomass. Increasing biomass from 1.0 to 2.0 Mg/ha, which is also a one unit increase, would only provide an additional 2.4 kg N/ha reduction in N leaching beyond that achieved at 1.0 Mg/ha of biomass. This indicated there may be a minimum desirable level of cover crop biomass recommended to achieve desired water quality outcomes.

Beyond such nuances, low biomass production (e.g., < 0.5 Mg/ha) presented a risk of increased N leaching (Figure 33a and c). This finding was consistent with (but goes further than) the observation by Bawa et al. (2023) that at least 0.3 Mg/ha of cereal rye biomass was needed to provide N leaching reduction benefits in South Dakota. Nineteen of the 95 datapoints resulted in increased N leaching under a cover crop treatment (20% of datapoints), and the average increase in N leaching was 4.4 ± 2.9 kg N/ha (or, a -4.4 kg N/ha loss reduction; n = 19). The potential for increased N leaching while cover cropping was previously observed by Aronsson and Torstensson (1998) who reported an increase of 7 kg N/ha leached was related to poor establishment of the catch crop combined with mineralization of the preceding catch crops residues. Cates et al. (2018) highlighted that additional understanding is needed about how low cover crop biomass levels impact accrual of environmental benefits. Understanding the nuances and limits of N leaching reduction when cover crop biomass is low is especially relevant in Minnesota.

Not surprisingly, cereal rye biomass and N uptake were strongly correlated across this dataset because N uptake is a function of biomass (R^2: 0.61, Figure 34). These data reiterated the finding from Wilson et al. (2013) that at least 0.30 Mg/ha biomass was necessary to uptake 10 kg N/ha. However, extrapolating N uptake further to N leaching reduction introduces additional uncertainty given the low explanatory power of the N uptake/N loss dataset (Figure 33d, R^2: 0.126). In other words, cereal rye N uptake was not a robust proxy for N leaching reduction in these data. While there was a weak correlation between N uptake and kg N/ha reduced, the slope illustrated this relationship was certainly not one-to-one. That is, for every additional 10 kg N/ha taken up by a rye cover crop, a reduction in leaching of only 2.2 kg N/ha was achieved (Figure 33d slope).

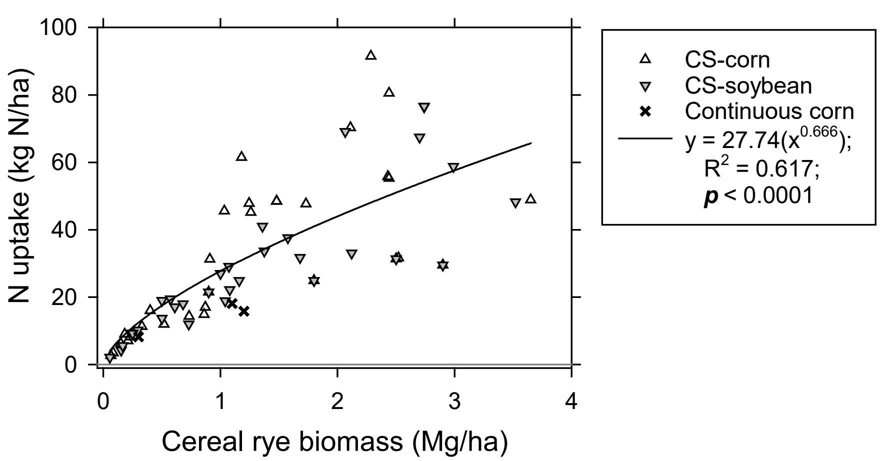


Figure 34. Cereal rye biomass versus N uptake in water quality studies represented in Figure 33 above.

Cereal rye planting and termination dates

The importance of cover crop planting date to maximize N leaching benefits has been known for several decades (Fisher et al., 2011; Ritter et al., 1998; Shepherd and Webb, 1999; Thapa et al., 2018). However, in Midwestern cropping systems, management of the cash crop often dictates the available window for cover crop biomass production (e.g., Figure 32). In Minnesota, cover crop planting is generally recommended to occur by September 1 (244 day of year, DOY; (Moncada and Sheaffer, 2010), although mid-August to early November (222 to 312 DOY) may be the window used in practice (Everett et al., 2019; Garcia y Garcia et al., 2021). Everett et al. (2019) reported a strong relationship between cover crop planting date and biomass in Minnesota (R^2: 0.58) with a 0.4 Mg/ha biomass decrease for every week cover crop planting was delayed. The current dataset (which included data from that earlier work) did not exhibit a correlation between planting date and cereal rye biomass (R^2: 0.03; data not shown).

Given this lack of direct correlation, the Minnesota cereal rye biomass data were binned by quartiles to assess differences in planting and termination dates based on relatively "low" versus "high" biomass production. The lowest biomass quartile had the latest planting dates averaging 276 DOY versus means of 265-268 DOY for when biomass was ≥ 0.42 Mg/ha (dashed lines on white versus green bins in Figure 35a). Comparing termination dates across biomass quartiles, the lowest biomass (Q1 and Q2) had mean termination dates of 114 and 111 DOY versus a mean of 119 DOY for Q3 and Q4 (Figure 35b). In other words, the lowest biomass site-years tended to be planted just over week later and terminated about a week earlier than the higher-biomass site-years.

The impact of these two timing factors combined into a slightly stronger effect when they were assessed together as total "*days of growth*" (Figure 35c). More specifically, while the effect of neither planting date nor termination date was especially strong (ANOVA $p > 0.34$), the "*days of growth*" provided evidence of the importance of small management changes at both ends of the cover crop window (albeit weak evidence: ANOVA $p = 0.130$). The mean total days of growth from the lowest to highest biomass quartiles were 202, 211, 216, and 219 days, respectively. This meant that relatively small changes (e.g., one week) with earlier planting dates and with later termination dates could make a difference for cereal rye biomass production in Minnesota. Air and soil temperature obviously also factor into cover crop biomass growth, but few studies reported growing degree days, for example.

The average planting and termination dates across this Minnesota biomass dataset were September 26 (269 ± 17 DOY) and April 26 (116 ± 13 DOY), respectively. Modelling by Feyereisen et al. (2006) provided the most comprehensive assessment of how these cover crop management dates could impact N leaching reduction in Minnesota. Extrapolating from their assessment showed that planting by October 1 would provide more than 1.0 Mg/ha cereal rye biomass if it was terminated no earlier than April 26. Approximately 1.0 Mg/ha of biomass would result in an estimated 33% N leaching reduction based on the logarithmic relationship in the current work (Figure 33a). However, Feyereisen et al. (2006) predicted this combination of management dates would only provide approximately 15% N leaching reduction. They recommended that to achieve a 30% N leaching reduction, a cover crop planted by October 1 should not be terminated until the end of May. This difference in predictions reiterates an earlier call for additional studies of cover cropping to be performed using replicated tile drainage plots to better calibrate understanding of cover crops in Minnesota.

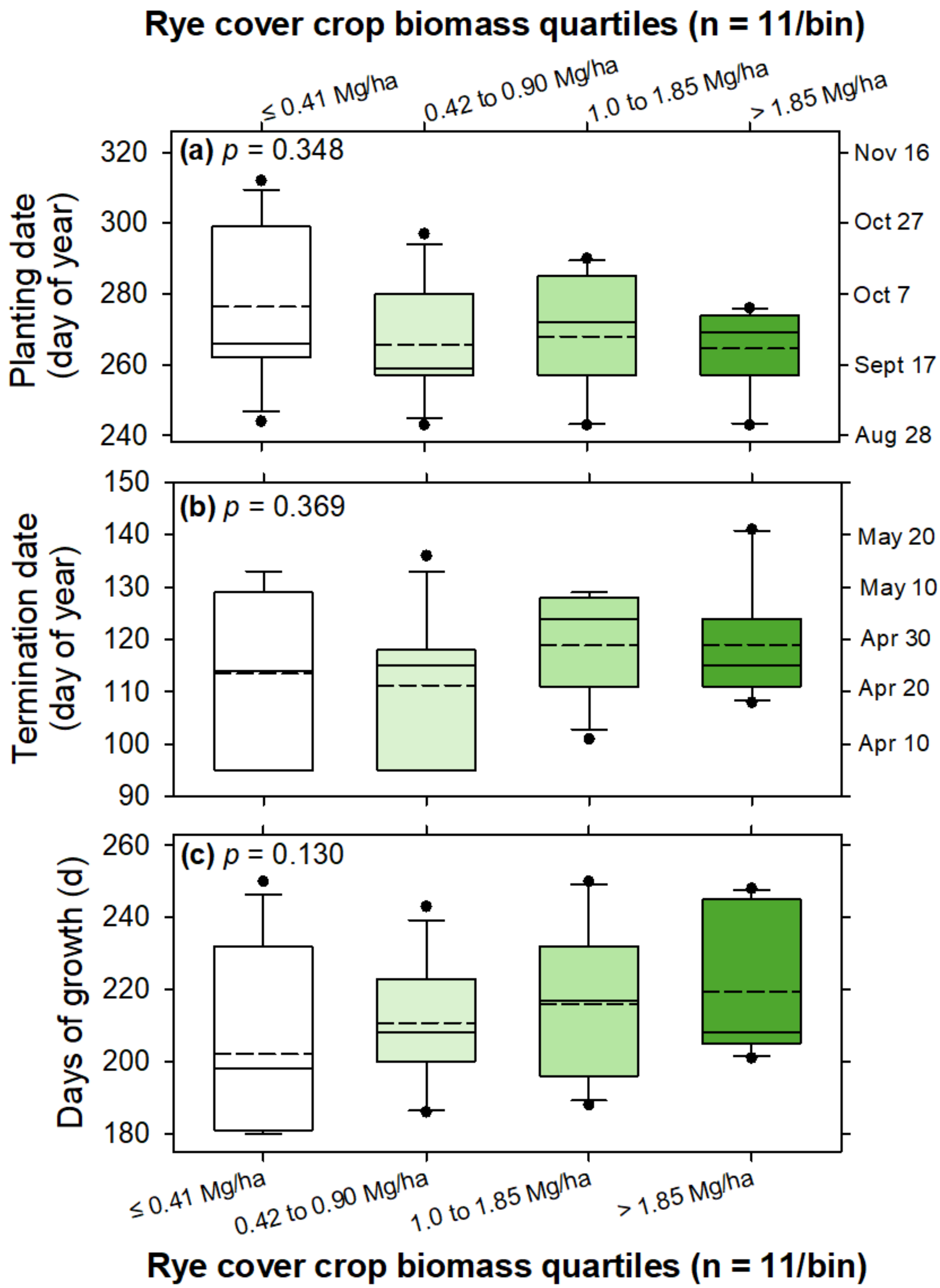


Figure 35. Ranges of cereal rye biomass in Minnesota from 44 site-years, binned by quartiles (n = 11 site-years per bin) to assess the ranges of planting (a) and termination (b) dates and days of growth (c) for relatively "low" (Q1) versus "high" (Q4) biomass production. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 15.

Finally, while not assessed here, it is also important to note that the timeliness of appropriate rainfall, especially during the establishment period, is a significant driver of cover crop biomass production (Dabney et al., 2001; Kaspar et al., 2012; Meisinger and Ricigliano, 2017; Ritter et al., 1998; Thapa et al., 2018; Wilson et al., 2013). Cover crop seeding method is also worth mentioning, but Wilson et al. (2019) noted that research should focus on seeding rates and timing rather than method to maximize cover crop benefits. It is generally accepted that planting methods that ensure contact between the seed and the soil are the best (drilling preferred; broadcast plus incorporation is also ok; Moncada and Sheaffer, 2010). Undersown (i.e., overseeding, interseeding) cover crops merit special discussion in cold climates (see section "*Cover crops in short season cropping systems*").

Cereal rye cover crop impacts on a cash crop yield

A corn-soybean rotation with cereal rye resulted in an average yield penalty of 4.5 ± 9% for corn (0.6 ± 1.0 Mg/ha reduction) and 4.6 ± 5% for soybean (0.2 ± 0.2 Mg/ha reduction; Figure 36). The sample size was small for continuous corn, but there was a mean yield penalty of 3.3 ± 12% (0.3 ± 1.2 Mg/ha reduction). None of these yield reductions associated with cover cropping were strongly statistically significant (t-test p values ≥ 0.050; Figure 36), but an average corn yield reduction of 0.3 Mg/ha (i.e., 4.8 bu corn/ac), for example, may not be insignificant to a farmer.

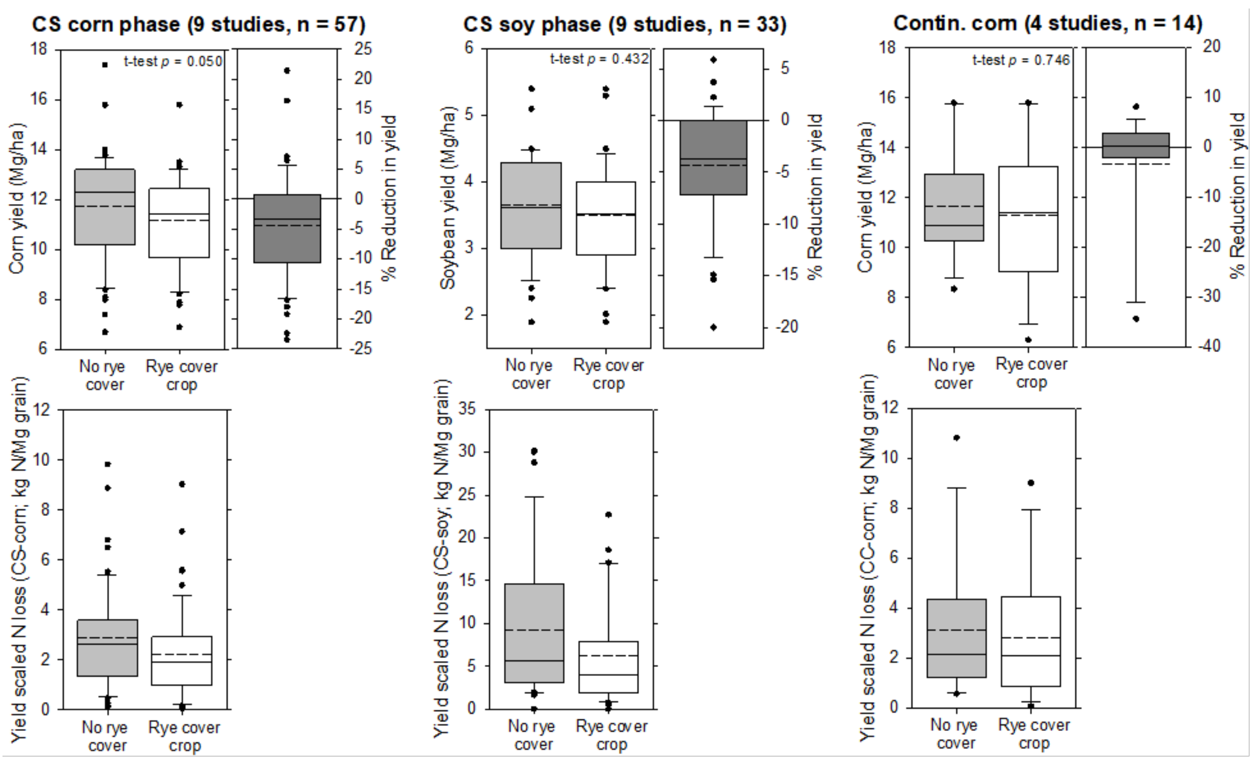


Figure 36. Ranges of corn and soybean yield impact and yield-scaled N losses due to a cereal rye cover crop. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. "CS" refers to corn-soybean rotation. Data were sourced from studies listed in Table 14.

Reported corn grain yield reductions associated with the practice of cover cropping were often found during the course of this review (Adler et al., 2020; Kaspar et al., 2007; Martinez-Feria et al., 2016; Rogovska et al., 2023; Ruffatti et al., 2019). However, corn grain increases (Coelho et al., 2005; Drury et al., 2014) or no significant impact on corn yield due to a cover crop (Everett et al., 2019; Gentry et al., 2024; Preza-Fontes et al., 2021; Rasse et al., 2000; Waring et al., 2023) have also been reported. The impact of cover cropping on soybean yield is similarly variable. Studies have reported cover cropping can lead to soybean yield penalties (Eckert, 1988; Kaspar et al., 2012; Waring et al., 2023), yield increases (Drury et al., 2014), no impact on yield (Ruffatti et al., 2019), or both increases and decreases (Adler et al., 2020). In addition to corn and soybean, a recent three-year study conducted in Minnesota's Red River Valley region demonstrated that interseeding cover crops such as rye can protect the soil without adversely affecting sugar beet root yield or sucrose concentration (Sigdel et al., 2021). However, the impact of this strategy on water quality was not assessed to confirm its potential environmental benefits.

Accordingly, review articles assessing the impact of cover crops on cash crop yield are inconsistent. Abdalla et al. (2019) reported a primary crop yield penalty of 4% due to cover cropping, which was very similar to that found in the current work. However, reviews by Thapa et al. (2018) and Miguez and Bollero (2005) reported non-legume cover crops had no impact on the main crop. Yet another recent review noted there was generally an increase in cash crop yield due to cover cropping (Daryanto et al., 2018). This variability across individual studies and across reviews highlights the importance of performing quantitative reviews where data extraction and assessment can be tailored to best reflect specific cropping systems, management approaches, and climates (e.g., tile drained corn-soybean rotations in the upper Midwest).

Despite the average yield penalty calculated here, inclusion of a cereal rye cover crop improved yield-scaled N losses in the three scenarios (Figure 36, bottom panels). For example, the corn phase in a corn-soybean rotation leached a mean of 2.9 kg N per Mg grain produced without a rye cover crop but only leached 2.3 kg N per Mg grain when a rye cover crop was present. For the soybean phase, these means were even more notably different at 9.4 vs. 6.3 kg N/Mg grain for the no-cover vs. rye cover crop, respectively. This agri-environmental metric gives an indication of the environmental efficiency of each unit of grain produced and is a useful concept linking agronomic and environmental goals (Zhao et al., 2016).

Non-rye cover crops, mixed cropping systems, and nitrate leaching

Oats as a cover crop

Eleven studies reported oat as a cover crop but only four of those provided relevant, extractable water quality data (Table 16; Kaspar et al., 2012; O'Brien et al., 2022; Singh et al., 2018; Waring et al., 2023). Of those, only two provided appropriate annual N leaching losses for the use of oat as a cover crop in a corn-soybean rotation. These two studies reported a total of 9 crop phase-years but these data paired into only four 2-y rotation site-years. Thus, two studies from the same set of replicated tile drainage plots in Iowa provided all of the N leaching reduction data for this practice of an oat cover crop (Kaspar et al., 2012; O'Brien et al., 2022).

The use of an oat cover crop in a corn-soybean rotation provided a mean 2-y rotation average of 23 ± 23% N leaching reduction (data not shown; n = 4). The Iowa Nutrient Reduction Strategy originally recommended a value of 28 ± 2% for an oat cover crop (IDALS, 2014), and this was revised to 25% ± 6% in 2024 (IDALS, 2024). Given the same data are presumably available to generate these values, the reason for the slight difference is unclear (23 vs. 25%). Nevertheless, because an oat cover crop winter-kills, it is questionable if a N leaching reduction of 23% would be appropriate for Minnesota which has a colder climate than Iowa where these studies were performed. Given a N leaching reduction of only 20% for winter hardy cereal rye is being recommended, additional research on oats as a cover crop needs to be performed using replicated tile drainage plots in Minnesota before a defensible value can be developed.

Table 16. Studies reporting measures of annual water quality impacts due to an oat cover crop. Note a "site-year" for a corn-soybean rotation was a 2-y rotation average.

Study	**# site-years**	**Cash crop**	**Cover crop**	**Location**	**Method and notes;**
Kaspar et al. (2012)	2	Corn-soybean	Oat	Iowa	Replicated tile plots; Corn and soybean phases reported separately
O'Brien et al. (2022)	2	Corn-soybean	Oat	Iowa	Replicated tile plots; Corn and soybean phases reported separately
Waring et al. (2023) *	NA (conc. only)	Corn-soybean	Oat	Iowa	Suction lysimeter at 61 cm
Singh et al. (2018) *	1	CS-corn	Oat + radish	Illinois	Pan lysimeters at 23-30 cm
Plaza-Bonilla et al. (2015) *	NA	Sorghum, sunflower, wheat, peas	Vetch-oat	France	Modelling study (STICS model)
Lozier et al. (2017) *	NA	Corn-soybean-wheat	Oat clippings	Ontario	Runoff plot study for P leaching;
Logsdon et al. (2002) *	NA	NA	Oat	Iowa	Soil monolith lysimeters; confidence intervals rather than means reported
Kovar et al. (2011) *	NA	Corn-soybean	Oat	Iowa	Rainfall simulations (2 events; not annual)
Jones (1942) *	NA	Soybeans, cowpea, crotalaria	Oat	Alabama	30" x 30" repacked soil columns (not a field study)
Cambardella et al. (2010) *	NA	Corn-soybean	Oat	Iowa	Soil nitrate only
Aryal et al. (2018)	NA	Cotton or corn	Oat	Arkansas	Mixed experimental design for oat

* not used in recommended average for oat cover cropping

Cover crops in short season cropping systems

The original Minnesota Nutrient Reduction Strategy assigned a 51% N leaching reduction to the practice of cover cropping in a short season cropping system. These systems typically involve crops with shorter growing periods, such as sweet corn, peas, edible beans, small grains, and corn silage, which allow for earlier harvests and more time for a cover crop to establish in the fall. Thus, a short season assessment was relevant although the type of cover crop studied in these cropping systems varied. There were 79 N leaching site-years from twelve studies of cover crops used in cash cropping rotations that were not corn grain-based (i.e., the crop following the cover was not CS-corn, CS-soybeans, or continuous corn; Figure 37b). This included seven general cover crop types (Figure 37a) that resulted in a N leaching reduction of 48 ± 65% across all locations (Figure 38a).

Only 42 of the 79 mixed cover crop site-years were from a cold climate, which was defined as being from a latitude greater than 43° N (i.e., studies from MN, OR, NY, Sweden, northern France; Table 17). Many of the cold climate site-years were from Sweden (21 of 42) and 18 of those reported an undersown ryegrass cover crop where the cover was planted at the time of the cash crop planting. Accounting for these differences, cover cropping in a non-corn grain-based rotation in a cold climate provided 39 ± 26% N leaching reduction when not undersown and 51 ± 28% when the cover was undersown (Figure 38b and c).

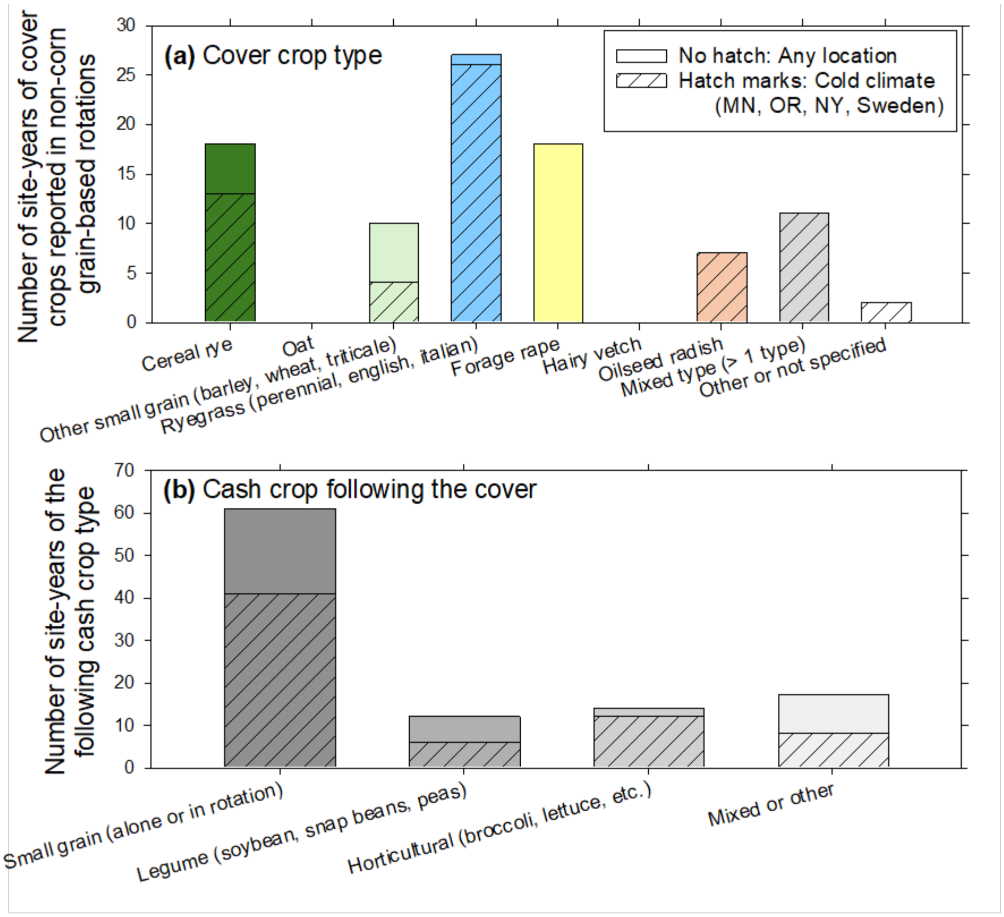


Figure 37. Count of site-years for cover crops used in relatively short season cropping systems (i.e., not corn grain-based) across both leaching and runoff studies. The cover crop types are shown in (a) with the following cash crop in (b) for all locations versus cold climate studies. Cold climate was defined as a latitude > 43°N.

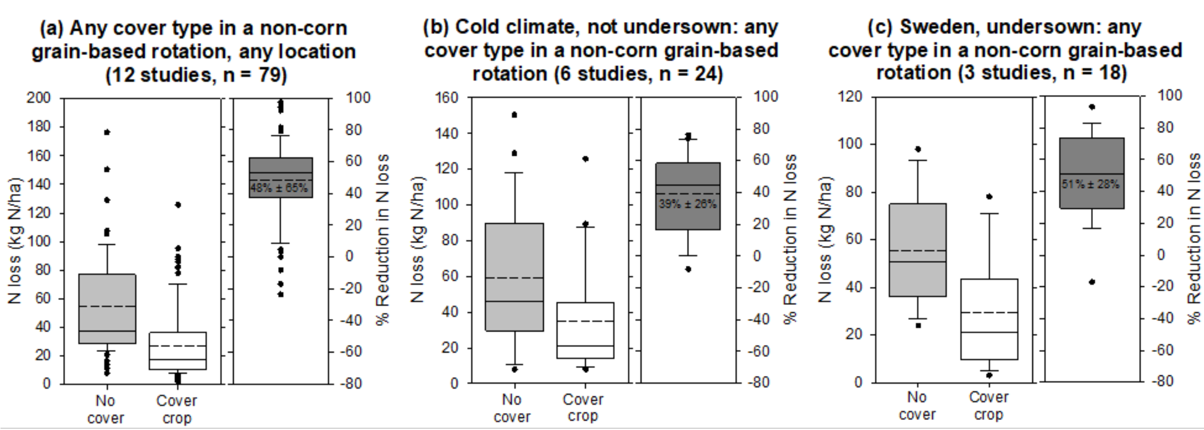


Figure 38. Ranges of nitrate leaching loss and loss reduction due cover cropping (any cover type) in non-corn grain-based cash crop rotations. Results are shown across all locations (a), only cold climates where the cover was not undersown (b) and from Sweden where the cover was undersown at the time of cash crop planting (c). The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 17.

It is widely accepted that achieving desirable outcomes with cover cropping is increasingly difficult in cooler climates (Malone et al., 2014; Miguez and Bollero, 2005), and this cannot be underemphasized for this Minnesota-focused review. Christianson et al. (2021) illustrated that N loss reduction decreased by 8% for every 1° increase in latitude. For example, based on this latitude relationship alone, cover cropping in Ames, Iowa (42° N) would be able to achieve 20% N leaching reduction, whereas in Waseca, Minnesota (44° N), cover cropping would only be able to achieve 4% N loss reduction over the long term average. This is relatively consistent with the finding by both Strock et al. (2004) and Wilson et al. (2014) that cover crop establishment in southern Minnesota would be successful in approximately one out of every four years.

Table 17. Cover crop studies reporting N leaching losses where a corn grain-based rotation was not the main cash crop rotation (i.e., short season cropping system). Cold climate was defined as ≥ 43° latitude.

Study	# site-years	Cash crop following the cover	Cover	Location	Method and notes
Krueger et al. (2013)	1	Corn silage	Cereal rye	Minnesota	Edge of field
Krueger et al. (2012)	2	Corn silage	Cereal rye	Minnesota	Suction lysimeter at 120 cm
Feaga et al. (2010)	11	Sweet corn, broccoli, or snap beans	Cereal rye, triticale, or common vetch + triticale	Oregon	Wick lysimeter
Griffith et al. (2020)	1	Corn silage	Cereal rye	New York	Replicated tile
Constantin et al. (2010)	6	Mixed or small grain	Ceral or radish; Italian ryegrass; white mustard	Northern France	Suction lysimeter at 90 or 110 cm
Torstensson and Aronsson (2000)	3	Spring barley	Winter rye	Sweden	Replicated tile
Torstensson and Aronsson (2000) *	12	Spring wheat or barley, oats, or potatoes	Perennial or Italian ryegrass (undersown)	Sweden	Replicated tile
Aronsson and Torstensson (1998) *	3	Oats or barley	Perennial ryegrass (undersown)	Sweden	Replicated tile
Lewan (1994) *	3	Oats, wheat, or barley	Italian ryegrass (undersown)	Sweden	Replicated tile
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; not truly paired
Meisinger and Ricigliano (2017) *	9	Buckwheat	Barley, wheat, or winter rye	Maryland (not cold climate)	Suction lysimeter at 100 cm; cover was fertilized
Heinrich et al. (2014) *	2	Lettuce	Cereal rye	California (not cold climate)	Suction lysimeter at 60 cm
Fraser et al. (2013) *	18	Wheat, pea, or barley	Forage rape	New Zealand (not cold climate)	Suction lysimeter at 60 cm

* not used in recommended average for cold climate short-season non-undersown cover cropping

The average 51% leaching reduction provided by undersown crop crops in Sweden (Figure 38c) was especially notable given the >56° N latitude of those studies (i.e., compared to Minnesota at roughly 43° to 49° N). This mean value reiterated the results of a Nordic meta-analysis that reported an average 50% N leaching reduction for undersown non-legume cover crops (Valkama et al., 2015). Under sowing, interseeding, and overseeding could be important methods to improve cover crop biomass growth and associated environmental benefits in other cold climates such as

Minnesota. While no studies included in this review specifically assessed cover crops following wheat in the Red River Valley, it is possible that earlier fall establishment after wheat harvest could lead to comparable N loss reductions. Myers-Bailey (2024) recently showed that interseeding a cover crop in Iowa reduced N leaching. However, this practice was more successful when corn was planted in twin-row spacing (two rows spaced 20 cm apart, with 130 cm between each set of twin rows) to allow better cover crop establishment. The wider spacing resulted in a corn yield penalty despite improved cover crop growth. Current data in Minnesota show interseeding can be riskier than post-harvest seeding for establishment of the cover (Cates and Lewandowski, 2022), so additional management guidance may be needed if high N leaching reductions are a prime objective of cover cropping.

In summary, these cold-climate data illustrated that cover crops can perform well for N leaching reduction (i.e., nearly 40% reduction or greater) in these challenging conditions. However, to achieve this, the cropping rotation and approach to cover crop management needs to differ from the current status quo in Minnesota.

Cover cropping and phosphorus leaching

The available P leaching data associated with cover cropping was much more limited than N leaching data (e.g., Figure 26a). So, corn and soybean phases of a corn-soybean rotation were not combined into a 2-y average for P loss as was done for N leaching. A strong rotation effect under a corn-soybean system was not expected for P dynamics, although it is acknowledged that P fertilizer is often only applied once during the 2-y rotation.

A second distinction from the above cereal rye N leaching assessment was that the limited data for P leaching losses and surface runoff nutrient losses meant only one nutrient loss efficiency could be developed per pathway. That is, P leaching and surface runoff nutrient losses were assessed for the bin "*cover cropping in general*" rather than being developed by cover type (Table 13). Even so, the cover crops assessed in the remainder of this review tended to be grass-based cereals (cereal rye, wheat, etc.).

Dissolved phosphorus leaching

Cover cropping in a corn-soybean or continuous corn system increased dissolved P leaching losses by 45 ± 86% (that is, a -45% leaching reduction; Figure 39 left panels). These studies were from Iowa, Illinois, and Ontario and were all performed using replicated tile drainage plots (Table 18). The median dissolved P impact was also negative (-31%) which provided additional evidence of this negative water quality impact.

Including three additional studies that used different cropping systems (silage, corn-soybean-wheat; Table 18) resulted in even greater leaching losses of dissolved P when a cover crop was grown (mean -61%; Figure 39 right panels). The edge-of-field study by Hanrahan et al. (2021) in Ohio merits particular discussion. This work was performed using 40 paired fields across 20 locations in Ohio over eight years. Median annual nutrient losses (DRP, TP, nitrate, TN) for the cover cropped versus non-cover fields for both surface runoff and tile drainage were reported for all eight years. Thus, the results represented much more than the 8 site-years used here (i.e., results represented more than just 1 site x 8 y). This was especially notable given the annual tile drainage DRP and total P losses increased by averages of 126 and 63% for the cover cropped fields in their study. This one study accounts for most of the difference between the left and right panels in Figure 39, and is one of the most comprehensive field-scale assessments of the water quality impacts of cover cropping to date.

The state of Iowa does not differentiate between dissolved and total P in their Nutrient Reduction Strategy and the recent review in the state of Indiana did not result in P leaching loss recommendations for cover cropping. A cover crop review by Christianson et al. (2021) only reported 4 site-years for P leaching losses (all from Daigh et al., 2015). Nevertheless, there is a notable body of work assessing the potential for cover crops to release dissolved P from biomass during freeze-thaw periods (Cober et al., 2019; Liu et al., 2019a; Liu et al., 2014; Muñoz-Ventura et al., 2022). Despite the lack of findings in previous science assessments and reviews, the data compiled here indicate this nutrient loss pathway could be important for Minnesota's cold climate. Tradeoffs associated with P releases are more fully contextualized below in the section "*Cover crop P impacts in above- vs. below-ground pathways*".

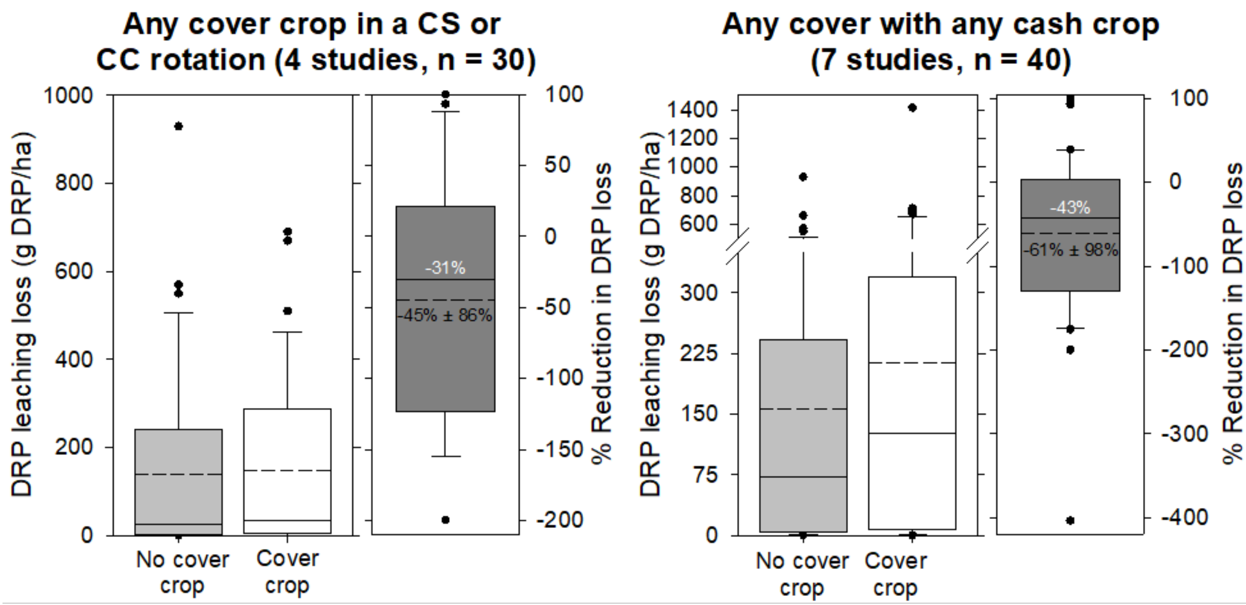


Figure 39. Ranges of dissolved phosphorus (generally, dissolved reactive phosphorus, DRP) annual leaching losses with and without cover crops. CS, corn-soybean rotation; CC, continuous corn. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. A negative "% Reduction…" indicated the practice increased losses of the nutrient. Data were sourced from studies listed in Table 18.

Table 18. Cover crop studies reporting dissolved phosphorus annual leaching losses (generally DRP).

Study	**# site-years**	**Cash crop following the cover**	**Cover**	**Location**	**Method and notes**
Dougherty et al. (2020)	7	CS-corn	Cereal rye	Iowa	Replicated tile
Dougherty et al. (2020)	7	CS-soybean	Cereal rye	Iowa	Replicated tile
Zhang et al. (2017)	4	CS-corn	Wheat	Ontario	Replicated tile
Zhang et al. (2017)	4	CS- soybean	Wheat	Ontario	Replicated tile
Daigh et al. (2015)	4	Continuous corn	Cereal rye	Iowa	Replicated tile; TRP reported
Alves de Oliveira et al. (2022)	4	Continuous corn	Cereal rye	Illinois	Replicated tile
Krueger et al. (2013) *	1	Corn silage	Cereal rye	Minnesota	Edge of field; not truly paired
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; multiple sites reported as one
Griffith et al. (2020) *	1	Corn silage	Cereal rye	New York	Replicated tile

* not used in recommended average for corn grain-based rotations

Total phosphorus leaching

Cover cropping in a corn-soybean rotation had a negative impact on total P leaching losses but this assessment was only based on the one available study (Table 19; Zhang et al., 2017). The mean (-13 ± 43%) and median (-7%) total P loss reduction values were both negative from these relatively limited 8 site-years (Figure 40, left panels). Including additional site-years from more diverse cropping systems provided supporting evidence for the possibility of this negative leaching impact (mean: -12 ± 41%; median: -6%; Figure 40, right panels).

Despite the limited data in this specific review, the overall recommendation was supported by broader results from Nordic studies that are relevant for Minnesota's climate. Bergström et al. (2015) reported cover crops were not an effective practice to reduce total P leaching in Sweden and Aronsson et al. (2016) found this practice was not effective for total P in either leachate or surface runoff in southern Scandinavia. Cover crop leaching tradeoffs between N (reduction) and P (increase) highlight the increasing importance of appropriately balancing water quality resource concerns as well as appropriate mitigation activities (Aronsson et al., 2016; Askar et al., 2023).

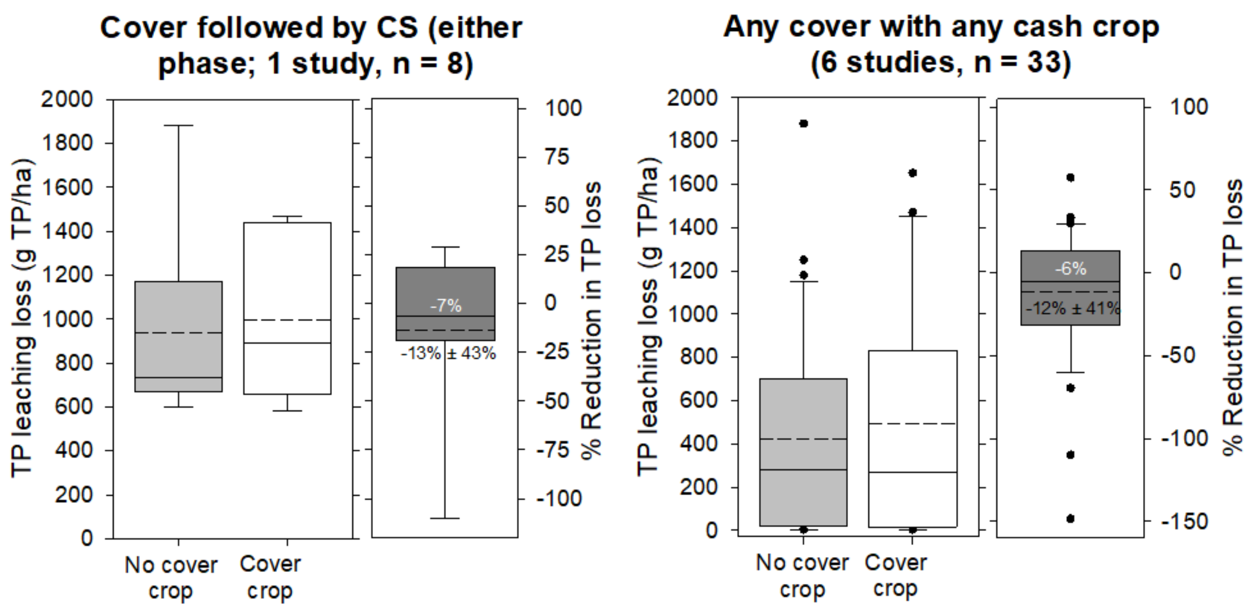


Figure 40. Ranges of total phosphorus (TP) annual leaching losses with and without cover crops. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. A negative "% Reduction…" indicated the practice increased losses of the nutrient. Data were sourced from studies listed in Table 19.

Table 19. Cover crop studies reporting total phosphorus (TP) annual leaching losses.

Study	# site-years	Cash crop following the cover	Cover	Location	Method and notes
Zhang et al. (2017)	4	CS-corn	Wheat	Ontario	Replicated tile
Zhang et al. (2017)	4	CS-soybean	Wheat	Ontario	Replicated tile
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; multiple sites reported as one; one outlier removed
Griffith et al. (2020) *	1	Corn silage	Cereal rye	New York	Replicated tile
Norberg and Aronsson (2020) *	12	Barley	Oilseed radish; hairy vetch + rye; buckwheat + radish	Sweden	Replicated tile; Cover only in the fall
Aronsson et al. (2011) *	2	Barley; oats	Perennial ryegrass	Sweden	Replicated tile;
Liu et al. (2012) *	3	Spring cereals	Not specified	Sweden	Replicated tile; 15 study-years reported as one

* not used in recommended average for corn grain-based rotations

Cover cropping and surface runoff nutrient loss

Successful cover cropping reduces runoff and erosion which plays an important role in reducing sediment loss (Blanco-Canqui, 2018; Kaspar and Singer, 2011). However, the impact of cover crops on nutrients in surface runoff is more variable and can be relatively uncertain in areas where a large portion of annual runoff is due to snowmelt (Liu et al., 2019a). A review by Blanco-Canqui (2018) noted that cover crops reduced nitrate leaching in 90% of studies and sediment loss in 65% of studies, but only reduced losses of soluble nutrients in runoff less than 25% of the time.

Nitrate in surface runoff

In this review, cover cropping in a corn-soybean system provided a 16 ± 76% reduction of nitrate losses in surface runoff (Figure 41 left panels). Nitrate losses in surface runoff tend to be lower than in tile drainage. However, the range of surface runoff N loss reduction that cover cropping provided of 3.1 ± 6.4 kg N/ha notably overlapped with that provided for tile drainage (8.3 ± 14 kg N/ha; Figure 27).

There is some uncertainty in how these values may apply to Minnesota given the studies were from Ontario and Missouri (Table 20). A surface runoff study performed in Morris, Minnesota that evaluated winter rye and forage radish cover crops was included in this review, but there was some difficulty in appropriately extracting the nutrient losses (Weyers et al., 2021; unclear axes, personal communication initiated). Nevertheless, including additional site-years from more diverse cropping systems from Ohio and New York state confirmed the benefit for nitrate in surface runoff (mean: 11 ± 72%; Figure 41 right panels). A few studies reported the potential for cover cropping to increase soil nitrate levels after termination (Jewett and Thelen, 2007; Wyland et al., 1996) but this effect seemingly did not impact the extracted surface runoff nitrate data.

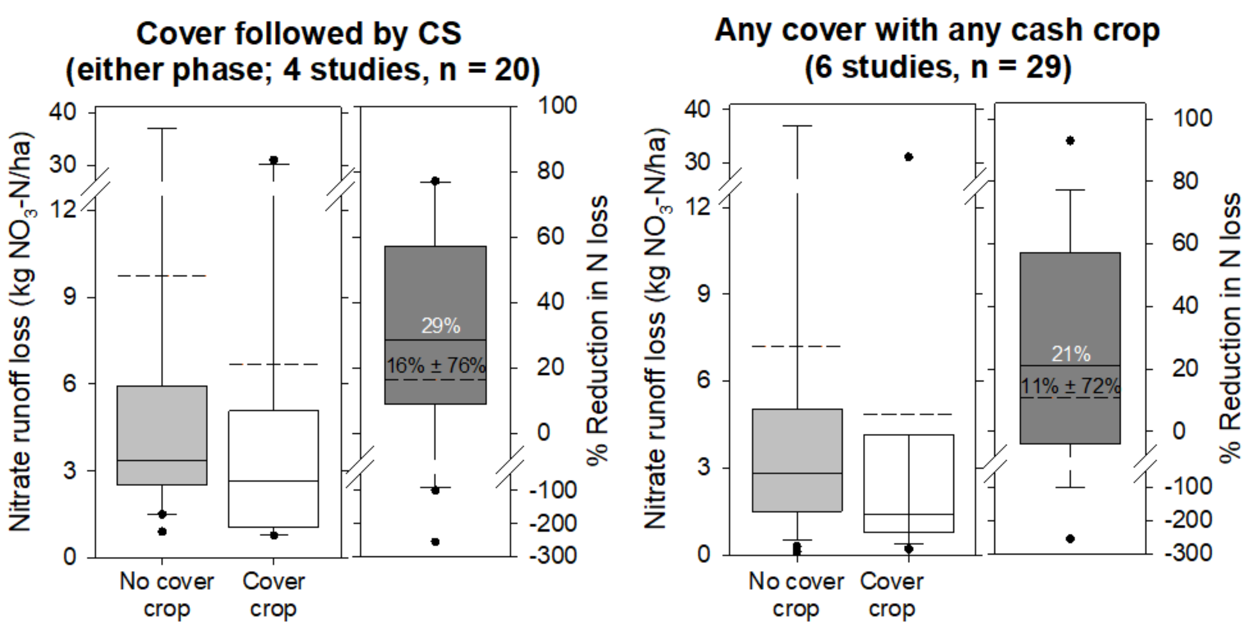


Figure 41.Ranges of nitrate annual surface runoff losses with and without cover crops. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 20.

Table 20. Cover crop studies reporting annual nitrate surface runoff losses from natural rainfall (not rainfall simulation).

Study	# site-years	Cash crop following the cover	Cover	Location	Method and notes
Kaur et al. (2024)	2	CS-corn	Wheat, radish, turnip	Missouri	Terrace tiles draining surface runoff
Drury et al. (2014)	4	CS-corn	Wheat	Ontario	0.75% average slope
Adler et al. (2020)	1	CS-corn	Wheat, radish, turnip	Missouri	Terrace tiles draining surface runoff
Kaur et al. (2024)	2	CS-soybean	Cereal rye	Missouri	Terrace tiles draining surface runoff
Drury et al. (2014)	6	CS-soybean	Wheat	Ontario	0.75% average slope
Adler et al. (2020)	1	CS-soybean	Cereal rye	Missouri	Terrace tiles draining surface runoff
Zhu et al. (1989)	4	CS-soybean	Downy brome, chickweed	Missouri	3% slope
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; Slopes 0.5-5%
Griffith et al. (2020) *	1	Corn silage	Cereal rye	New York	5% slope

* not used in recommended average for corn grain-based rotations

Dissolved phosphorus in surface runoff

Cover cropping in a corn-soybean system increased dissolved P losses in surface runoff by 20 ± 80% (that is, a -20% reduction; Figure 42 left panels). Although this mean showed cover cropping had a negative dissolved P impact in surface runoff, the median impact was nearly neutral (-2%; Figure 42 left panels). This illustrated that while most of the time cover cropping may not have an impact on surface runoff DRP losses, there is a risk of a negative impact (i.e., neutral median and negative mean). This difference between the mean (negative) and median (nearly neutral) was one of the few instances where the median was recommended as the overall efficiency rather than the mean. The recent review for the Indiana Nutrient Reduction Strategy similarly found a negative mean (-22%) and neutral median (0%) for the impact of cover crops on surface DRP losses (personal communication, J. Frankenberger). Including four additional studies that used different cropping systems (Table 21) resulted in an even greater risk for dissolved P losses in runoff based on the mean but still a relatively neutral median (-51 ± 171%, median: 4%; Figure 42 right panels).

These studies were from Kansas, Missouri, and Ontario, thus there is some uncertainty in how these values may apply to Minnesota. Soluble P losses were not able to be successfully extracted from a surface runoff study by Weyers et al. (2021) performed in Morris, Minnesota, but their results generally confirmed the possibility for this negative impact. Even so, they reiterated the importance of the surface runoff sediment reduction benefits of this practice.

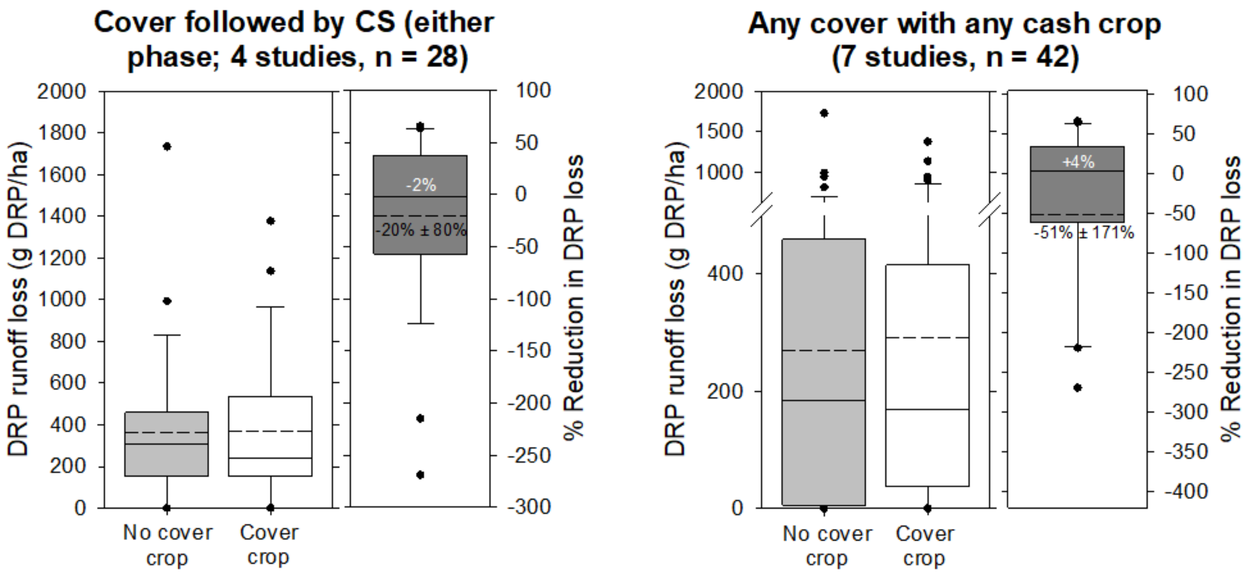


Figure 42. Ranges of dissolved reactive phosphorus (DRP) annual surface runoff losses with and without cover crops. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. A negative "% Reduction…" indicated the practice increased losses of the nutrient. Data were sourced from studies listed in Table 21.

Table 21. Cover crop studies reporting dissolved phosphorus annual surface runoff losses (generally DRP).

Study	# site-years	Cash crop following the cover	Cover	Location	Method and notes
Carver et al. (2022)	3	CS-corn	Wheat, rapeseed	Kansas	3-7% slope
Carver et al. (2022)	3	CS-corn	Triticale, rapeseed	Kansas	3-7% slope
Kaur et al. (2024)	2	CS-corn	Wheat, radish, turnip	Missouri	Terrace tiles draining surface runoff
Zhang et al. (2017)	4	CS-corn	wheat	Ontario	Runoff from tiled plots
Kaur et al. (2024)	2	CS-soybean	Cereal rye	Missouri	Terrace tiles draining surface runoff
Zhang et al. (2017)	4	CS-soybean	wheat	Ontario	Runoff from tiled plots
Carver et al. (2022)	3	CS-soybean	Wheat	Kansas	3-7% slope
Carver et al. (2022)	3	CS-soybean	Triticale, rapeseed	Kansas	3-7% slope
Zhu et al. (1989)	4	CS-soybean	Downy brome, chickweed	Missouri	3% slope
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; Slopes 0.5-5%; one outlier removed
Griffith et al. (2020) *	1	Corn silage	Cereal rye	New York	5% slope
Ulén (1997) *	6	Oats, barley, or wheat	English ryegrass	Sweden	10% slope

* not used in recommended average for corn-soybean rotations

Nearly 35 years ago, Sharpley and Smith (1991) concluded that although cover crops reduce total N and P losses in runoff, the proportion of bioavailable forms may increase. Indeed, a number of studies in the current review found cover cropping could increase the risk of runoff dissolved P losses (Carver et al., 2022; Gutknecht et al., 2023; Sharpley et al., 1995; Ulén, 1997) and this effect can be exacerbated in a cold climate (Aronsson et al., 2016; Bechmann et al., 2005). Daryanto et al. (2018) wrote that an average 5% increase in DRP losses in their review of cover crops was surprising. Smith et al. (2017) noted that because cover crops are so effective at reducing nitrate leaching, a perception may exist that they are also effective for dissolved P loss reduction. The current review provided further quantification of this tradeoff risk.

The results in this space are variable, however, because several studies have reported that cover crops do not increase losses of DRP in runoff (Blanco-Canqui et al., 2013; Griffith et al., 2020; Kleinman et al., 2005). Although it is well known that P can be released into solution from cover crop biomass (Carver et al., 2020; Kovar et al., 2011; Liu et al., 2014), this P can be retained in the field and not present itself as an increase in edge of field losses (Cober et al., 2019; Elliott, 2013; Lozier et al., 2017; Riddle and Bergstrőm, 2013). Likewise, dissolved P concentrations in runoff may increase due to the presence of a living cover (Trentman et al., 2020) but overall dissolved P losses may not increase due to reduction in runoff volume (Christianson et al., 2021; Zhu et al., 1989). This review confirmed the potential for a DRP runoff risk, although much of the time, cover cropping may not have an impact on surface runoff DRP losses (i.e., neutral median in Figure 42).

Snowmelt runoff deserves a special note in this Minnesota-focused assessment. Several cover cropping studies documented high DRP concentrations in snowmelt (Krueger et al., 2013) or that a notable proportion of annual runoff losses can occur as snowmelt (Griffith et al., 2020; Sturite et al., 2007). The effectiveness of vegetative strategies to reduce P losses in snowmelt runoff are uncertain and highly variable (e.g., due to the aspect of the runoff slope; Nater and Krueger, 2012). Cober et al. (2018) implied that cover crops should be used with caution in regions that do not have mild winters because of the risk for increased P losses.

While the number of Minnesota studies was small in this review, there are, of course, a number of excellent P runoff studies that have been performed in the Canadian prairies where snowmelt runoff conditions are relatively similar. However, none were suitable for use in this cover crop chapter because they studied different conservation practices (Grant and Flaten, 2019; Kokulan et al., 2022; Li et al., 2011) or were studies of hydrologic P load partitioning (Kokulan et al., 2019; Kokulan et al., 2023). Tiessen et al. (2010) performed a conservation tillage study in Manitoba and concluded that practices intended to reduce sediment loss can be less effective for dissolved P loss in regions where snowmelt runoff dominates. Studies of vegetive buffers in the Canadian prairies strongly emphasize there is significant uncertainty in the ability of buffers to reduce P losses in cold climates where a large portion of runoff occurs as snowmelt (Habibiandehkordi et al., 2017; Kieta et al., 2018). While these earlier findings pertained to conservation tillage and vegetated buffers, similar limitations (and risks) may need to be considered for cover cropping in Minnesota.

Total phosphorus in surface runoff

Cover cropping in a corn-soybean system reduced total P losses in surface runoff by 10 ± 41% (Figure 43 left panels). This value was developed from most of the same studies reporting runoff dissolved P losses (Table 22), thus this recommendation suffers from the same limitations (i.e., uncertainty in how these values may apply to Minnesota). The median total P impact for surface runoff of 14% reduction confirmed this positive benefit of cover cropping.

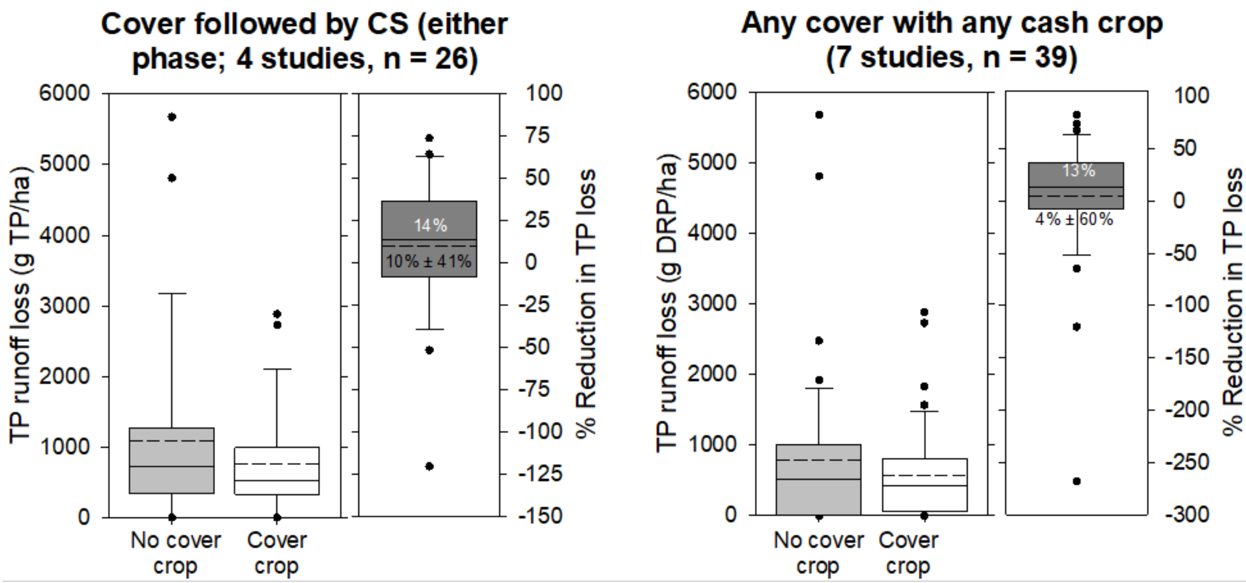


Figure 43.Ranges of total phosphorus (TP) annual surface runoff losses with and without cover crops. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. A negative "% Reduction..." indicated the practice increased losses of the nutrient. Data were sourced from studies listed in Table 22.

Table 22. Cover crop studies reporting total phosphorus annual surface runoff losses from natural rainfall.

Study	# site-years	Cash crop following the cover	Cover	Location	Method and notes
Carver et al. (2022)	3	CS-corn	Wheat, rapeseed	Kansas	3-7% slope
Carver et al. (2022)	3	CS-corn	Triticale, rapeseed	Kansas	3-7% slope
Kaur et al. (2024)	2	CS-corn	Wheat, radish, turnip	Missouri	Terrace tiles draining surface runoff
Zhang et al. (2017)	4	CS-corn	wheat	Ontario	Runoff from tiled plots
Adler et al. (2020)	1	CS-corn	Wheat, radish, turnip	Missouri	Terrace tiles draining surface runoff
Kaur et al. (2024)	2	CS-soybean	Cereal rye	Missouri	Terrace tiles draining surface runoff
Zhang et al. (2017)	4	CS-soybean	wheat	Ontario	Runoff from tiled plots
Carver et al. (2022)	3	CS-soybean	Wheat	Kansas	3-7% slope
Carver et al. (2022)	3	CS-soybean	Triticale, rapeseed	Kansas	3-7% slope
Adler et al. (2020)	1	CS-soybean	Cereal rye	Missouri	Terrace tiles draining surface runoff
Hanrahan et al. (2021) *	8	Corn-soybean-wheat	Not specified	Ohio	Edge of field; Slopes 0.5-5%; two outliers removed
Griffith et al. (2020) *	1	Corn silage	Cereal rye	New York	5% slope
Ulén (1997) *	6	Oats, barley, or wheat	English ryegrass	Sweden	

* not used in recommended average for corn grain-based rotations

The recent review for the Indiana Nutrient Reduction Strategy recommended a mean of 9% loss reduction for total P in surface runoff (median 14%) which was very similar to the current work (personal communication, J. Frankenberger). The Minnesota Watershed Pollutant Load Calculator uses a higher default value for cover cropping of 29% total P reduction in surface runoff (MPCA, 2022). That value was identical to the 29% total P reduction for a cereal rye cover crop recommended in the original Iowa Nutrient Reduction Strategy (IDALS, 2014). All of the studies used to develop the surface runoff total P recommendation here were published following the release of the Iowa Nutrient Strategy in 2014 which may account for the difference in recommended values (e.g., 10% v. 29%).

Cover crop P impacts in above- vs. below-ground pathways

It's useful to put the magnitude of P leaching and surface runoff impacts due to cover cropping in context of each other. For example, this practice provided a mean 10% reduction in surface runoff total P loss but a 13% increase in total P leaching loss (Table 13). So how do these pathways compare in terms of units of loss (g P/ha) rather than relative reduction (%)?

Across this review, the mean total P leaching and surface runoff impacts when a cover crop was used in a corn grain-based system were -61 ± 371 and +317 ± 698 g P/ha (dashed lines in dark gray bars in Figure 44). Taken together, cover cropping provided a net positive total P impact in this simple example (-61 + 317 = 256 g P/ha reduced).

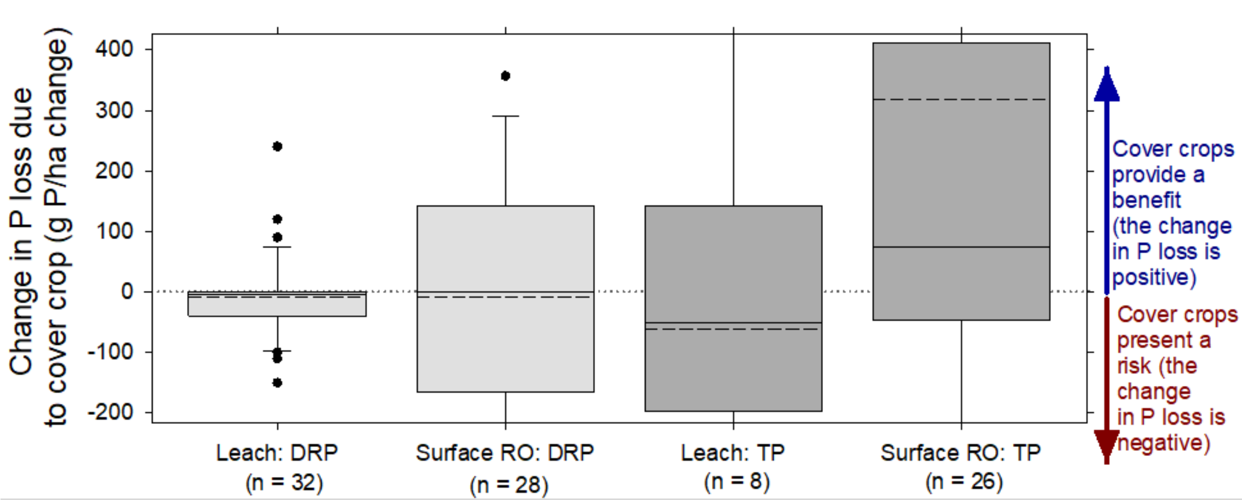


Figure 44. Phosphorus (P) loss impact due to the presence of a cover crop for leaching losses and surface runoff losses and for dissolved P (mostly, DRP) and total P (TP). The convention of the y-axis was: "control P loss – cover crop P loss" where a positive value indicated the cover crop reduced the loss and a negative value indicated the cover crop increased the loss. Y-axis zoomed in to show detail of the means and medians. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers.

This example was limited given the sample size for total P leaching losses in cover crop studies was small and all the site-years were from one study (8 site-years from Zhang et al., 2017; Table 19). When the complete dataset was assessed, the mean leaching impact due to a cover crop was a -74 ± 276 g P/ha reduction ("*any cover with any cash crop*", *n* = 34 from 6 studies, Table 19). Thus, even when a more robust (although less specific) dataset was used, cover cropping still provided a net positive total P impact (-74 + 317 = 244 g P/ha reduced).

A similar analysis showed a small net negative impact of cover cropping on dissolved P. Both below- and above-ground pathways trended toward increased dissolved P losses. The mean dissolved P leaching and surface runoff loss reductions were -9.2 ± 69 and -9.1 ± 168 g P/ha, respectively (-9.2 + -9.1 = 18 g/ha increased). This dissolved P impact should be considered in context of the net total P impact which was more than 10 times larger in magnitude (i.e., +256 versus -18 g P/ha). Soluble nutrient losses can be a tradeoff of vegetative practices but the benefit to losses in total forms is still considerable.

Kleinman et al. (2022) recently defined "*tradeoffs*" associated with conservation practices as strategic decisions where a negative outcome due to the practice is accepted with prior knowledge of the risks. This contrasts with the concept of "*unintended negative consequences*" where a negative outcome of a conservation decision is unforeseen. This distinction is particularly important for vegetated conservation practices that provide sediment control (e.g., cover crops, buffers, sedimentation basins) because these practices can release dissolved P (Kleinman et al., 2022). Acknowledging the potential for these tradeoffs is necessary to develop site-specific and comprehensive conservation plans to address the most appropriate resource concerns.

Chapter 4 References

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Chapter 5: Land use change perennials

Associated NRCS practice codes: CPS #327 Conservation cover; CPS #328 conservation crop rotation

Definition: These practices were broadly defined as cropping management strategies that transitioned away from conventional cropping rotations focused on corn grain production. Most of these practices involved perennials.

Values in the original Minnesota Nutrient Reduction Strategy:

- Perennial energy crops: 95% N reduction, 34% P reduction
- Hay land in marginal cropland (replacing row crops): 95% N reduction, 59% P reduction
- Conservation easements and land retirement: 83% N reduction, 56% P reduction

Table 23. Recommended nutrient reduction efficiencies for land use change practices.

		-------------------- Leaching ----------------			-------------- Surface runoff --------------		
		Nitrate	**Dissolved P**	**Total P**	**Nitrate**	**Dissolved P**	**Total P**
Land use change / perennials in general		---	---	-94 ± 134%	52 ± 40%	21 ± 25% limited data	46 ± 26% limited data
Extended rotation		41 ± 21%	-108 ± 155%	---	---	---	---
	In rotation: Alfalfa	63 ± 41%	---	---	---	---	---
	In rotation: Oats	60 ± 12%	---	---	---	---	---
	"*Conservation crop rotation*" of corn-soybean	-44 ± 61%	---	---	---	---	---
Living covers							
	Kura clover	49 ± 38%	---	---	---	---	---
	Winter oilseed relay crops	Insufficient data	---	---	---	---	---
	Intermediate wheatgrass	Insufficient data	---	---	---	---	---
Land conversion							
	Conversion to prairie	94 ± 9%	-55 ± 119%	---	---	---	---
	Prairie strips	---	---	---	66 ± 5% limited data	---	81 ± 14% limited data
	Conversion to hay land or pasture	74 ± 25%	---	---	---	---	---
	Conversion to bioenergy crops	61 ± 54%	3% [†]	---	---	---	---

[†] Recommended using the median value rather than mean ± standard deviation due to inconsistent impacts of those two metrics (positive or neutral median versus negative mean)

Key highlights

- Conversion of row crops to a prairie land use was the most effective for reduction of N leaching of any conservation practice evaluated (highest efficiency of 94%; lowest coefficient of variation of 0.1).
- All land use change perennials provided greater than 40% N leaching reduction.
- Management of some of these practices impacted their N leaching reduction potential (e.g., fertilized v. unfertilized alfalfa; juvenile v. mature miscanthus).
- Land use change perennials tended to increase P leaching losses. However, the magnitude of these leached losses was small (in units of loss, g P/ha) and the practices provided a net P benefit when surface runoff was considered.
- There were limited annual surface runoff nutrient data for these land use change practices despite their widely known erosion benefits.

Land use change background and database overview

This review chapter considers land use change practices that represent a transition away from conventional cropping rotations focused on corn grain production. These practices ranged from total conversion of row crops to different land uses (e.g., to prairie or bioenergy grasses) to less drastic changes consisting of extended rotations that included corn for grain. The use of living mulches, relay cropping a winter annual oilseed, and intermediate wheatgrass were also included in this chapter. Many of these practices are often termed "*continuous living covers*" (Ecotone Analytics et al., 2023).

Incorporating perennials into the upper Midwest's agricultural landscape is thought to reduce overall N leaching with "*extremely high*" confidence (Kaspar et al., 2008). This quantitative review strongly supports this statement because: (1) the N leaching reductions all exceeded 40% (when a value could be developed, Table 23), and (2) none of the standard deviations overlapped 0 (i.e., the coefficients of variation of these practices are all less than 1.0; Figure 13). Future forecasting by Ecotone Analytics et al. (2023) estimated the practices assessed in this chapter (perennials and winter annual oilseeds) could cover 12 million acres in Minnesota by 2050 and provide 23% N loss reduction for the state (medium adoption scenario). However, one theme that emerged from this review was that the management of these land use change practices influences the resulting N leaching benefit. Kaspar et al. (2008) noted it was a "*misconception that pasture- or forage-based systems always have nutrient loss rates much less than annual grain crops because they are perennial.*" Operation and management are highlighted below as appropriate (e.g., N fertilization; establishment vs. mature periods).

Scientific confidence of perennials to reduce pollutants in surface runoff is also thought to be "*extremely high*" (Kaspar et al., 2008). So, it was surprising that the available surface runoff data were relatively thin in this review (e.g., Figure 45b vs a). This lack of data presents a notable gap especially for the state of Minnesota where a significant portion of total annual runoff comes from snowmelt and where freezing and thawing of living cover vegetation could be a concern for nutrient release. Studies from the Canadian prairies note practices intended to reduce sediment ultimately may not be beneficial for P trapping (or even present a P risk) (Habibiandehkordi et al., 2017; Kieta et al., 2018; Tiessen et al., 2010). Tradeoffs of living covers, especially in terms of dissolved P lysed from biomass, need deeper consideration to develop site-specific conservation plans that address the most appropriate resource concerns (Kleinman et al., 2022).

Additional benefits beyond water quality provided by these practices are not discussed here. For example, wildlife and pollinator habitat (Forcella et al., 2021; Schulte et al., 2017) and the reduction of greenhouse gases of some of these land use changes provide significant public services (Gutknecht and Jungers, 2021). On the other hand, it is also well acknowledged that practices that remove land from agricultural production may not be widely implemented due to economic considerations (MPCA, 2022a).

A total of 135 land use change studies were reviewed. Of these, 45 studies contained water quality data which was extracted to compile 345 water quality site-years. Each "*site-year*" consisted of measured water quality data resulting from a corn grain-based control (i.e., a corn-soybean rotation or continuous corn) paired with water quality data from a land use change treatment in the same location and in the same year. These 345 site-years spanned both leaching and runoff pathways (Figure 45a and b); study locations (Figure 45d); and a variety of land use change practices (Figure 45e). These site-years were further sorted during analysis, so values presented in Figure 45 represent the total available data pool.

Nitrate leaching benefits due to these land use changes have been more documented than these practice's impacts on phosphorus (P) leaching or nutrient losses in surface runoff (i.e., 263 versus < 80 site-years; Figure 45a and b). Thus, there was sufficient data to develop N leaching reduction values for a number of these practices, but dissolved P leaching recommendations were only made for the practices of extended rotations and prairie and bioenergy grass conversion (n = 14, 20, and 17, respectively; Table 23). There were very few site-years for the other nutrient pathways, so the practices were grouped to develop one generic "*land use change*" recommendation for those pathways.

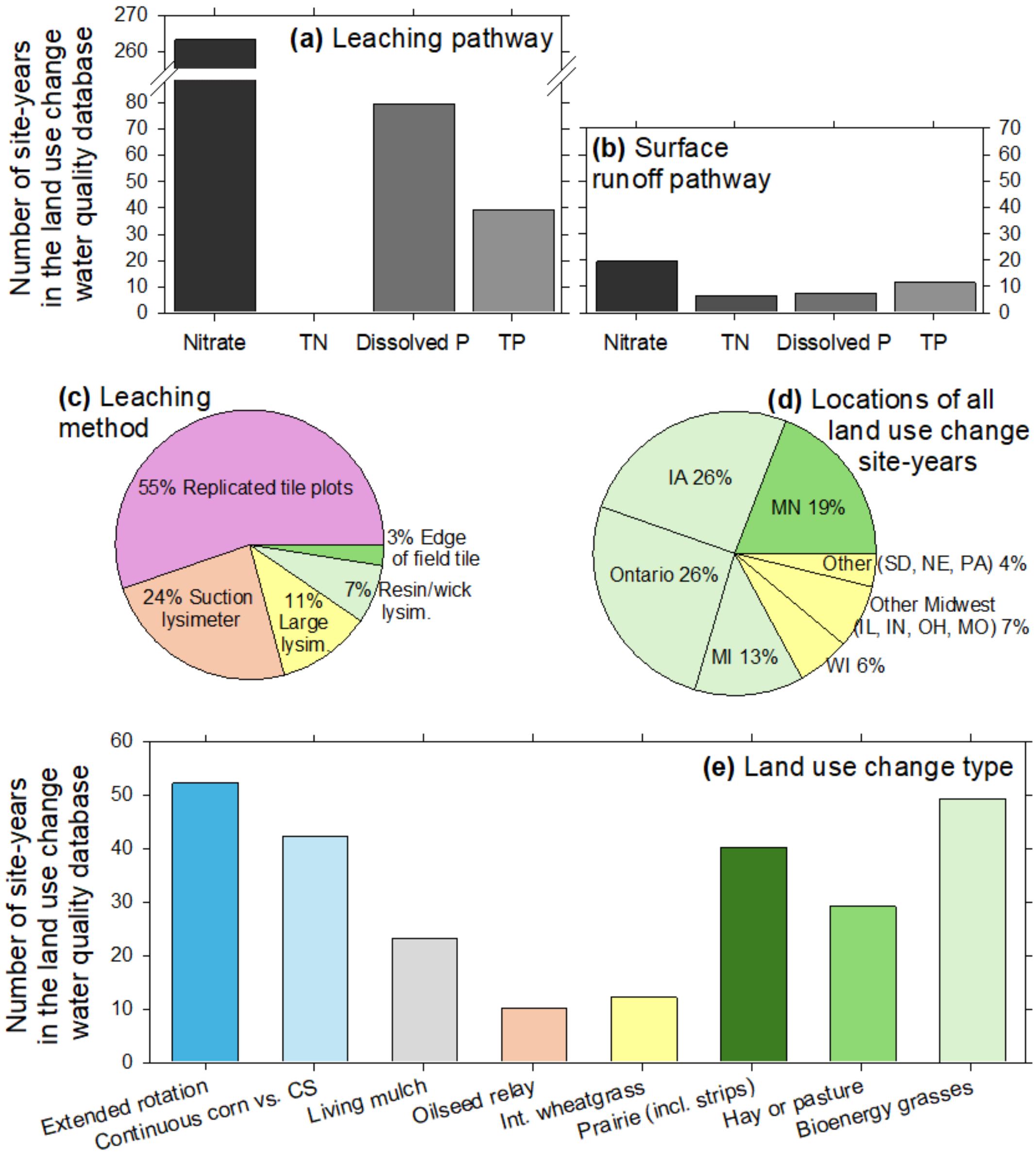

Figure 45. Description of the 345 land use change / perennial site-years compiled into a water quality database showing: (a) and (b) the site-year count for each pathway and nutrient; (c) prevalence of monitoring methods for nutrient leaching; (d) site-year locations; and (e) the site-year count for each land cover change type (individual rotation years are not shown).

Land use change perennials and nitrate leaching loss

Extended rotations and nitrate leaching losses

An “*extended rotation*” was defined as including one or more years of a perennial, which was always alfalfa based on available literature (Table 24). These rotations nearly always also included oats as a small grain (e.g., COAA, CSOAA; Table 24). The five suitable studies showed an extended rotation reduced N loss compared to a corn-grain control by an average of 41 ± 21% (Figure 46). This assessment was limited by the control being defined somewhat inconsistently across the available studies. A corn-soybean rotation was used as the control for 7 site-years (2 studies) whereas 10 site-years (2 studies) used continuous corn. Only one of the five studies used was performed in Minnesota, although Kuehner et al. (2020) reported that a “*corn + alfalfa*” land use provided a 70% reduction in nitrate concentration compared to row cropping in southeast Minnesota (6.6 v. 22.3 mg NO_3-N/L mean values from more than 500 suction lysimeter samples per treatment).

There was no “*extended rotation*” practice included in the original Minnesota Nutrient Reduction Strategy. However, the 42% total N leaching reduction for both groundwater and tile drainage used in the MPCA Watershed Pollutant Load Reduction Calculator for the practice of a “conservation crop rotation” was consistent with the 41% mean recommended in the current work (MPCA, 2022b). These values were also similar to the 42 ± 12% N leaching reduction for the practice of an extended rotation reported in the original Iowa Nutrient Reduction Strategy (“*at least 2 years of alfalfa in a 4 or 5 year rotation*”; IDALS, 2014).

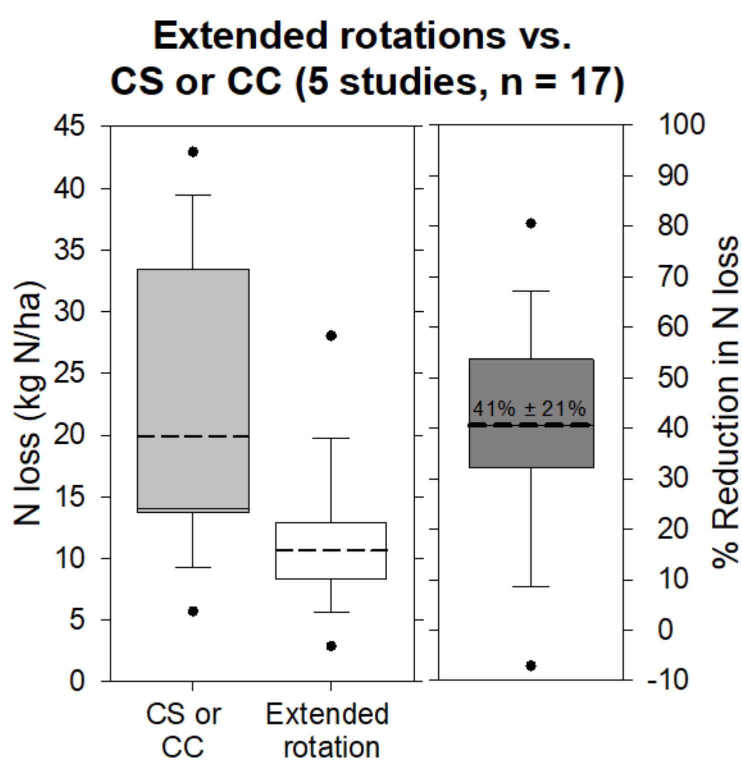

Figure 46. Annual nitrate leaching loss and loss reduction due to an extended rotation (including at least one year of a perennial) versus either a corn-soybean (CS) or continuous corn (CC) system. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 24.

Table 24. Extended rotation studies reporting annual nitrate leaching losses. C, corn; S, soybean, O, oat; A, alfalfa.

Study	# site-years	Conventional crop rotation (control)	Extended rotation details	Location	Method and notes
Oquist et al. (2007)	3	"conventional"; CS-average	"alternative" farming; CSOA	Minnesota	Edge of field tile;
Kanwar et al. (2005)	1	CS average	AAACSO	Iowa	Replicated tile;
Cambardella et al. (2015)	3	CS average	CSOAA	Iowa	Replicated tile; Organic rotation;
Bolton et al. (1970)	7	Continuous corn	COAA	Ontario	Replicated tile; One 7-y average reported;
Tan et al. (2002)	3	Continuous corn	COAA	Ontario	Replicated tile;
Kuehner et al. (2020) *	5 (conc. only)	"Row crop"	"Corn + alfalfa"	Minnesota	Suction lysimeter at 122 cm

* not used in recommended average value for N losses

Alfalfa nitrate leaching losses

Use of alfalfa either alone or in rotation has been relatively well studied and this crop is widely known to reduce N leaching compared to corn-based cropping systems (Kanwar et al., 1996; Koropeckyj-Cox et al., 2021; Randall et al., 1997; Rasse and Smucker, 1999; Singh et al., 2023; Toth and Fox, 1998). This review resulted in 47 site-years for alfalfa N leaching losses but 15 of those used corn silage as the control (Table 25; Basso and Ritchie, 2005; Gamble et al., 2022). Sorting these data further to include use of only a corn grain control resulted in sufficient data to also separate fertilized alfalfa from unfertilized (*n* = 20 and 12, respectively; Table 25).

Table 25. Alfalfa studies reporting annual nitrate leaching losses.

Study	# site-years	Cash crop rotation (control)	Alfalfa details	Location	Method and notes
Bolton et al. (1970)	14	Continuous corn	Fertilized	Ontario	Replicated tile; Alfalfa years 1 and 2 reported for 7 y with one value each
Tan et al. (2002)	6	Continuous corn	Fertilized	Ontario	Replicated tile; Alfalfa years 1 and 2 reported;
Toth and Fox (1998)	1	Continuous corn	Unfertilized	Pennsylvania	Zero-tension pan lysimeter; 3 y of concentrations reported, 1 y of N loss with corn at 200 kg N/ha;
Randall et al. (1997)	4	CS-average	Unfertilized	Minnesota	Replicated tile;
Kanwar et al. (2005)	3	CS-average	Unfertilized	Iowa	Replicated tile; 3 consecutive y of alfalfa in a 6 y rotation
Syswerda (2009)	4	Corn in rotation (CSW)	Unfertilized	Michigan	Suction lysimeter at 120 cm; Also see Syswerda et al. (2012)
Radatz (2021) *	1 (conc. only)	CS-average	NA	Minnesota, Wisconsin	Edge of field tile; Methods and site-years varied so counted as one
Basso and Ritchie (2005) *	12	Corn silage	Unfertilized	Michigan	Large undisturbed lysimeters (#1, 2, 7, and 8)
Gamble et al. (2022) *	3	Corn silage	Fertilized (manure)	Minnesota	Edge of field tile;

* not used in recommended average value for corn grain-based control

On average, alfalfa reduced N loss compared to a corn-grain control by 63 ± 41% (Figure 47a, *n* = 32). Separating these data, N leaching reduction was notably higher when the alfalfa was not fertilized versus fertilized (87 ± 28% and 49 ± 42%, respectively; Figure 47c and d). Even so, a

49% mean reduction for the fertilized alfalfa still exceeded the N leaching benefit provided by many other in-field practices (e.g., cover crops, N fertilizer management). Caution is also somewhat merited for interpretation of the high leaching reduction calculated for the unfertilized alfalfa. Three different methods were used across those 12 site-years and the suction lysimeter method used in Michigan resulted in very high corn N losses (> 130 kg N/ha, upper outliers for corn grain control in Figure 47a and c; Syswerda, 2009). The two studies that used corn silage as the control resulted in a mean N leaching reduction of 38 ± 46% provided by alfalfa (Figure 47b; median: 52%). Finally, it is important to note there could be an increase in N leaching following alfalfa termination. The magnitude and overall impact of this effect (over the long-term) needs to be better understood.

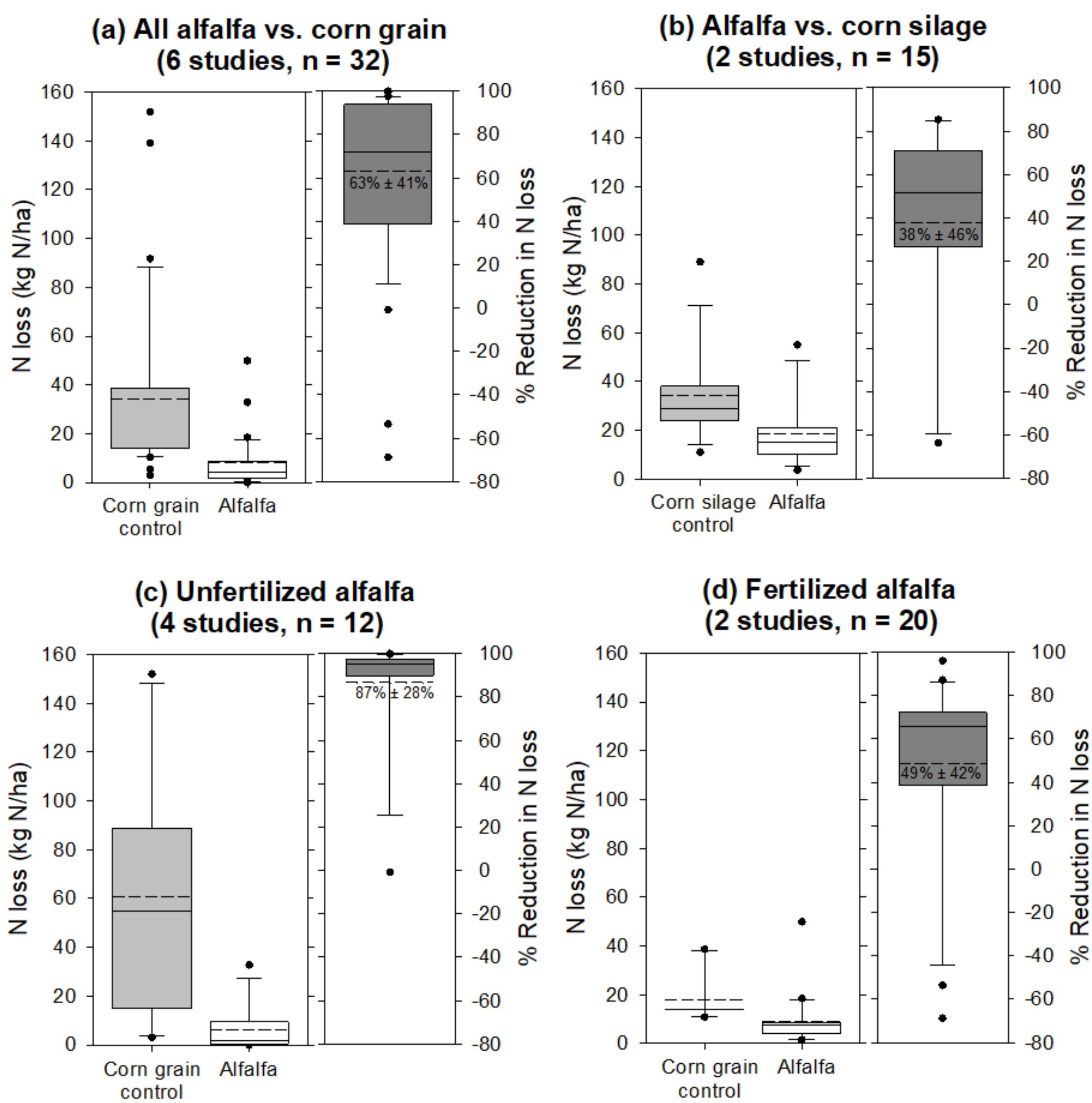

Figure 47. Annual nitrate leaching loss and reduction due to alfalfa from all site-years where corn for grain was the control (a) or where corn silage was the control (b). Panel a can be separated into unfertilized alfalfa (c) and fertilized alfalfa (d). Panel (b) shows site years The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 25.

Alfalfa is effective for leaching reduction because it can reduce the drainage nitrate concentration and the drainage discharge (Randall et al., 1997), as well as the variability of drainage discharge (Arrueta et al., 2022). In summary, alfalfa is a robust and flexible option to reduce N losses in corn-based cropping systems, even when extreme mitigation of N is needed (Russelle et al., 2001).

Small grain nitrate leaching losses

Oat in rotation resulted in an average N loss reduction of 60 ± 12% versus a corn grain control for that given small grain year (Figure 48). There were three studies where N loss from a small grain (oat) could be assessed individually versus a corn-soybean or continuous corn system (Table 26). This site-year assessment was limited because one study only reported the 7-year average rather than N losses from the individual years (Bolton et al., 1970). Thus, that one value of 59% was used in the database seven times and heavily influenced the result.

Despite this data limitation, small grains are expected to provide relatively high N leaching reduction compared to a corn grain-based system. The Minnesota Discovery Farms reported a small grains treatment provided a 39% concentration reduction (wheat vs. corn-soybean average: 7.5 vs. 12.4 mg NO_3-N/L; Radatz, 2021). A recent review of extended rotations by Koropeckyj-Cox et al. (2021) reported crop phase-specific N losses of more than 28 kg N/ha for either corn or soybeans but only 12.3 kg N/ha for wheat and 6.3 kg N/ha for oat. However, as with the current work, the sample sizes for the wheat and oat phases were small (n < 16).

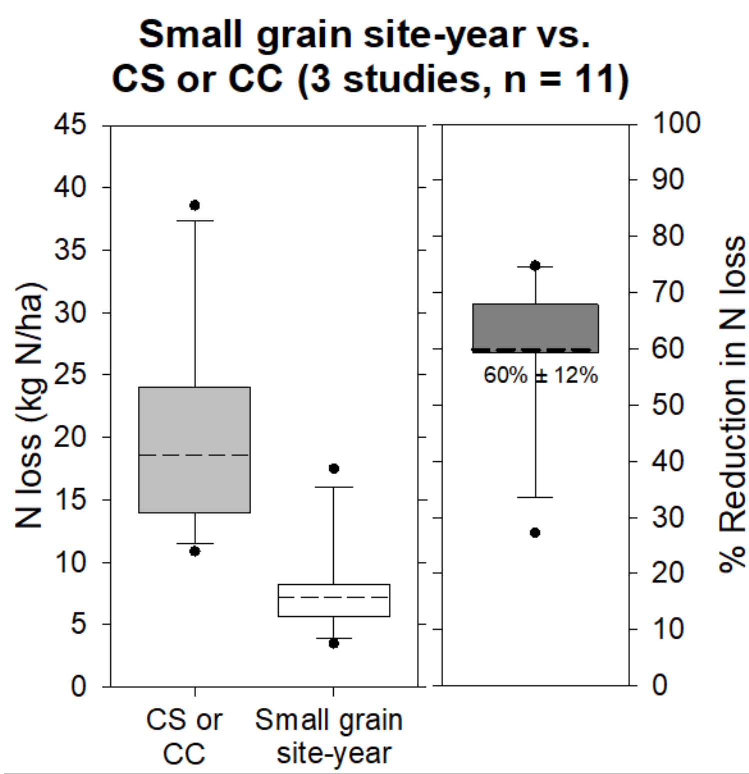

Figure 48. Annual nitrate leaching loss and loss reduction due to a small grain site-year (usually oat) versus either a corn-soybean or continuous corn system. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 26.

Table 26. Studies of small grains, primarily within an extended rotation, that reported annual nitrate leaching losses.

Study	# site-years	Conventional crop rotation (control)	Small grain	Location	Method and notes
Kanwar et al. (2005)	1	CS average	Oat in rotation	Iowa	Replicated tile;
Bolton et al. (1970)	7	Continuous corn	Oat in rotation	Ontario	Replicated tile; One 7-y average reported;
Tan et al. (2002)	3	Continuous corn	Oat in rotation	Ontario	Replicated tile;
Radatz (2021) *	1 (conc. only)	CS average	Wheat	Minnesota, Wisconsin	Edge of field tile; Methods and site-years varied so counted as one

* not used in recommended average values for N losses

Should the term "conservation crop rotation" include a corn-soybean rotation?

The practice of a "*conservation crop rotation*" is defined by the NRCS as "*a planned sequence of crops grown on the same ground over a period of time (i.e. the rotation cycle)*" with the purpose of addressing soil, water, and/or wildlife habitat resource concerns (MN NRCS, 2016). The most common agricultural land use in Minnesota and across much of the US Midwest of a corn-soybean rotation falls within this definition. However, consideration of a corn-soybean rotation as a "*conservation practice*" from the specific perspectives of water quality and N leaching merits deeper assessment.

Older studies indicated that a corn-soybean rotation could decrease N leaching compared to a continuous corn system (Bakhsh and S. Kanwar, 2007; Kanwar et al., 1997; Owens et al., 1995). In the "*Measured Annual Nutrient loads from AGricultural Environments*" (MANAGE) database, N losses from 2-y rotation corn and soybean phases ranged lower than those from continuous corn in wet years, but not in dry years (Christianson and Harmel, 2015). Most recently, a review by Koropeckyj-Cox et al. (2021) concluded a corn-soybean rotation was a cost effective conservation crop rotation compared to continuous corn even though they reported higher average N leaching from the corn-soybean rotation (rotation vs. continuous corn: 32.4 vs. 31.3 kg N/ha).

These previous comparisons did not account for the higher N application rates that are typically applied to corn that is grown continuously compared to in rotation. The MRTN for corn in a corn-soybean rotation is lower than that for continuous corn to account for decomposition of the soybean residue (Sawyer et al., 2006). The effect of N rate on N leaching is well acknowledged (e.g., Baker and Johnson, 1981) and the use of different N rates is one reason why a 2-y rotation may leach less N when studies comparing these two corn-based systems are not normalized by N rate (Helmers et al., 2012b; Weed and Kanwar, 1996).

Nitrate leaching from a corn-soybean rotation compared to continuous corn can be assessed --- in an apples to apples way --- using studies where both corn-based systems received relatively consistent N application rates. This literature review resulted in six studies (27 site-years) where continuous corn and a corn-soybean rotation were both studied and N rates to corn were within 15% for both systems (Table 27; Figure 49b). A corn-soybean rotation increased N leaching loss by an average of 39 ± 60% compared to continuous corn when both were assessed at similar N rates to corn (Figure 49a and b). Thus, where nitrate leaching is concerned, a corn-soybean rotation did not inherently provide a benefit over continuous corn when the two systems were compared at the same N application rate (Helmers et al., 2012b; Klocke et al., 1999).

Table 27. Studies evaluating nitrate leaching loss from both continuous corn and a corn-soybean rotation (2-y average), where the corn received relatively consistent N application rates. Only site-years where the N application rates were within 15% of each other were included in the analysis.

Study	**# site-years**	**---- N rate (kg N/ha) ----**		**Location**	**Method and notes**
		CC	**CS-corn**		
Randall et al. (1997)	1	135	120	Minnesota	Replicated tile; 3 additional site-years had different N rates
Wayment (2021)	5	300	300	Minnesota	Suction lysimeter; irrigated sands
Ochsner et al. (2018)	4	146	146	Minnesota	Eq. tension lysimeter at 105 cm
Radatz (2021) *	3 (conc.)	100, 150, 200	100, 150, 200	Minnesota, Wisconsin	Edge of field tile; Methods and site-years varied so counted as three comparisons;
Daigh et al. (2015)	4	123, 143, 200, 247	105, 127, 222, 247	Iowa	Replicated tile; discharge in Daigh et al. (2014)
Helmers et al. (2012b)	7	112, 168	112, 168	Iowa	Replicated tile
Hernandez-Ramirez et al. (2011)	6	180	180	Indiana	Replicated tile

* not used in recommended average value for N losses

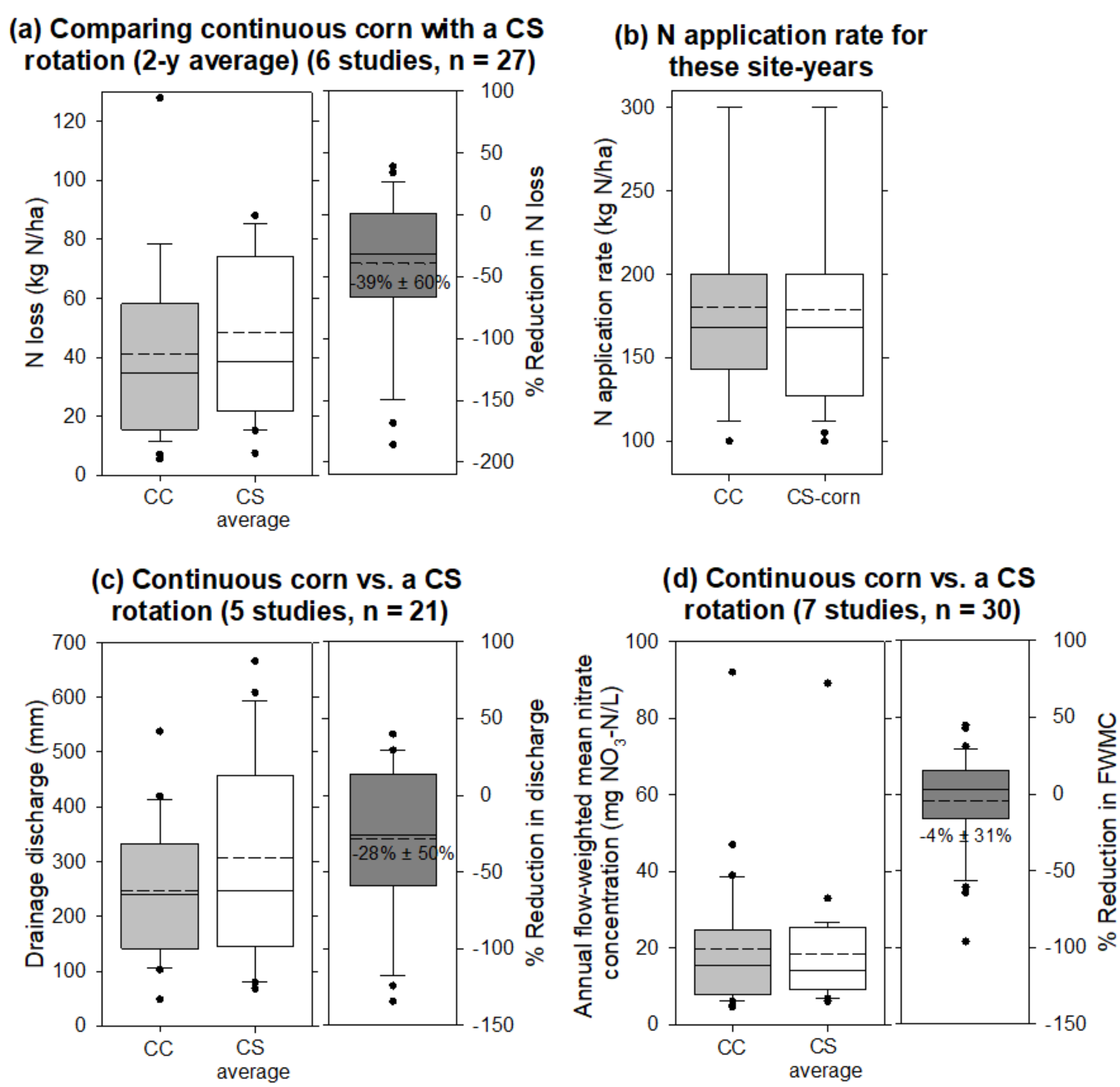

Figure 49. Annual nitrate leaching loss and loss reduction from continuous corn and corn-soybean rotation (a) when both systems are compared at the same N rate to corn (b). Drainage discharge (c) and annual flow-weighted mean nitrate concentrations (FWMC; d) are also shown. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. A negative "% Reduction…" indicated the corn-soybean rotation increased the metric compared to continuous corn. Data were sourced from studies listed in Table 27.

Although a study performed across Discovery Farm sites in Minnesota and Wisconsin did not report N losses, Radatz (2021) found that corn-soybean rotations had tile drainage nitrate concentrations approximately 15% lower than continuous corn when both were assessed at application rates of 168 and 224 kg N/ha. Daigh et al. (2015) similarly observed that a corn-soybean rotation had a lower 4-y mean N concentration compared to continuous corn (numerically, not significantly). However, the 4-y mean N loss was lower from the continuous corn at these replicated tile drainage plots in Iowa (also not significantly). Across the current assessment, leachate nitrate concentrations did not differ between corn-soybean versus continuous corn (mean: -4 ± 31%; median: +3%; Figure 49d). However, the corn-soybean rotation tended to have greater drainage discharge than continuous corn (Figure 49c) which may have accounted for the relatively higher N loss from the former.

Corn grain yield is obviously an important practical consideration for these two corn-based cropping systems. Corn in the continuous versus rotation systems yielded 8.0 ± 1.8 versus 9.3 ± 1.6 Mg/ha, respectively, when they were compared at the same N rate (Figure 50 left panels). Thus, the 2-y rotation provided a notable corn yield benefit averaging 18% more. Koropeckyj-Cox et al. (2021) noted that a corn-soybean rotation was more profitable than a continuous corn system with an average benefit of $5 per kg N reduced. This was contrary to the present findings since the corn-soybean rotation didn't reduce N leaching (Figure 49a) although the yield benefit would increase gross income for the corn years.

Despite this higher corn yield in the corn-soybean rotation, this system's higher N leaching resulted in a higher average yield-scaled N loss than continuous corn (means: 5.5 vs. 4.6 kg N leached per Mg grain produced; Figure 50 right panel). In other words, continuously cropped corn produced less risk to water quality when N losses were normalized by yield (and the comparison was performed at the same N application rates). In contrast, Zhao et al. (016) previously reported that corn-soybean rotations reduced yield-scaled N losses by 46% compared to continuous corn (weighted means: just over 3 versus 6 kg N/Mg, respectively). In that earlier work, this difference was driven by the higher corn yield in the rotation whereas there was no difference in N loads between the two systems. The earlier work was not normalized by N rate as was done here. Comparison of the earlier and current results thus illustrates differences in practical management of these two corn systems (and the importance of N rate).

In summary, to address the direct question, these data did not support a corn-soybean rotation being considered a conservation crop rotation from the specific context of N leaching. In practice, however, this comparison is linked with N application rate. If lower N rates are applied to corn in rotation versus corn in a continuous system, those lower N rates may benefit N loss. It is near impossible to separate the impact of rotation from N rate in reality. However, to be able to attribute a N leaching benefit to the rotation rather than the N application rate reduction, the soybean phase of a corn-soybean rotation would need to provide a disproportionate benefit in N leaching. Nevertheless, it is well documented that N leaching from the soybean phase is often similar to (or, can be more than) that of the corn phase in a 2-y rotation (Hernandez-Ramirez et al., 2011; McIsaac et al., 2010; Owens et al., 2000).

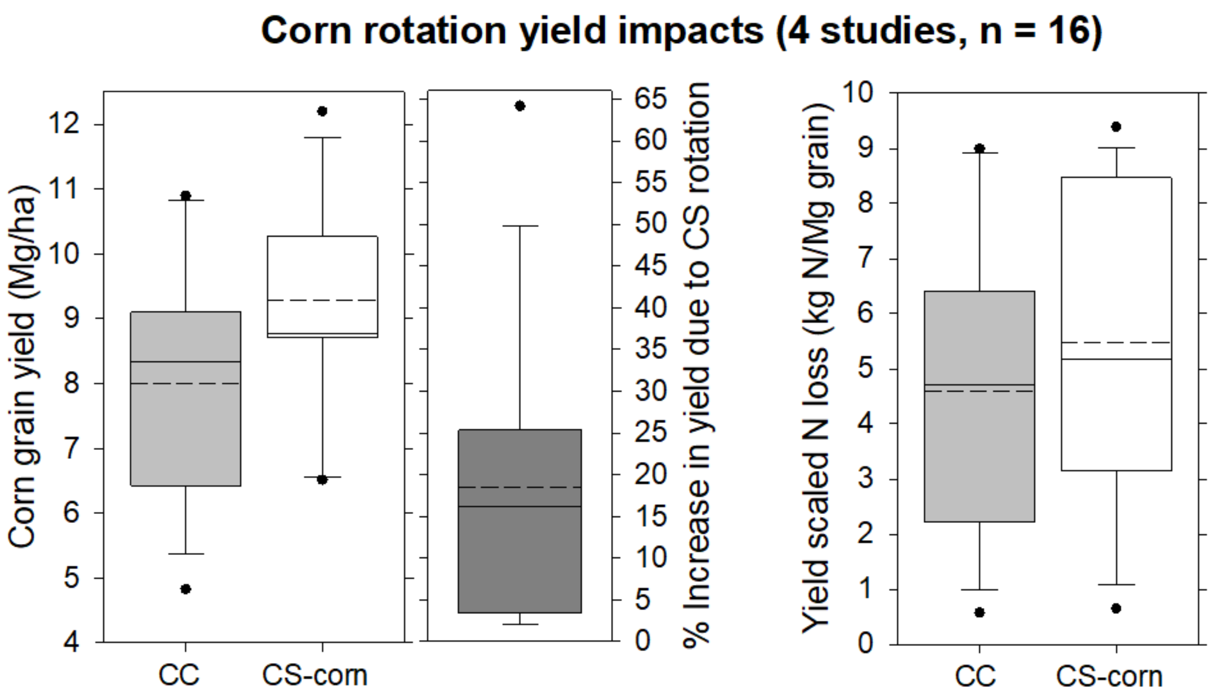

Figure 50. Corn grain yield impacts due to a continuous corn versus corn-soybean rotation with corn in both systems fertilized at similar N rates in a given site-year comparison. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 27.

Living covers or relay cropping and nitrate leaching losses

Kura clover as a living mulch

A living mulch is described as a cover crop, usually a perennial, planted before or with the primary cash crop that is maintained as ground cover during the growing season (Hartwig and Ammon, 2002; Kaspar et al., 2008). Kura clover has been the primary living mulch used in water quality studies in the US Midwest (Table 28), although other perennial covers have been studied (e.g., Kentucky fescue, creeping red fescue, Italian ryegrass, alfalfa, Kentucky bluegrass; Banik et al., 2020; Eberlein et al., 1992; Liedgens et al., 2004). Kura clover grown in a corn-grain rotation reduced N leaching loss by an average of 49 ± 38% (Figure 51). This recommendation was based on three studies from Iowa and Minnesota (Table 28). A fourth study from Wisconsin was excluded because corn silage was the control (Ochsner et al., 2010). Nevertheless, the N leaching reductions they reported of 31% (when fertilized) or 74% (when not fertilized) spanned the average of 49% calculated here. Two additional Kura clover studies were considered for the surface runoff assessment (Table 35: Eleki, 2003; Siller et al., 2016).

Table 28. *Kura clover studies reporting annual nitrate leaching losses.*

Study	**# site-years**	**Cash crop following or with the Kura**	**Location**	**Method and notes**
Qi et al. (2011)	4	CS-corn	Iowa	Replicated tile;
Wayment (2021)	5	CS-average	Minnesota	Suction lysimeter below the root zone
Wayment (2021)	5	Continuous corn	Minnesota	Suction lysimeter below the root zone
Ochsner et al. (2018)	3	CS-average	Minnesota	Equilibrium tension lysimeter at 105 cm
Ochsner et al. (2010) *	4	Corn silage	Wisconsin	Suction lysimeter at 100 cm;

* not used in recommended average value for corn grain-based systems

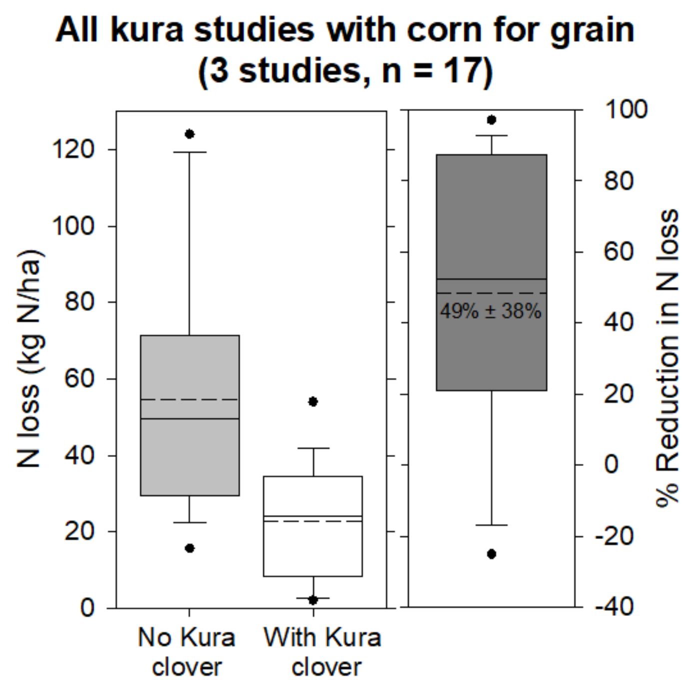

Figure 51. Annual nitrate leaching loss and loss reduction due to Kura clover grown with corn for grain. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 28.

Use of a living mulch would be expected to reduce annual N loss by altering both the drainage/leachate volume and the concentration of nitrate in the leachate. However, a variety of methods have been used to assess these impacts (Table 28: replicated tile plots, suction lysimeters, equilibrium tension lysimeters). Two studies reported a Kura clover treatment did not reduce the discharge compared to the control (Ochsner et al., 2010; Qi et al., 2011). More recently, Ochsner et al. (2018) noted a roughly 70% reduction in leachate volume due to Kura clover. This highlights the difficulty in comparing water quality impacts across methods and reiterates the importance of direct measures of water quality (both directly measured volume and concentration).

Compared to cover cropping (e.g., cereal rye), this land use cover provided a good example of being able to put on biomass even in a cold climate (average: 2.0 ± 1.5 Mg/ha; median 1.6 Mg/ha; *n* = 12; data not shown). This resulted in notably high N uptake averaging 58 ± 34 kg N/ha; for comparison, cereal rye cover crop N uptake in this review averaged 29 ± 21 kg N/ha (*n* = 82 for individual crop phases, e.g., Figure 33). All of the Kura clover biomass data were from either Iowa (*n* = 4; Qi et al., 2011) or Minnesota (*n* = 8; Wayment, 2021).

When compiled across eight corn-soybean corn phase site-years and four continuous corn site years, the average corn yields without and with Kura clover were 11.2 ± 3.0 and 7.7 ± 3.8 Mg/ha, respectively (Figure 52; average of -36 ± 24%). Reduced yield of the main cash crop is the major challenge for use of this practice (Banik et al., 2020; Ochsner et al., 2018; Qi et al., 2011; Wayment, 2021). There was not a consistent impact on yield-scaled N loss when Kura was grown with corn. That is, the median yield-scaled N loss decreased whereas the mean increased when Kura clover was grown. One site-year with Kura clover had an extremely high yield-scaled N loss because although the Kura clover reduced N leaching by 41% (from 66 to 39 kg N/ha without and with Kura, respectively), the corn yield with the Kura clover was only 1.0 Mg/ha (Qi et al., 2011).

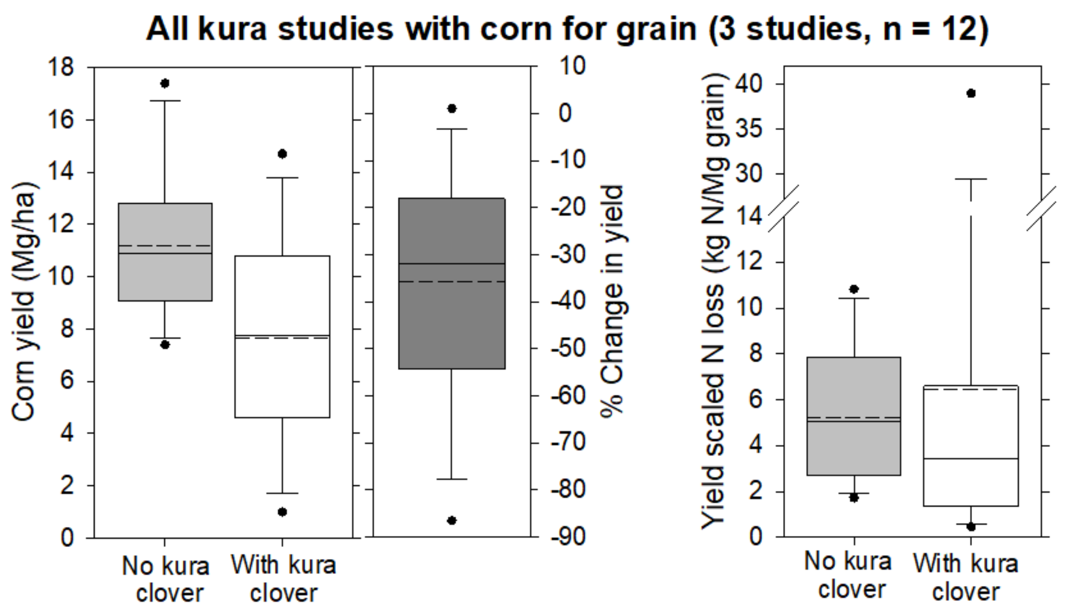

Figure 52. Corn yield impact and yield-scaled N losses due to Kura clover. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies in Table 28.

Winter annual oilseed relay crops (winter camelina and pennycress)

Winter annual oilseeds, such as winter camelina and pennycress, are receiving attention as a relatively new emphasis in the living cover movement in Minnesota (Vondracek, 2023; 2024). These crops are planted in the late fall like traditional cover crops but are harvested in the late spring or early summer as a "*cash*" cover crop (Gesch et al., 2014; Gesch and Cermak, 2011). They are often relay cropped which means the primary cash crop is planted before the winter annual oilseed is harvested (Emmett et al., 2022). Winter camelina and pennycress have a high tolerance for cold temperatures which makes them especially suitable for Minnesota (Forcella, 2023; Gesch and Cermak, 2011). These winter annuals are forecasted to have a large market potential and accordingly are expected to have a substantial future acreage in the state (5 million acres of winter annual oilseeds by 2050; Ecotone Analytics et al., 2023).

The Forever Green Initiative (https://forevergreen.umn.edu/crops) is actively engaged in developing a range of perennial and winter annual crops, including camelina and pennycress, with a focus on improving soil and water quality, biodiversity, and economic opportunities for Minnesota farmers. While research on these crops is still evolving, a recommendation for future studies under this initiative could include a stronger focus on quantifying nutrient leaching reduction, particularly nitrogen, to better inform updates to the science assessment and conservation practices.

Relay cropping winter annual oilseeds is an example of where management could significantly influence the potential for N leaching reduction. Nitrogen fertilization of winter camelina and pennycress is necessary for optimal oilseed production (e.g., roughly 50-100 kg N/ha; (Forcella, 2023). However, fertilizing these crops reduces the N leaching reduction benefit and can increase the nitrous oxide emissions (Emmett et al., 2022). The summary analysis by Ecotone Analytics et al. (2023) noted winter annual oilseeds were the only continuous living cover crop that increased greenhouse gas emissions (by 14%) but also noted there may be off-farm benefits given their use in sustainable transportation fuel.

Fourteen studies of these relay crops were reviewed and three provided direct measurements of water quality (Table 29: Emmett et al., 2022; Weyers et al., 2019; Weyers et al., 2021). However, Weyers et al. (2019) presented only seasonal leached nitrate concentrations (not annual, not N losses) and there was some difficulty extracting surface runoff nutrient losses appropriately from Weyers et al. (2021) (unclear axes, personal communication initiated). Thus, Emmett et al. (2022) provided the only sufficient data from a winter camelina relay crop versus a corn-based control for this specific review. The practice of relay cropping would be expected to impact both the annual drainage/leachate volume as well as the nitrate concentration, thus assessment of N loss rather than just concentration matters for this conservation practice.

Averaging across the 2-year corn and soybean rotation, relay cropping winter camelina provided -3 ± 16% annual N loss reduction (median: -2%, n = 3; Emmett et al., 2022). Assessing this practice based on nitrate concentrations to be able to include the Minnesota results resulted in a mean N concentration reduction of 18 ± 47% (n = 8, Table 29). This average N concentration reduction illustrates the possible N leaching benefit of winter camelina, but more research is needed as its development was inconsistent with the methods used in this systematic review (e.g., the three seasonal concentrations reported by Weyers et al. (2019) had to be averaged to develop an annual value; N losses, not concentrations, are needed).

Table 29. Total water quality review of studies of the winter annual oilseeds camelina and pennycress.

Study	# site-years	Primary cash crop	Winter oilseed	Location	Method and notes
Emmett et al. (2022)	6 (3 x 2-y CS rotation)	CS rotation	Camelina; fertilized	Iowa	Replicated tile
Weyers et al. (2019) *	2 (conc. only)	Soybean	Camelina	Minnesota	Suction lysimeter at 100 cm; additional data in Ott (2018) and Ott et al. (2019)
Weyers et al. (2019) *	2 (conc. only)	Soybean	Pennycress	Minnesota	Suction lysimeter at 100 cm; additional data in Ott (2018) and Ott et al. (2019)
Weyers et al. (2021)	2	Soybean	Camelina	Minnesota	Annual surface runoff; unable to extract loss data
Weyers et al. (2021)	2	Soybean	Pennycress	Minnesota	Annual surface runoff; unable to extract loss data
Berti et al. (2017) *	NA	---	Camelina	Minnesota	No original water quality data: Life cycle assessment
Cecchin et al. (2021a) *	NA	---	Camelina; pennycress	Midwest	No original water quality data: Life cycle assessment; also see Cecchin et al. (2021b)
Forcella et al. (2021) *	NA	---	Camelina; pennycress	Minnesota, Iowa	No water quality data: pollinator study
Forcella (2023) *	NA	---	Camelina	Minnesota	No original water quality data: Review only
Gesch et al. (2014) *	NA	Soybean	Camelina	Minnesota	No water quality data: yield and economics
Gesch and Cermak (2011) *	NA	---	Camelina	Minnesota	No water quality data: sowing date study
Mohammed et al. (2020) *	NA	Corn, soybean	Camelina; pennycress	Midwest	No water quality data: Soil N measures
Liu et al. (2019) *	NA	---	Camelina; pennycress	Minnesota	No water quality data: Soil nitrate from 0-30 cm
Moore et al. (2020) *	NA	Sweet corn	Pennycress	Minnesota	No water quality data: Soil inorganic N concentrations

* not used in recommended average loss reduction values (insufficient data)

While this review also included relay cropped pennycress, the available data were insufficient at this time to recommend a representative N leaching reduction value. Moore et al. (2020) found pennycress reduced soil mineralizable N by 27-42% at two locations in Minnesota and Mohammed et al. (2020) reported pennycress reduced soil nitrate concentrations at one of four study locations. Soil nitrate concentrations (i.e., soil extracts) are not direct measures of water quality, thus these data were not appropriate for use in this analysis.

The lack of annual N leaching losses reported for the practice of a relay cropped winter annual oilseed and the disparity of results based on management differences (e.g., fertilized vs. unfertilized camelina) means additional data is needed across management approaches to develop a representative N leaching reduction value. Future research to develop a recommended value for these winter oilseeds should consider use of: (1) an appropriate control that is coordinated and consistent across camelina/pennycress studies and also across studies of other practices (e.g., a 2-y corn-soybean rotation); (2) direct measures of both N concentration and leachate volume that are made below the root zone (e.g., using replicated drainage plots); (3) clear reporting of annual values (e.g., in addition to seasonal values which are important for assessing impacts of relay cropping).

It is expected that these crops will continue to be developed in Minnesota (Cecchin et al., 2021a; Ecotone Analytics et al., 2023; Mohammed et al., 2020) especially given the multiple environmental benefits they can provide (pollinator habitat, erosion reduction; Forcella et al., 2021; Weyers et al., 2021). Additional research and development are needed to address the potential for these relay crops to cause yield penalties for the primary cash crop (Liu et al., 2019; Ott et al., 2019). Relay intercropping in general faces many of the same barriers as cover cropping, and a series of recent outreach workshops indicated this practice would likely not become commonplace, at least in Iowa (Licht, 2024). Overall, further assessment of the possible N leaching benefit of relay cropped winter annual oilseeds, in tandem with the variety development process, is necessary to more appropriately calibrate society's expectation of the water quality benefit of this practice.

Intermediate wheatgrass

Of the six intermediate wheatgrass studies identified in this water quality review, three studied leaching and one studied surface runoff using rainfall simulations (not applicable, Katuwal et al., 2023; Table 30). Use of a consistent control for the treatment of intermediate wheatgrass was an issue because a corn grain system was used in six of the extracted leaching site-years whereas annual wheat was used as the control in the other six site-years. Within the six site-years that used corn as the control (which was most appropriate for this review), only the study by Jungers et al. (2019) reported N losses. The other study by Reilly et al. (2022) reported suction lysimeter nitrate concentrations.

The average N loss reduction reported by Jungers et al. (2019) for intermediate wheatgrass was 99% (0.2 vs. 21.7 kg N/ha, high fertility treatment). However, this was assessed at a relatively shallow depth (i.e., suction lysimeters at 50 cm, Table 30) and the variability across sites and years was not transparent since only one value was presented. Additional data for the individual years and locations across Minnesota was not available for this review (personal communication initiated). The average N concentration reduction across the two Minnesota leaching studies was 94 ± 8% (n = 6). To echo the suggested future research for relay cropped winter annual oilseeds, assessment of intermediate wheatgrass using: (1) an appropriate and consistent control and (2) direct measures of both N concentration and leachate volume (i.e., using replicated drainage plots) are recommended to refine understanding of the leaching reduction benefits.

It is useful to note that five of the six intermediate wheatgrass studies reviewed were published since the original Minnesota Nutrient Strategy was released in 2014. This indicates positive research momentum for this crop in the state of Minnesota and the upper Midwest. Further study is needed of a possible establishment effect; intermediate wheatgrass may not reduce N leaching as much in the first year after establishment as in later years (Culman et al., 2013; Reilly et al., 2022). Lastly, further research on and development of grain yield for this crop will be important for future adoption.

Table 30. Total water quality review of intermediate wheatgrass studies.

Study	# site-years	Conventional crop rotation (control)	Location	Method and notes
Jungers et al. (2019)	3 (1 value used for 3 y)	Corn	Minnesota	One average value reported for 3 y and 3 locations; high fertilizer treatment; suction lysimeter at 50 cm
Reilly et al. (2022)	3 (conc. only)	Corn-soybean	Minnesota	Suction lysimeter at 60 and 120 cm (avg'd across depths)
Rakkar et al. (2023)	0	Corn-soybean; winter wheat-red clover rotation; alfalfa	Minnesota	No direct measures of water quality (soil health study)
Culman et al. (2013)	6	Annual winter wheat	Michigan	Suction lysimeter at 135 cm
Sprunger et al. (2018)	0	Annual winter wheat	Michigan	Not a water quality study (biomass and nitrogen use efficiency)
Katuwal et al. (2023)	0	Winter wheat; switchgrass; Gama grass	Arkansas	Rainfall simulations (4) for surface runoff events

Land conversion and nitrate leaching losses

Prairie and continuous sod cover

Prairie land use provided the highest N loss reduction of any practice evaluated (94 ± 9%; Figure 53a). Studies that assessed N leaching from continuous sod and Conservation Reserve Program (CRP) land uses were deemed functionally similar to a prairie and were included in this category (Table 31). Notably, even when this land use was fertilized, there was almost no N leaching (Figure 53b: 95 ± 3%). The mean and median N losses across all 48 fertilized and unfertilized prairie/sod site-years were 1.3 ± 1.8 and 0.75 kg N/ha. The N leaching benefits of prairie and continuous sod have been known for decades (Bolton et al., 1970; Mitchell et al., 2000; Randall et al., 1997). Prairie restorations combined with wetland restoration on a large scale would provide many ecological benefits in the Minnesota landscape (e.g., water quality improvement, peak flood reduction; Cowdery et al., 2019).

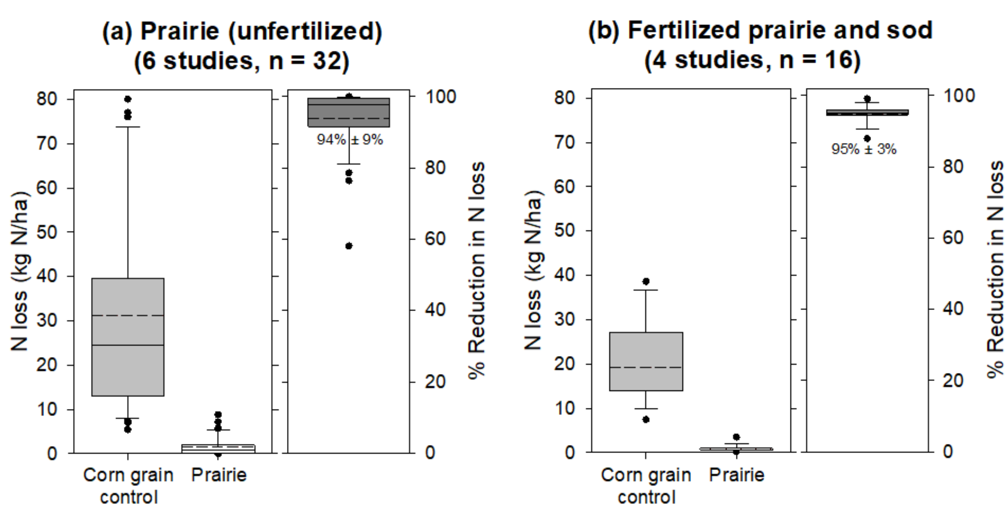

Figure 53. Annual nitrate leaching loss and loss reduction due to land conversion from a corn-soybean rotation to a prairie or unfertilized sod-type land use. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers.

Table 31. Prairie and unfertilized sod-type land use studies reporting annual nitrate leaching. C, corn, S, soybean.

Study	# site-years	Conventional crop (control)	Prairie land use	Location	Method and notes
Daigh et al. (2015)	4	CS average	Prairie (unfertilized)	Iowa	Replicated tile; aboveground biomass was harvested
Smith et al. (2013)	3	CSC	Prairie (restored, 28 species)	Illinois	Replicated tile
Hernandez-Ramirez et al. (2011)	6	CS average	Prairie (restored, predominantly Big Bluestem	Indiana	Replicated tile; 135 kg N/ha rate for CS-corn
Hussain et al. (2020)	7	Continuous corn	Prairie	Michigan	Suction lysimeter at 120 cm
Masarik et al. (2014)	8	Continuous corn	Mesic tall grass prairie	Wisconsin	Equilibrium tension lysimeter at 140 cm
Randall et al. (1997)	4	CS average	*"CRP"*	Minnesota	Replicated tile
Kuehner et al. (2020) *	5 (conc. only) *	*"Row crop"*	*"Prairie"*	Minnesota	Suction lysimeter at 122 cm
Daigh et al. (2015)	4	CS average	Prairie (fertilized)	Iowa	Replicated tile; aboveground biomass was harvested
Tan et al. (2002)	3	Continuous corn	Bluegrass sod (fertilized)	Ontario	Replicated tile
Bolton et al. (1970)	7	Continuous corn	Bluegrass sod (fertilized)	Ontario	Replicated tile
Drury et al. (1993)	2	Continuous corn	Kentucky bluegrass sod (fertilized)	Ontario	Replicated tile, conventional tillage

* not used in recommended average loss reduction values

The original Minnesota Nutrient Reduction Strategy recommended a N leaching reduction of 83% for the practice of conservation easements land retirement (MPCA, 2013) which was lower than the value developed here (94%). The Strategy recommended a N leaching reduction value of 95% for both hay land and bioenergy production land uses; that value was similar to the prairie value developed here but higher than those developed for bioenergy crops and hay land (below). Functionally, it makes sense that a prairie land use would have less N leaching (and thus higher N loss reduction efficiency) than more managed land uses where perennials are fertilized (e.g., bioenergy grasses and perennial forages for hay). The current recommendation is more consistent with the MPCA Watershed Pollutant Load Reduction Calculator that uses a N leaching reduction of 93% for the practice of conservation cover perennials (MPCA, 2022b).

Prairie strips

Eleven studies that discussed the practice of prairie strips were reviewed but only one provided suitable water quality data for this review (Zhou et al., 2014). Five site-years reported by (Zhou et al., 2014) result in average surface runoff nitrate, total N, and total P loss reductions of 66 ± 5%, 78 ± 16%, and 81 ± 14%, respectively, for the practice of prairie strips (medians: 64%, 77%, 80%, respectively).

The other water-related prairie strip studies in this review assessed runoff volumes (Hernandez-Santana et al., 2013); sediment or dissolved carbon in surface runoff (i.e., not N or P; Helmers et al., 2012a; Nelson, 2022; Smith et al., 2014; Stephenson et al., 2024); or nitrate in leachate but didn't provide an annual assessment (Zhou et al., 2010). Other studies have explored the impact of prairie strips on soil properties (Henning, 2022), the economics of this practice (Tyndall et al., 2013), or provided a general overview (Schulte et al., 2017). Kaspar et al. (2007) provided an early variation of the practice of prairie strips by assessing Gama grass strips over a tile drain, but that application was deemed functionally different from the current practice of prairie strips (i.e., NRCS Conservation Practice 43; USDA FSA, 2024).

The practice of prairie strips provides a myriad of environmental benefits beyond water quality improvement, whereas tradeoffs include 100% crop yield penalty for the area converted and additional management complexity (Schulte et al., 2017; Tyndall et al., 2013). Tyndall et al. (2013) estimated prairie strips cost $59 to $87 per treated hectare, and these costs can be significantly less if government programs are used (e.g., CRP). The nutrient treatment cost efficiency of this practice ranges from $1.60 to $2.30/kg N retained and $7.0 to $10.3/kg P retained in surface runoff (Tyndall et al., 2013). This practice is widely applicable in Iowa, with an estimate of 40% of Iowa's cropland suitable for implementation of prairie strips (Schulte et al., 2017). It is likely this practice would be similarly suitable (and beneficial) in Minnesota's landscape.

Hay land and pasture

Water quality data were extracted from six studies that involved hay forages and pasture land use (Table 32). Two of those were surface runoff studies (Harms et al., 1974; et al., 1973; See following section) and one study contained nitrate leaching concentrations rather than losses (Kuehner et al., 2020). The three remaining studies were all from Iowa. These 17 site-years resulted in an average nitrate leaching loss reduction of 74 ± 25% for a hay or pasture-type land use compared to a conventional row crop land use (Figure 54; median: 82%).

Table 32. Hay land and pasture land use studies reporting annual nitrate leaching.

Study	**# site-years**	**Conventional crop rotation (control)**	**Treatment land use details**	**Location**	**Method and notes**
Qi et al. (2011)	4	CS-average	"*Perennial forage*"; orchard grass	Iowa	Replicated tile plots
Waring et al. (2024)	10	CS-average	Orchard grass and clover	Iowa	Replicated tile plots
Cambardella et al. (2015)	3	CS-average	Organic pasture of bromegrass, fescue, alfalfa, and white clover	Iowa	Replicated tile plots
Kuehner et al. (2020) *	5 (conc. only) *	"*Row crop*"	"*Pasture*"	Minnesota	Suction lysimeter at 122 cm

* not used in recommended average N loss reduction values

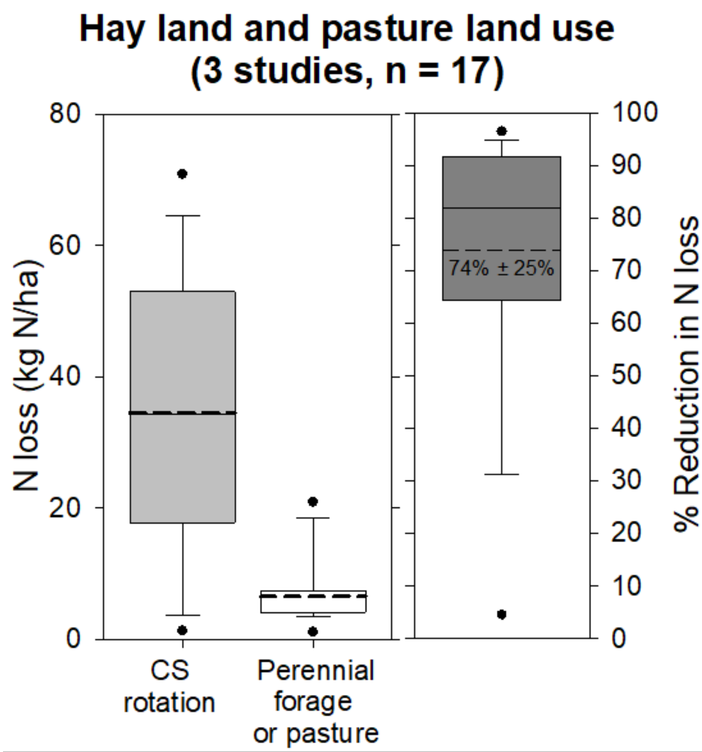

Figure 54. Annual nitrate leaching loss and loss reduction due to land conversion from a corn-soybean rotation to a hay or pasture land use. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 32.

The 5-y suction lysimeter study by Kuehner et al. (2020) showed pastures in southeast Minnesota provided a 77% nitrate concentration reduction compared to row crops (means: 5.1 vs. 22.3 mg NO_3-N/L). Goeken et al. (2015) reported that a perennial forage land use did not reduce the total drainage discharge over the drainage season compared to row crops, which may help account for the similarity between the 77% mean concentration reduction reported by Kuehner et al. (2020) and the mean 74% N loss reduction calculated in the current work. It is worth noting that while Goeken et al. (2015) found perennial forage did not reduce the total discharge volume over the entire season, their perennial treatment did significantly reduce drainage amounts in May which is a critical time for N loss when considering hypoxia in the Gulf of Mexico.

The original Minnesota Nutrient Reduction Strategy reported a 95% N reduction efficiency for the practice of "*hayland in marginal cropland (replacing row crops)*" (MPCA, 2013). Curiously, that value was higher (i.e., indicated more leaching reduction benefit) than the 83% reduction used for land retirement in Minnesota although hay land would likely be fertilized and more managed (MPCA, 2013). The original Iowa Nutrient Reduction Strategy used a value of 85% for "*grazed pastures*" but noted there was no pertinent information for the state and assumed a value similar to CRP (IDALS, 2014). Regardless, in Minnesota, there is potential for growth of this land conversion practice. Ecotone Analytics et al. (2023) forecasted an additional 1.7 million acres of perennial forage and pasture are achievable by 2050.

Bioenergy crops (switchgrass, miscanthus)

Eleven studies that investigated either switchgrass and/or miscanthus were reviewed and suitable water quality data were extracted from six (Table 33). Three of the six were excluded from analysis because they measured leaching at depths of 50 cm which was considered too shallow to reflect conditions beyond the root zone (Jungers et al., 2019; McIsaac et al., 2010; Studt et al., 2021) The remaining 23 site-years indicated this bioenergy land use provided a 61 ± 54% N leaching reduction compared to a corn-grain control (Figure 55a). There was one high outlier in the control dataset sourced from a study using equilibrium tension lysimeters (272 kg N/ha leached by the continuous corn control, Stenjem et al. (2019); outlier not shown for axis clarity in Figure 55a and c). However, the resulting reduction efficiency for that site-year of 92% was not notably different from the other site-years.

Table 33. Bioenergy crop land use studies reporting annual nitrate leaching losses. C, corn; S, soybean

Study	**# site-years**	**Conventional crop (control)**	**Bioenergy crop**	**Location**	**Method and notes**
Smith et al. (2013)	3	CSC	Switchgrass	Illinois	Replicated tile;
Smith et al. (2013)	3	CSC	Miscanthus (unfertilized)	Illinois	Replicated tile;
Hussain et al. (2020)	7	Continuous corn	Switchgrass	Michigan	Suction lysimeter at 120 cm
Hussain et al. (2020)	7	Continuous corn	Miscanthus	Michigan	Suction lysimeter at 120 cm
Stenjem et al. (2019)	3	Continuous corn	Switchgrass	Wisconsin	Equilibrium tension lysimeter averaging at 100 cm
Jungers et al. (2019) *	3	Continuous corn	Switchgrass	Minnesota	Suction lysimeter at 50 cm
Studt et al. (2021) *	16	Continuous corn	Miscanthus	Iowa	Ion exchange resin lysimeter at 50 cm
McIsaac et al. (2010) *	4	CS average	Miscanthus	Illinois	Suction lysimeter at 50 cm
McIsaac et al. (2010) *	3	CS average	Switchgrass	Illinois	Suction lysimeter at 50 cm

* not used in recommended average N loss reduction value

A nuance with this particular perennial land use is the relatively lower N leaching benefit during the bioenergy grass establishment period (Christian and Riche, 1998; McIsaac et al., 2010; Smith et al., 2013; Studt et al., 2021). This effect is very apparent in the data extracted for this review when comparing the first 2 years after establishment versus year 3 and older (N reduction means: -12% versus 86%, respectively; Figure 55b and c). Nevertheless, even considering these early years, this practice still provides an overall N leaching benefit compared to a corn-based system over the long term (Hussain et al., 2019) (e.g., Figure 55a).

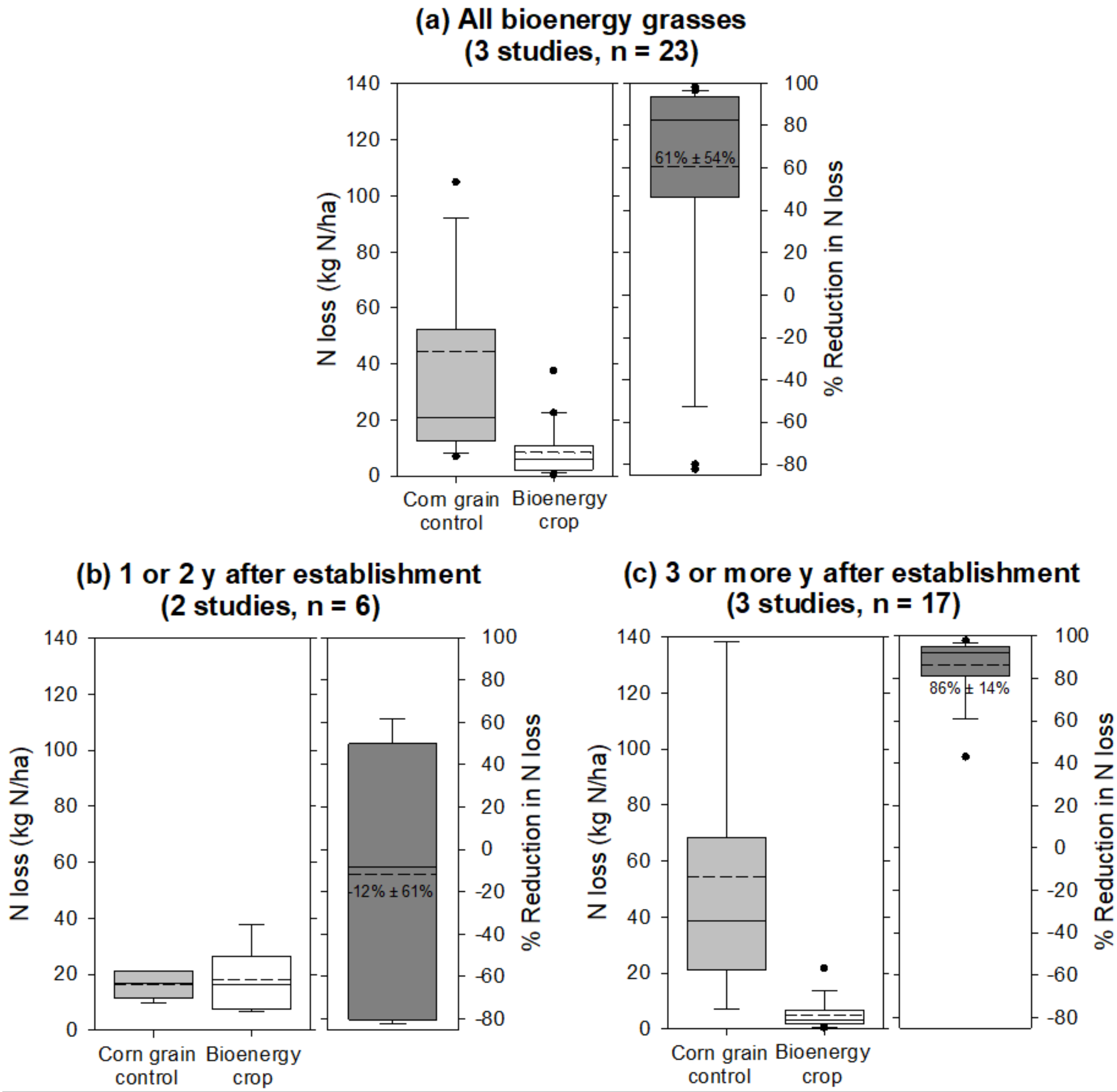

Figure 55. Annual nitrate leaching loss and loss reduction due to land conversion from a corn grain-based rotation to a bioenergy crop (a). The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 33.

The Minnesota Nutrient Reduction Strategy originally used a value of 95% N loss reduction for this practice ("*perennial energy crops*"; MPCA, 2013). The original Illinois Nutrient Loss Reduction Strategy similarly used 90% N leaching reduction for "*perennial/energy crops*" (IDOA, 2015). These values are similar to the 86% mean calculated here for the established and mature bioenergy grasses (Figure 55c). However, the newly recommended overall 61 ± 54% reduction (developed here; Figure 55a) is more conservative, possibly due to its inclusion of the initial growth years. This large decrease in effectiveness is notable, but it should also be recognized that Minnesota currently has very few acres of switchgrass and miscanthus in production for bioenergy. Thes recommended value is also relatively aligned more with the 72 ± 23% N loss reduction originally reported in the Iowa Nutrient Reduction Strategy (IDALS, 2014).

Land use change perennials and phosphorus leaching loss

Leaching of P was relatively less studied than N for these land use change practices (Figure 45a and b). However, there were sufficient site-years to develop dissolved P leaching reduction values for three land use practices (extended rotation and conversion to either prairie or bioenergy grasses; Table 34). There were sufficient data to develop one recommended total P leaching value for a combined set of land use practices (combination of extended rotation and sod site-years, *n* = 21; Table 34).

Table 34. Land use change studies reporting annual dissolved or total phosphorus (DP, TP) leaching losses. DP was reported as dissolved reactive P (DRP) unless noted. C, corn; S, soybean; O, oats; A, alfalfa.

Study	**# site-years for given leached nutrient**		**Cash crop (control)**	**Land use change practice**	**Location**	**Method and notes**
	DP	TP				
Oquist et al. (2007)	3	3	"*conventional*"; CS-average	"*alternative*"; CSOA	Minnesota	Edge of field tile; orthophosphate reported
Zhang et al. (2015)	8	8	Continuous corn	Extended rotation: COAA	Ontario	Replicated tile; fertilized and unfertilized treatments; one DP outlier removed
Culley et al. (1983)	4	4	Continuous corn	Extended rotation: COAA	Ontario	Replicated tile; TDP reported
Daigh et al. (2015)	4	---	CS average	Prairie	Iowa	Replicated tile; unfertilized; TRP reported
Brye et al. (2002)	2	---	Continuous corn	Prairie	Wisconsin	Equilibrium tension lysimeter at 140 cm; MRP reported
Hernandez-Ramirez et al. (2011)	2	---	CS average	Prairie	Indiana	Replicated tile; Orthophosphate reported
Hussain et al. (2021)	7	---	Continuous corn	Prairie	Michigan	Suction lysimeter at 120 cm; TDP reported
Zhang et al. (2015)	4	4	Continuous corn	Prairie-type: Kentucky Bluegrass	Ontario	Replicated tile; unfertilized; one DP outlier removed
Culley et al. (1983)	2	2	Continuous corn	Prairie-type: bluegrass	Ontario	Replicated tile; unfertilized; TDP reported
Hussain et al. (2021)	7	---	Continuous corn	Bioenergy: Switchgrass	Michigan	Suction lysimeter at 120 cm; TDP reported
Hussain et al. (2021)	7	---	Continuous corn	Bioenergy: Miscanthus	Michigan	Suction lysimeter at 120 cm; TDP reported
Stenjem et al. (2019)	3	---	Continuous corn	Bioenergy: Switchgrass	Wisconsin	Equilibrium tension lysimeter averaging at 100 cm

Annual P leaching losses tend to be small, roughly an order of magnitude less compared to P losses in surface runoff. For example, the means from the corn-based control were less than 120 g dissolved P/ha for leaching (gray control bars in Figure 56a-c) versus a mean of greater than 1000 g dissolved P/ha for surface runoff (Figure 58b). Regardless, land use change practices often increased dissolved P leaching losses, especially for the practices of extended rotation and prairie (Figure 56a and b). These two practices had relatively large, albeit variable, dissolved P increases averaging 108 ± 155% and 55 ± 119%, respectively. Their medians also illustrated a negative relative impact (median increases: 57% and 35%, respectively).

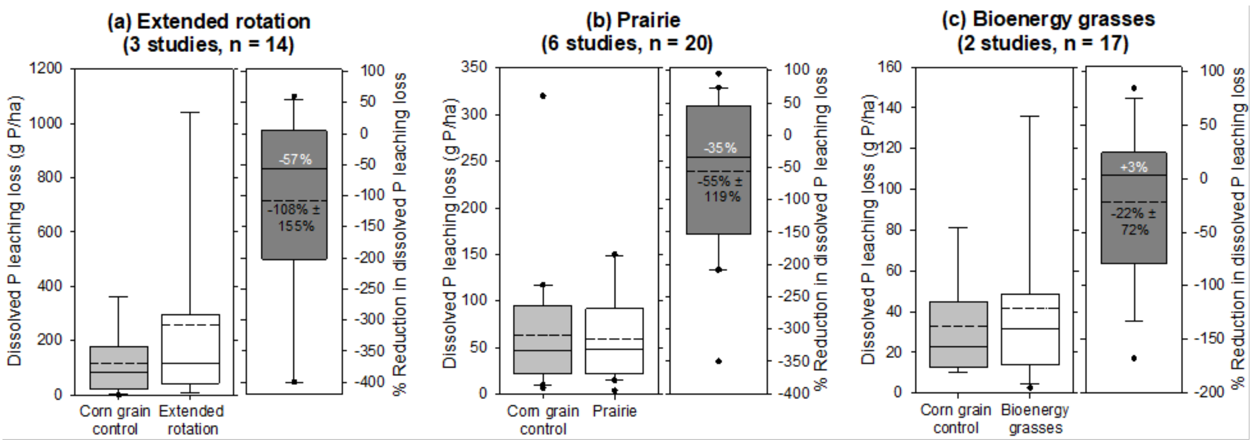

Figure 56. Annual dissolved phosphorus leaching loss and loss reduction due to land conversion from a corn-based rotation to either an extended rotation (a), prairie-type (b), or bioenergy grass (c) land use. A negative % reduction represents an increase in dissolved phosphorus loss due to the practice. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 34.

Although this relative impact was consistent across the prairie dataset (both mean and median % values were negative), these dissolved P data were especially variable. Three of the six studies only demonstrated a benefit (mean dissolved P reduction: 52%, n = 8; Daigh et al., 2015; Brye et al., 2002; Hernandez-Ramirez et al., 2011). The other three studies showed only a strong negative effect of this practice (mean: -127%, n = 12; Hussain et al., 2021; Zhang et al., 2015; Culley et al., 1983).

Conversion of a corn-grain cropping system to bioenergy grasses increased the dissolved P leaching losses slightly less by 22 ± 72% with a median that indicated a small benefit (median decrease: 3%). This directional difference between the mean (negative) and median (positive) for the conversion to bioenergy grasses was one of the few instances where the median was recommended as the overall nutrient efficiency rather than the mean (see also surface runoff dissolved P impacts for cover crops).

Grouping the land uses conversions together for the total P analysis resulted in these perennials increasing leaching by an average of 94 ± 134% (Figure 57; median increase: 71%).

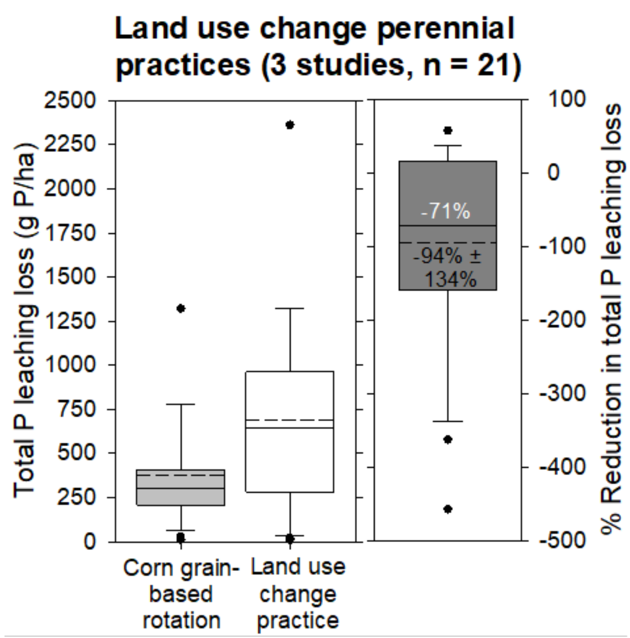

Figure 57. Annual total phosphorus leaching loss and loss reduction due to land conversion from a corn-based rotation to a perennial land use practice. A negative % reduction represents an increase in dissolved phosphorus loss due to the practice. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 34.

Land use change perennials and runoff nutrient losses

The available data supported the development of one recommended value to represent all of the land use change perennials for reduction of each nutrient in surface runoff (i.e., one value each for runoff nitrate, dissolved P, and total P loss reduction). Data representing prairie strips (e.g., Zhou et al., 2014) or vegetated buffers were excluded because the application of those targeted practices fundamentally differed from the practice of a comprehensive land use change as defined in this chapter (Table 35). The surface runoff P data available for this analysis were especially limited (e.g., n ≤ 8; Table 35). This was a notable gap given the relatively high scientific consensus that perennials reduce many pollutants in surface runoff (Kaspar et al., 2008).

Rainfall simulations studies generally did not present the annual values necessary for this systematic review and analysis. These studies nevertheless provide important context for the impact these perennials have on pollutant losses during storm events (e.g., Kura clover: Grabber and Jokela, 2013; vegetated filter strips: Katuwal et al., 2023; Vanrobaeys et al., 2019; alfalfa: Osterholz et al., 2019; Wendt and Corey, 1980; Zemenchik et al., 2002). Due to the shortage of data, a study that performed a series of rainfall simulations was included in the analysis (summation of 5 simulations; Siller et al., 2016).

Table 35. Land use change studies reporting annual surface runoff losses from natural rainfall (primarily). NO_3, nitrate; DP, dissolved phosphorus; TP, total phosphorus; C, corn; S, soybean.

Study	# site-years for given nutrient			Cash crop rotation (control)	Land use change practice	Location	Method and notes
	NO_3	DP	TP				
Zhu et al. (1989)	2	2		Soybean	Canada bluegrass	Missouri	3% slope; treatment used as a perennial cover crop; phosphate reported
Thomas et al. (1992)	1	1		Continuous corn	Alfalfa	Ohio	3.5-4.0% slope; 11 events during the growing season; phosphate reported
Drury et al. (1993)	3			Continuous corn	Kentucky bluegrass	Ontario	0.5% slope;
Harms et al. (1974)	4		4	Corn, oats	Alfalfa & bromegrass; pasture	South Dakota	“Cultivated land” control
Schuman et al. (1973)	3			Corn	Bromegrass	Iowa	2-18% slope; Treatment rotationally grazed; Watershed 2 versus 3
Eleki (2003)		1	1	Corn	Kura clover	Wisconsin	7.7-8.8% slope; Summed 9 reported events of 17 total for an annual value
Bormann et al. (2012)		2	2	Corn	Alfalfa with bromegrass	Wisconsin	5% average slope; DRP reported; Small plot and field;
Siller et al. (2016)	1	1	1	Corn silage	Kura clover	Wisconsin	Sum of 5 rainfall simulations; 8-15% slope; DRP reported
Zhou et al. (2014) *	5		5	CS average	Prairie strips	Iowa	Excluded due to a fundamentally different practice application (strips)

* not used in recommended average values

Overall, these land use change perennials resulted in an average surface runoff nitrate reduction of 52 ± 40%, dissolved P reduction of 21 ± 25%, and total P reduction of 46 ± 26% (Figure 58a-c; compared to a corn-based control). The positive medians confirmed these benefits (64, 11 and 51%, respectively).

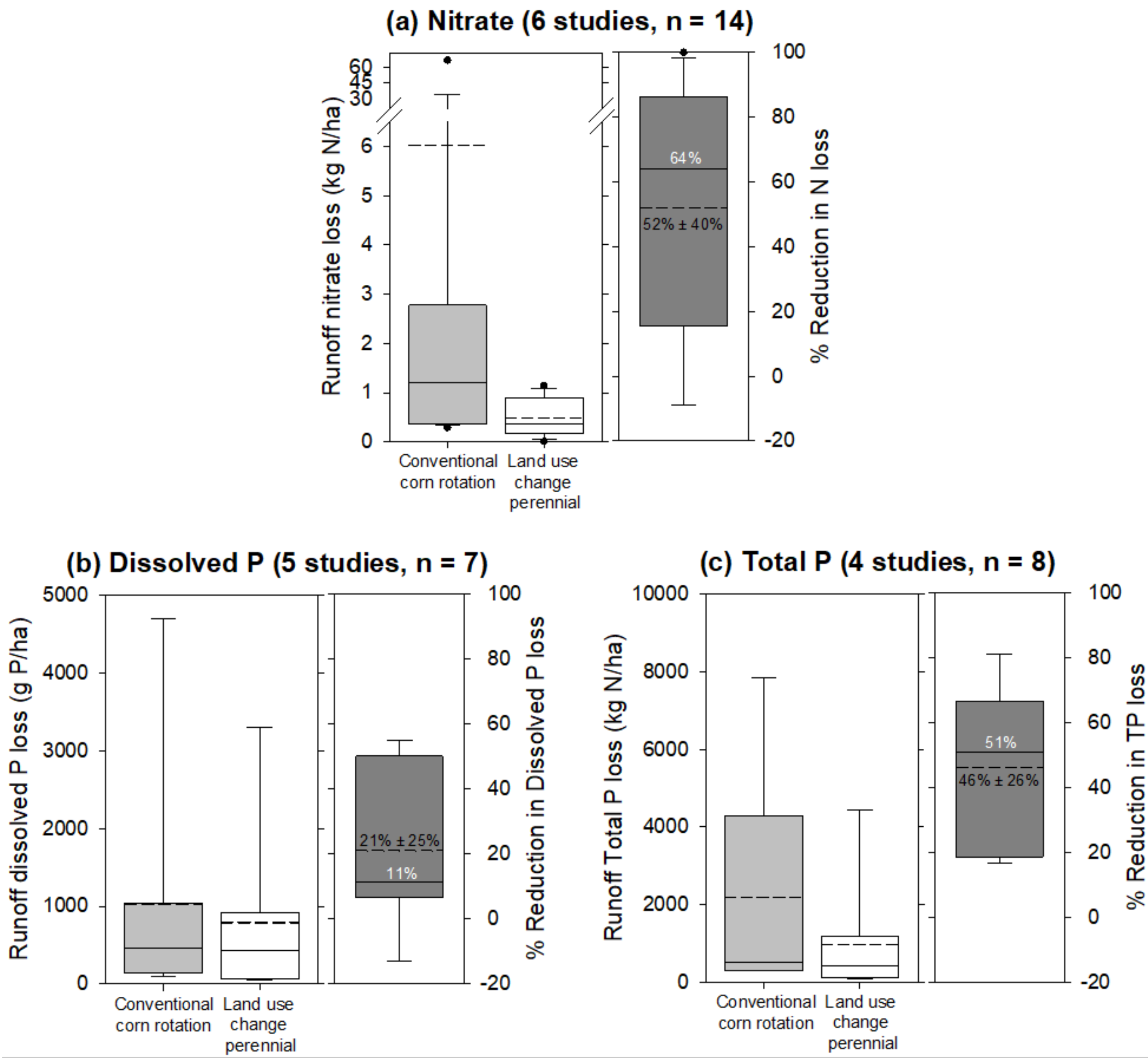

Figure 58. Surface runoff nitrate (a), dissolved phosphorus (b), and total phosphorus (c) losses and loss reductions due to land use change perennials. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were sourced from studies listed in Table 35.

One note about this analysis was that these surface runoff studies tended to be older than the studies for N leaching. For example, the median monitored year for the surface runoff site-years was 1986. The median years for N leaching monitored in extended rotations and prairie were 1998 and 2003, respectively. Given the few site-years especially for P in runoff, it may be useful to revisit questions of annual runoff nutrient losses from perennial land uses. The comprehensive analysis by Ecotone Analytics et al. (2023), which summarized the benefits of continuous living cover crops for Minnesota, specifically excluded consideration of P loss because of this lack of data. While these current data indicated consistent surface runoff P benefits for perennial land uses, the limited data, associated data uncertainty, and possibility for P loss tradeoffs in cold climates (e.g., Kleinman et al., 2022) needs to be acknowledged.

The original Minnesota Nutrient Reduction Strategy did not differentiate between surface runoff and leaching for the published N and P reductions provided by conservation practices. That original document reported a 34% P reduction for "*perennial energy crops*", a 59% P reduction for "*hay land in marginal cropland (replacing row crops)*", and a 56% P reduction for "*conservation easements and land retirement*" (MPCA, 2013). The newly recommended value of 46% reduction for total P in surface runoff is within the range of these older values. The MPCA Watershed Pollutant Load Reduction Calculator uses a value of 84% total P reduction for surface runoff for the practice of conservation crop perennials, which is beyond the upper range of the data documented here (Figure 58c).

Land use change perennials P impacts in above- vs. below-ground pathways

It's useful to put the magnitude of the P leaching and surface runoff impacts due to land use change perennials in context of each other (as was previously done for cover crops). For example, these practices provided a mean 46% reduction in surface runoff total P loss but a 94% increase in total P leaching loss (Table 23). So how do loss pathways compare in terms of units of loss (g P/ha) rather than relative loss reduction (%)?

Dissolved and total P losses both show the same trends when comparing pathways when a land use change is done (versus a corn grain cropping system; Figure 59). More specifically, in this analysis, land use change to perennials reduced losses of both dissolved and total P in surface runoff whereas both forms of P in leachate tended to increase. However, assessment of the magnitude of these above- and below-ground losses showed a net benefit of this practice. For example, the mean dissolved P leaching and surface runoff impacts were -47 and +236 g P/ha, respectively, where a negative value indicated the perennial increased P loss in that given pathway (Figure 59, dashed lines on two left bars). Summing these two means resulted in a net positive dissolved P impact due to conversion to perennials (-47+236 = +189 g P/ha loss reduction) relative to a row crop system. For total P, the mean leaching and surface runoff impacts were -311 and +1206 g P/ha. Their sum also yielded a positive impact (-311+1206 = +885 g P/ha loss reduction).

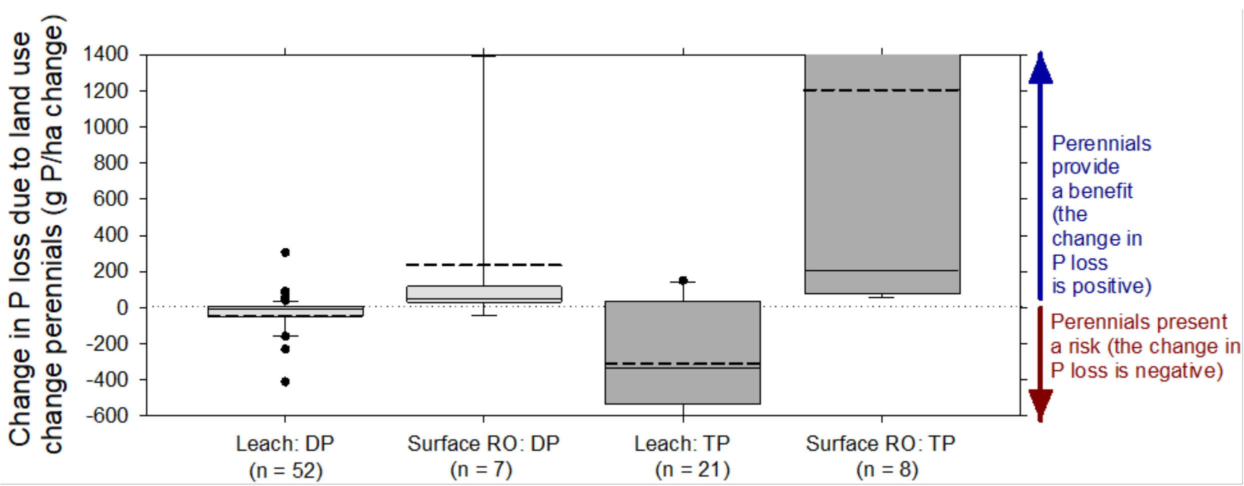

Figure 59. Phosphorus (P) loss impact due to the presence of a land use change perennial for leaching and surface runoff losses and for dissolved P (DP) and total P (TP). The convention of the y-axis was: "corn control P loss – perennial P loss" where a positive value indicated the land use change perennial practice reduced the loss and a negative value indicated the perennial practice increased the loss. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Y-axis was scaled to the interquartile ranges to show detail of the means and medians.

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Chapter 6: Conservation drainage practices

Associated NRCS practice codes:

- CPS #606 Subsurface drain
- CPS #587 Structure for water control
- CPS #554 Drainage water management
- CPS #605 Denitrifying bioreactor
- CPS #604 Saturated buffer
- CPS #620 Underground outlet
- CPS #447 Irrigation and drainage tailwater recovery

Definition: The concept of "*conservation drainage*" involves the use of practices that allow the prudent and necessary use of artificial drainage while also reducing the associated negative environmental impacts. Conservation drainage practices achieve this either by: (1) altering the design or management of a drainage system or (2) treating nutrients at the edge of the field.

Values in the original Minnesota Nutrient Reduction Strategy for N reduction:

- Controlled drainage: 33-44%
- Bioreactors: 13%
- Constructed wetlands: 50%

Table 36. Recommended nutrient reduction efficiencies for conservation drainage practices.

	-------------- Leaching -----------			--------- Surface runoff---------		
	Nitrate	**Dissolved P**	**Total P**	**Nitrate**	**Dissolved P**	**Total P**
Controlled drainage	45 ± 27%	30 ± 40%	30 ± 29%	---	---	---
Saturated buffers	43 ± 26%	Insufficient data	Insufficient data	See prairie conversion runoff values (applies to saturated buffer area, not drainage area)		
Denitrifying bioreactors	30 ± 21%	1% † Extrapolated	Insufficient data	---	---	---
Shallow drainage	41 ± 24%	Insufficient data	Insufficient data	---	---	---
Gravel / blind inlets	-60 ± 93% Very limited data	43 ± 29%	41 ± 31%	---	---	---
Drainage water recycling	51 ± 21% limited data	-1 ± 80% limited data	22 ± 57% limited data	---	---	---

† Recommended the median value rather than mean ± standard deviation due to inconsistent impacts of those two metrics (positive or neutral median versus negative mean)

Highlight bullets:

- Conservation drainage practices provide targeted, effective, and consistent N loss reduction for Minnesota's 8.25 million tile-drained acres (30-51% N loss reduction).
- These structural practices often require government assistance. Alternative conservation delivery mechanisms that accelerate their implementation should be considered.
- Limited data and confounding study design/methods creates uncertainty for several of the newer practices (alternative inlets, drainage water recycling) but they are important to increasingly consider in Minnesota's conservation toolbox.

Conservation drainage background and database overview

Minnesota ranks third in the United States in tile-drained area with 3.3 million hectares (8.25 million acres) of improved drainage that are located across more than 19,000 operations (USDA NASS, 2022). Many of these farms are within the Mississippi River Basin (i.e., south central and southwest Minnesota), although the Red River Valley is one of the few locations in the Midwest where agricultural land is newly being tiled (Jame et al., 2022). Like much of the Midwest, original drainage improvements in Minnesota were done between 1860 and 1920 during initial European settlement (Leitch and Kerestes, 1981). Even after a century of drainage works, Minnesota farmers still perceive that nearly 20% of land would benefit from further drainage improvement (Carlson, 2014). These infrastructure projects at both the field- and drainage district-scale are important opportunities to increasingly consider water quality (Lewandowski, 2010).

The agronomic benefits of artificial subsurface drainage are well established (Evans and Fausey, 1999; Maas et al., 2022; Paiao et al., 2021; Skaggs and van Schilfgaarde, 1999). Beyond providing proper soil aeration for roots and timely trafficability, drained soils warm faster which is a vital benefit in Minnesota's northern climate (Jin et al., 2008). Improved subsurface drainage lowers the water table and thus can provide water storage in drained soil pore spaces (Skaggs et al., 1994). This often results in reduced surface runoff, sediment-bound pollutants, and peak outflow rates compared to undrained conditions on the same land use (Blann et al., 2009; Gilliam et al., 1999). The climate benefits of artificially improved drainage are becoming newly acknowledged (Fabrizzi et al., 2024; Fernández et al., 2016).

Despite these agronomic and environmental benefits, the loss of dissolved nutrients, primarily nitrate, through tile drainage systems is a major tradeoff (Gillam et al., 2008; Skaggs et al., 1994). The concept of "*conservation drainage*" attempts to balance the agronomic necessity of improved drainage in humid climates with reduced (or, even improved) environmental impacts. This suite of drainage-specific practices includes those that alter the design or management of drainage systems (controlled drainage, shallow drainage, blind inlets) and those that treat nutrients at the edge of the field (saturated buffers, bioreactors, drainage water recycling, wetlands).

Since these are considered a "practice of engineering," many conservation drainage projects require a professional engineer to sign off on critical aspects of the project, including design and implementation. Conservation drainage practices can benefit both farmers and society. A few of these practices can provide a yield benefit (drainage water management; controlled drainage) or make farming easier (blind inlets). These practices generally do not require a change in in-field operations, which is likely desirable by many landowners (Christianson et al., 2018). Conservation drainage practices also stack well with other practices (Christianson et al., 2018). This is especially notable given that good drainage is required for the success of other conservation practices like cover crops (Kladivko, 2020). Finally, by providing efficient nitrate treatment at the edge of the field, these conservation drainage practices can provide a climate benefit by reducing indirect nitrous oxide emissions downstream (Frankenberger et al., 2024).

Most importantly for this review, conservation drainage practices provide effective N removal with relatively high certainty (i.e., high mean reduction, low coefficient of variation). It follows that these practice's cost efficiencies (\$/kg N treated) tend to be favorable (Christianson et al., 2013), in part due to the long-term nature of these structural practices (TNC, 2021).

Large capital costs and biophysical siting limitations (e.g., slope for controlled drainage; soil properties for saturated buffers) present obvious barriers to the adoption of conservation drainage practices across the Midwest (David et al., 2014; Dentzman et al., 2022; Lewandowski, 2010). However, the most difficult barriers to overcome may be systemic and institutional. Firstly, these practices present many broadly defined educational needs. Conservation drainage generally focuses on nitrate which is "*invisible*" compared to sediment, the main culprit of the past century. Increased acknowledgement that nitrate will be lost through tile drains in corn-based cropping systems is a fundamental outreach need.

A second educational opportunity is that conservation drainage practices are all new (as is the term itself; Lewandowski, 2010). "*Grassed waterways*" and "*sediment control basins*" are familiar terminology within the conservation community but "*bioreactor*" still sounds relatively foreign. Edge-of-field practices may even be viewed as "*black sheep*" (TNC, 2021). More field demonstrations to normalize these new practices would be helpful (Lewandowski, 2010). Going deeper, hands-on technical training to expand human capacity to design and construct these practices is a key gap. From a higher education perspective, curriculum that includes conservation drainage is needed at universities across the Midwest to create the next generation of professionals well versed in the nuances of N loss (Frankenberger et al., 2023a).

Beyond these educational needs, institutional barriers related to government programs are frequently cited as one of the most important (e.g., burdensome program requirements, onerous paperwork, "*red tape*"; Dentzman et al., 2022; Lewandowski, 2010; TNC, 2021). Unfortunately, the capital cost of many of these practices often necessitates some type of government assistance. The structural nature of these practices also increases complexity and the lead time to get a project in the ground (e.g., site surveying, design work).

Taken altogether this means that alternative conservation delivery mechanisms should be considered to increase adoption of these "*crucial but underutilized*" practices (TNC, 2021). **Iowa's "*Batch and Build*" approach** provides one example of how conservation efforts can rapidly accelerate toward ambitious 21st century water quality goals. Batching the site selection, design, and construction for a number of local practice installations creates an economy of scale for both engineering firms and contractors within a relatively local area (e.g., an N-impaired watershed). Moreover, the use of a single "*fiscal agent*" to coordinate both receiving and paying out funding removes this burden from across the various landowners (Johnson et al., 2023; Maxwell et al., 2024). Rather than landowners being (partially) reimbursed for project construction costs afterwards, they can receive a one-time payment for a temporary conservation easement to allow site access for construction. This approach streamlines conservation delivery for farmers and reduces the total public expenditure (Kult, 2022). A similar fledgling approach trialed in Minnesota merits additional assessment ("*Turn-Key Project*"; ADMC, 2021).

Accelerating research on, outreach about, and implementation of conservation drainage practices has never been more important. ***It is now clear that commonly promoted in-field conservation approaches will be insufficient on their own to achieve water quality goals in the Mississippi River Basin (Feyereisen et al., 2022; McLellan et al., 2015; TNC, 2021)***. The practices assessed in this chapter need to be an increasingly central part of the conservation toolbox in the heavily tile-drained state of Minnesota. It is worth reiterating, however, that all practices will be needed, and no one set of practices will be sufficient on its own (Feyereisen et al., 2022).

More than 300 tile drainage-specific and conservation drainage studies were reviewed for this science assessment. Of these, nearly 60 studies contained water quality data which were extracted to compile 367 water quality site-years for six practices (Figure 60c). These site-years focused on leaching/ subsurface pathways (Figure 60a) although surface runoff was discussed in several sections (controlled drainage, saturated buffers, drainage water recycling). The final studies used were primarily from the Corn Belt region (Figure 60b). In general, total N data were not extracted from these studies because nitrate tended to be reported. Finally, the control-treatment comparison for these site-years was not always consistent with the "*no practice*" versus "*practice*" convention. More specifically, the "*control*" for saturated buffers and bioreactors was the nutrient loss from the field just upstream of the practice considering both inflow to the practice and untreated bypass flow.

There was insufficient time to assess several practices that would have otherwise been in this chapter: wetlands (constructed and restored); P filters; and two-stage ditches. In particular, wetlands need to be quantitatively assessed for Minnesota state nutrient strategy purposes. The Iowa Nutrient Reduction Strategy originally recommended a N reduction of 52% for constructed wetlands but this has recently been revised to 30% (IDALS, 2024). The MPCA Watershed Pollutant Load Reduction Calculator uses a 52% N reduction, identical to the now outdated value from Iowa (MPCA, 2022). The nutrient reduction benefit of wetland restoration versus constructed wetland creation should be quantified because wetland N removal is a function of hydraulic loading (Crumpton et al., 2020). However, wetland restoration, which comprises much of the wetland activity in Minnesota, is generally not targeted to treat drainage waters (MPCA, 2020). A section on wetlands was added as supplementary information to further address this gap. Wetlands are an essential feature in Minnesota's landscape (e.g., Cowdery et al., 2019; Gordon et al., 2021; Lenhart et al., 2016), and deeper review of their benefits (both constructed and restored, for both N and P) would be valuable.

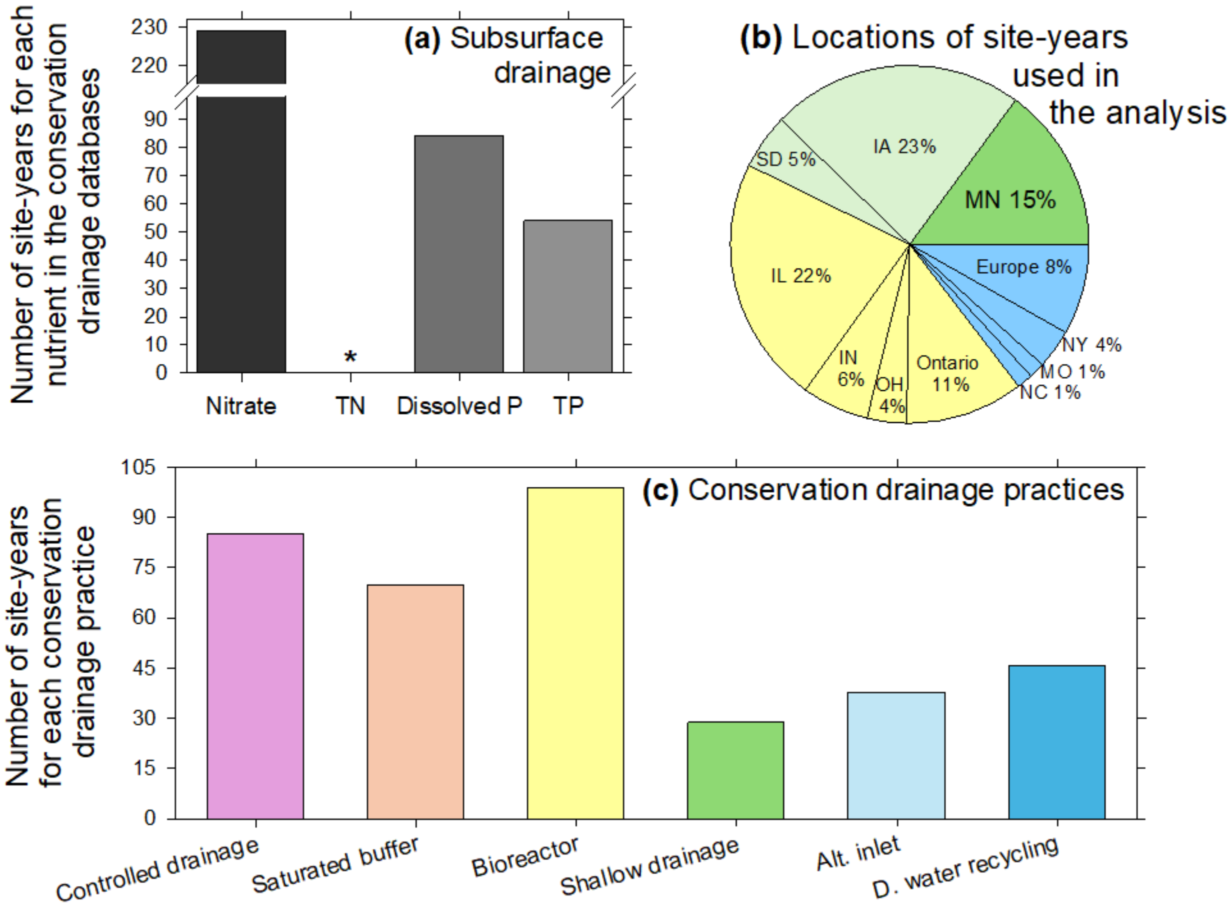

*Figure 60. Description of the 367 water quality site-years compiled for conservation drainage practices: (a) the site-year count by nutrient type; (b) site-year locations; and (c) the site-year count for each practice. *Two drainage water recycling studies reported total nitrogen (TN), but these were grouped with nitrate.*

Controlled drainage

The practice of controlled drainage (also known as drainage water management; MN NRCS, 2022a) involves use of control structures to manage the elevation of the drainage outlet during times of the year that higher drainage intensity is not necessary (Skaggs et al., 2012). This practice involves both infrastructure (generally a series of structures) and management activities and is applicable for both drainage ditches and subsurface tile drainage systems (Gilliam et al., 1979). Management of drainage in this way directly aligns with the Golden Rule of Drainage: "*Drain what is necessary for good crop growth and trafficability and not one drop more*" (attributed to R.W. Skaggs; Lewandowski, 2010). Controlled drainage has been reviewed several times (Carstensen et al., 2020; Cooke et al., 2008; Hoffmann et al., 2020; Ross et al., 2016; Strock et al., 2011), most recently as a part of the Indiana Science Assessment (Frankenberger et al., 2023b) and the Transforming Drainage project (Helmers et al., 2022).

Forty-eight controlled drainage studies were included in this quantitative review, but the overall synthesis was built from the drainage water management database used for the Indiana Science Assessment (Frankenberger et al., 2023b). That global review consisted of 140 site-years of paired free versus controlled drainage N loss comparisons. Two additions were made to the Indiana database (Chandrasoma et al., 2022b; Feser et al., 2010) and then the data were sorted to best reflect Minnesota's climate. Of the resulting 38 site-years used here, 19 were from Minnesota with 15 and 4 from Iowa and South Dakota, respectively (Table 37).

Table 37. Controlled drainage nitrate loss studies relevant for Minnesota's climate used in this review. Most were originally sourced from the Indiana Science Assessment (Frankenberger et al., 2023b).

Study	# site-years versus free drainage	Location	Notes
Jaynes (2012)	4	Iowa	Replicated drain line plots
Chighladze et al. (2021); Helmers et al. (2012); Schott et al. (2017)	11	Iowa	Replicated plots, multiple studies from Crawfordsville, IA
Chighladze et al. (2021)	10	Minnesota	Transforming Drainage database
ADMC (2012)	2	Minnesota	Dundas
ADMC (2012)	2	Minnesota	Hayfield
ADMC (2012)	2	Minnesota	Wilmont
ADMC (2012)	1	Minnesota	Windom
Feser et al. (2010)	2	Minnesota	Private farm, Tracy, MN; may be duplicated with the TD site above but values differed.
Sahani (2017)	2	South Dakota	Thesis
Sharma (2018)	2	South Dakota	Thesis

The practice of controlled drainage provided an average of 45 ±27% N loss reduction versus a free drainage control in this Minnesota-focused review (Figure 61a; median: 41%). This equated to an average annual loss reduction of 11.6 ± 10.9 kg N/ha (median: 9.8 kg N/ha). These results were notably similar to the 46% average (95% confidence intervals: 37.4 - 54.5%) reported by Frankenberger et al. (2023b) despite the difference in sample size (n = 38 versus 140).

The original Minnesota Nutrient Reduction Strategy used a range of 33-44% N loss reduction for this practice (MPCA, 2013). The newly recommended 45% N loss reduction was slightly higher than that range but was closely aligned with 43% total N reduction for tile drainage used in the MPCA Watershed Pollutant Load Reduction Calculator (MPCA, 2022). The original N loss reduction of 33 ± 32% used in the Iowa Nutrient Reduction Strategy recently has been increased to 40 ± 28% (IDALS, 2024).

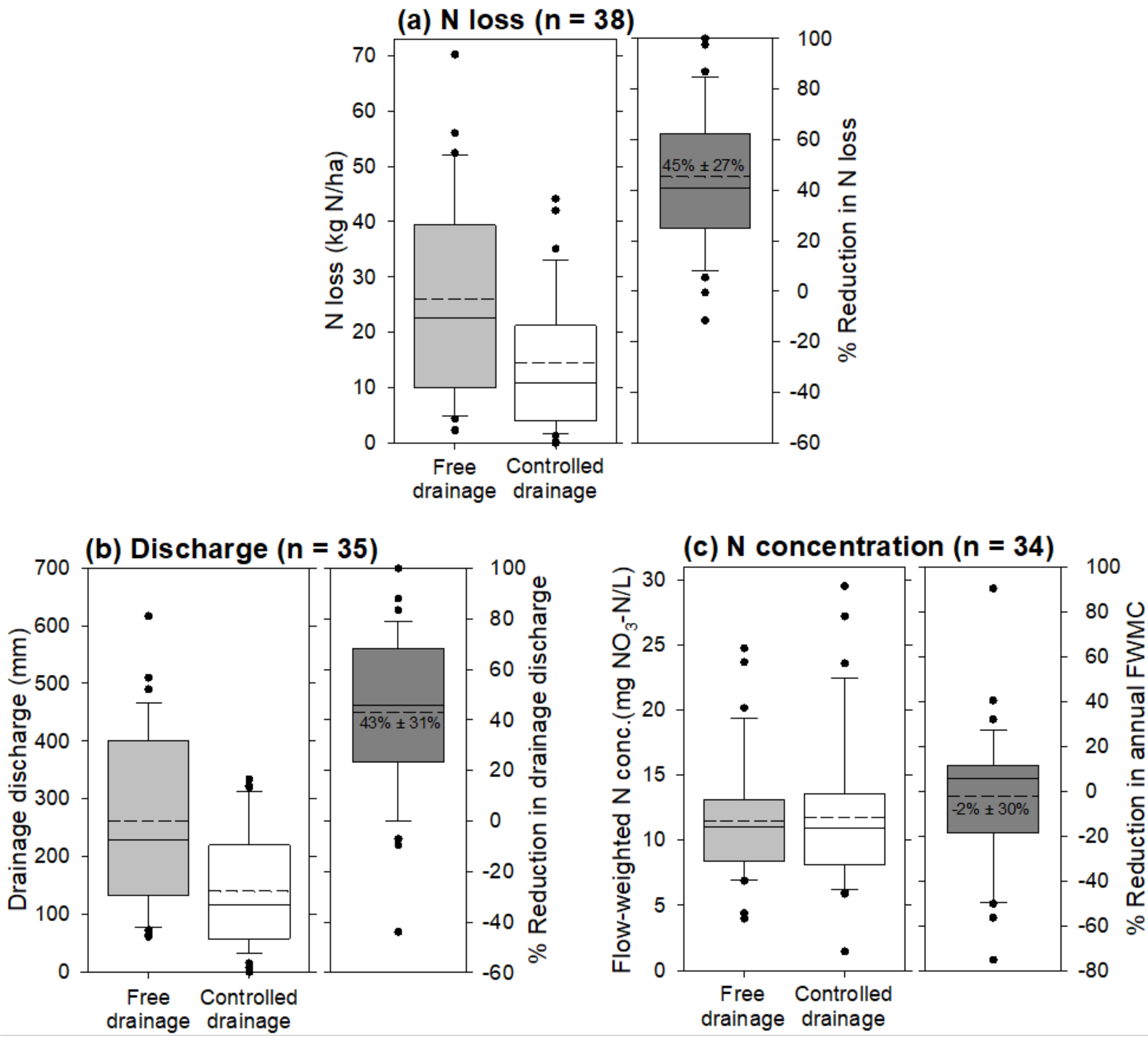

Figure 61. Nitrate-N loss and loss reduction provided by the practice of controlled drainage (a) along with the drainage discharge (b) and annual flow-weighted nitrate concentration (FWMC, c) impacts of this practice. Sites were in Minnesota, Iowa, and South Dakota (Table 37). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers.

The N loss reduction provided by controlled drainage is strongly related to this practice's reduction in drainage discharge rather than a change in nitrate concentration (Carstensen et al., 2019; Helmers et al., 2022; Luo et al., 2010; Ross et al., 2016; Williams et al., 2015a). For example, in the current dataset, the average reduction in drainage discharge provided by this practice was 43 ±31% (Figure 61b) whereas the reduction in annual flow-weighted nitrate concentration was nearly zero ((Figure 61c: -2 ±30%). Illustrated differently, the relationship between drainage discharge reduction and N loss reduction was strong both in relative terms (Figure 62a) and per unit (mm or kg N/ha, Figure 62c). However, there was no correlation between the reduction of N loss and changes in annual flow-weighted concentrations (Figure 62b and d).

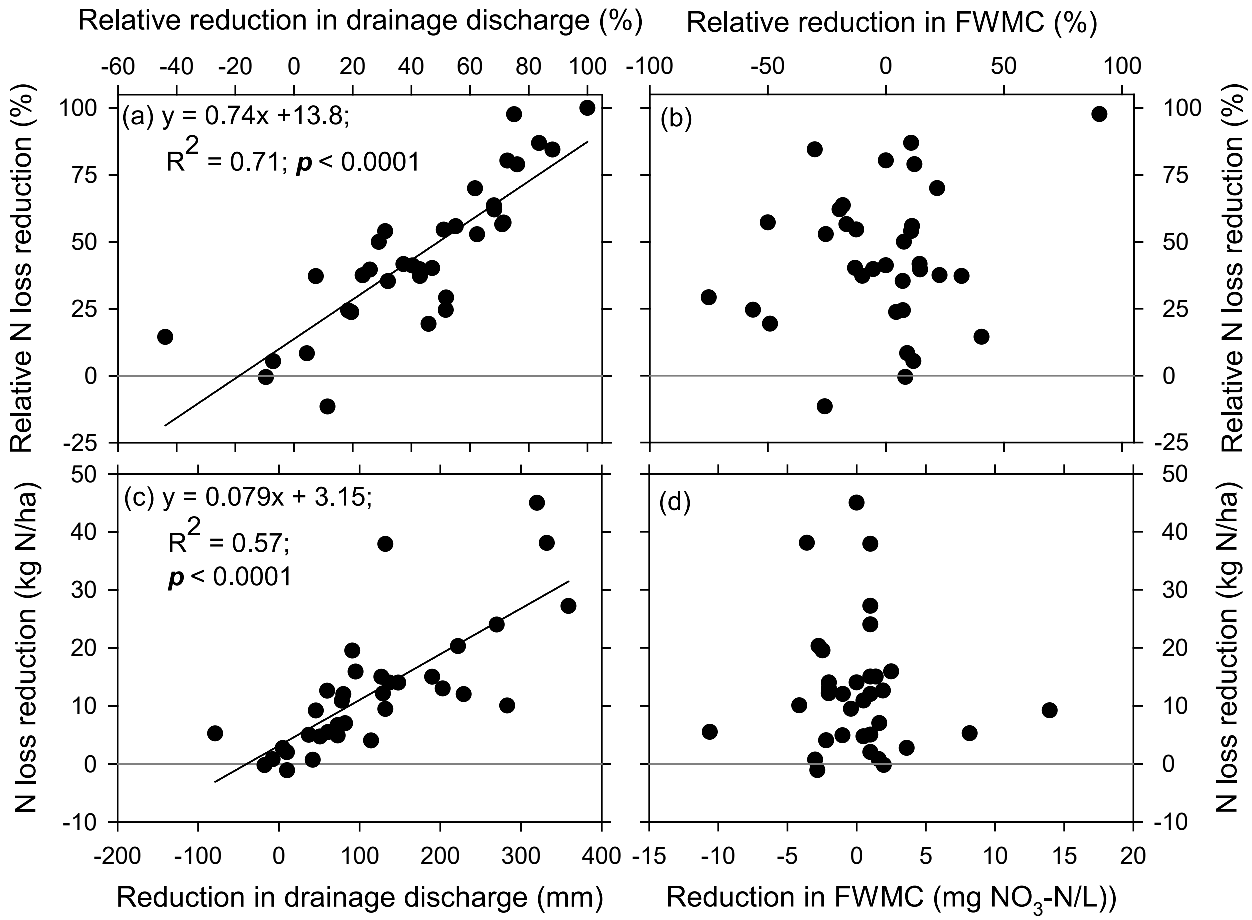

Figure 62. Nitrate-N loss reduction (%, top panels; kg N/ha removed, bottom panels) due to the practice of controlled drainage as a function of the reduction in drainage discharge (as %, a; or mm reduced, c) or flow-weighted mean concentration (%, b; mg NO_3-N/L reduced, d). Data sourced from studies listed in Table 37.

Cooke and Verma (2012) further noted that controlled drainage reduced the discharge compared to free drainage in all years of their study. That is, in no year did controlled drainage increase discharge or N loss. This consistency was reflected in the favorable coefficient of variation for N loss reduction provided by controlled drainage (0.60: 27/45), much like the other conservation drainage practices that each had coefficients of variation less than 1.0 (Figure 13).

The reduction in drainage discharge provided by controlled drainage, however, can vary across the Midwest (Thorp et al., 2008). Helmers et al. (2022) showed that more northern and western Midwestern locations had relatively more drainage in the spring versus winter (e.g., Minnesota has less winter drainage than Indiana). The intent of this practice is to reduce drainage flow over the winter and early spring, with the outlet generally managed in "*free drainage*" mode in the weeks prior to planting. In areas with little or no winter drainage (e.g., Minnesota), it would be desirable to control the outlet as long as possible in the spring to maximize the water quality benefit, but this would need to be weighed against agronomic conditions (Helmers et al., 2022).

The impact of controlled drainage on phosphorus loss was assessed using eleven studies (Table 38). Based on the data compiled here, controlled drainage provided a 30 ± 29% reduction in tile drainage total P loss and 30 ± 40% reduction in dissolved P loss (medians: 25 and 32%, respectively; Figure 63). These averaged 145 ± 237 g total P removed/ha and 62 ± 117 g dissolved P removed/ha.

Table 38. Controlled drainage dissolved and total P loss studies used in this review for tile drainage impacts. DRP, dissolved reactive phosphorus; SRP, soluble reactive phosphorus; orthoP, orthophosphate; PO_4, phosphate; BACI, before-after, control-impact study design.

Study	# of loss site-years		Location	Method notes
	DP (form)	**TP**		
Saadat et al. (2018)	5 (SRP)	5	Indiana	Two plots each for free and controlled drainage were averaged; one outlier year removed
Feser et al. (2010)	2 (orthoP)	2	Minnesota	Field-scale, east versus west management zones
Wesström and Messing (2007)	4 (orthoP conc. only)	4	Sweden	Oct-June monitoring; Two controlled drainage plots were averaged
Carstensen et al. (2019)	3 (PO_4)	3	Denmark	Two plots each for free and controlled drainage were averaged
Evans et al. (1989)	2 (PO_4)	2	North Carolina	Data extracted for the Indiana Science Assessment; two sites
Nash et al. (2015)	4 (orthoP)	0	Missouri	None
Williams et al. (2015a)	4 (DRP)	0	Ohio	BACI design, used “predicted” free drainage
Sahani (2017)	1 (DP)	0	South Dakota	Data extracted for the Indiana Science Assessment
Tan and Zhang (2011) *	4 (DRP)	4	Ontario	Controlled drainage + subirrigation; Used for the above- versus below-ground tradeoff evaluation
Zhang et al. (2017) *	8 (DRP)	8	Ontario	Controlled drainage + subirrigation, but only irrigated in 2 of 4 years; Used for the above- versus below-ground tradeoff evaluation
King et al. (2022) *	0 (DRP)	0	Ohio	4 annual BACI contrasts used for the above- versus below-ground tradeoff evaluation; Event mean loads from two paired fields

* Additionally used in the above- versus below-ground tradeoff evaluation (Figure 64)

These recommended efficiencies aligned relatively well with the 36 and 34% reported by Frankenberger et al. (2023b) for total and dissolved P, respectively, despite the earlier study’s smaller sample size (n = 3 and 7, respectively). To improve data robustness in the current work, several studies were included that were not used in the Indiana review (e.g., Feser et al. (2010) as well as Tan and Zhang (2011) and Zhang et al. (2017) who used subirrigation). Regardless, the total P loss reduction recommended here of 30% and the value of 36% from Frankenberger et al. (2023b) were both lower than the 43% total P reduction currently used in the MPCA Watershed Pollutant Load Reduction Calculator (MPCA, 2022).

The reduction in drainage nutrient losses due to the practice of controlled drainage is due to the reduction in drainage discharge. This has been established for both N (as discussed above, Figure 62) as well as P. However, P dynamics in terms of concentration may differ from N when controlled drainage is performed. In the data compiled here, drainage total P concentrations sometimes slightly increased when the practice of controlled drainage was performed (mean: 18 ±44% increase, median 3% increase; Figure 63c). This difference due to the practice for either total or dissolved P concentrations was not significant.

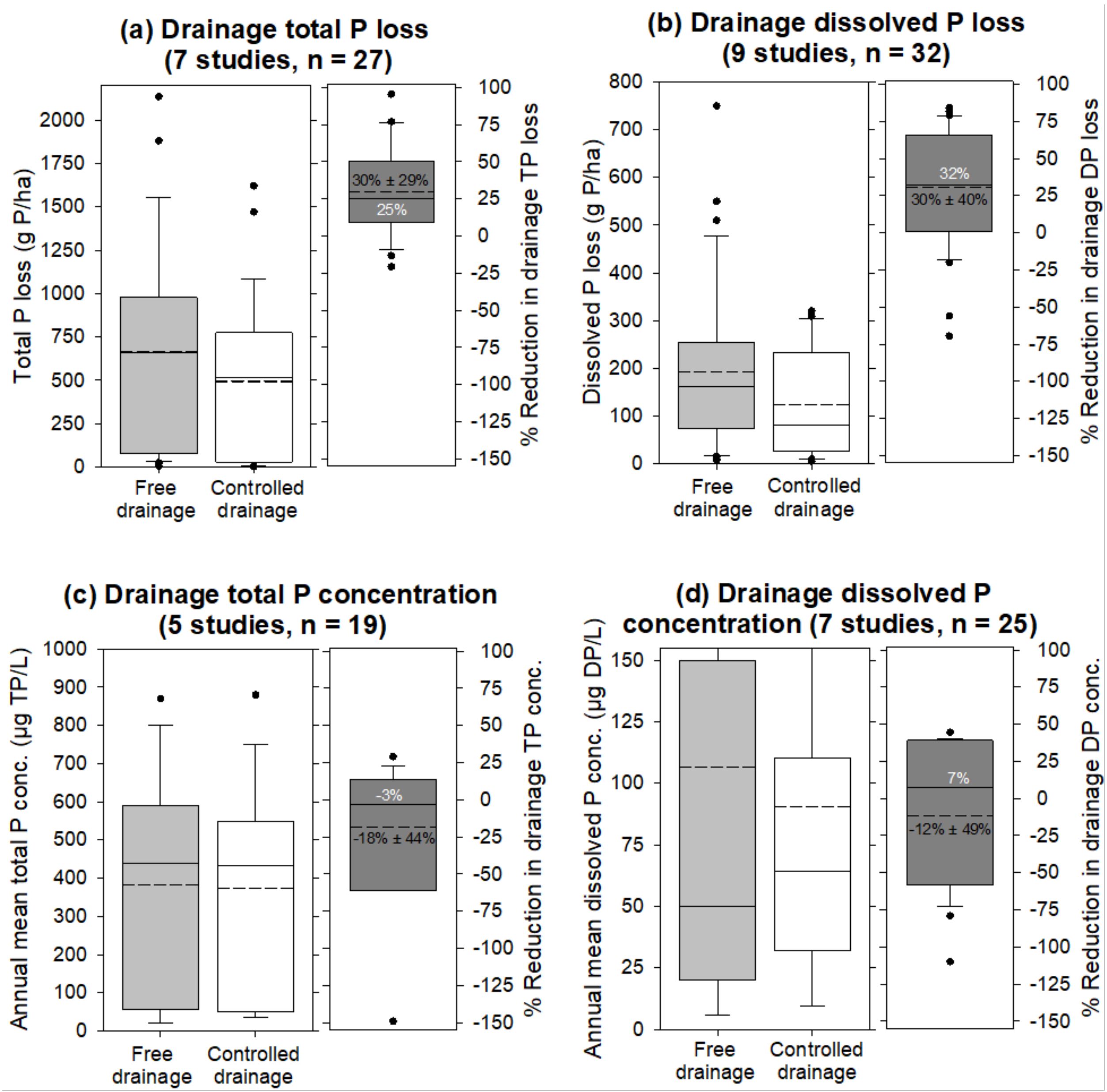

Figure 63. Total and dissolved phosphorus loss and loss reduction (a and b) and the impact on annual concentrations (c and d) provided by the practice of controlled drainage. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 38.

The relative impact that controlled drainage can have on runoff P losses also merits discussion because this practice can increase surface runoff compared to free drainage (Ross et al., 2016; Singh et al., 2007; Thorp et al., 2008; Zhang et al., 2017). King et al. (2022) emphasized that reductions in tile drainage P losses provided by controlled drainage may be negated, at least in part, by increases in P surface runoff losses.

Three controlled drainage studies reported both surface runoff and subsurface drainage total and dissolved P losses (Table 38, n = 16; King et al., 2022; Tan and Zhang, 2011; Zhang et al., 2017). Despite this small sample size (and despite that two of the three studies also included subirrigation), extracting and compiling these data helped illustrate P loss pathways in context. Controlled drainage provided a net total P benefit when considering both tile drainage and surface runoff losses. This practice provided a mean reduction of 242 ± 295 g TP/ha in drainage but increased the surface runoff total P loss by a mean of 106 ± 219 g TP/ha (dashed lines in dark gray bars in Figure 64). Summing those illustrates the benefit to total P, on average, this practice provided (242 - 106 = 136 g TP/ha removed). A similar analysis showed a similar (albeit smaller) impact for dissolved P losses. The mean tile drainage and runoff impacts were 60 ± 96 and -20 ± 80 g DP/ha, respectively, and summing resulted in a net benefit of 40 g DP/ha due to the practice.

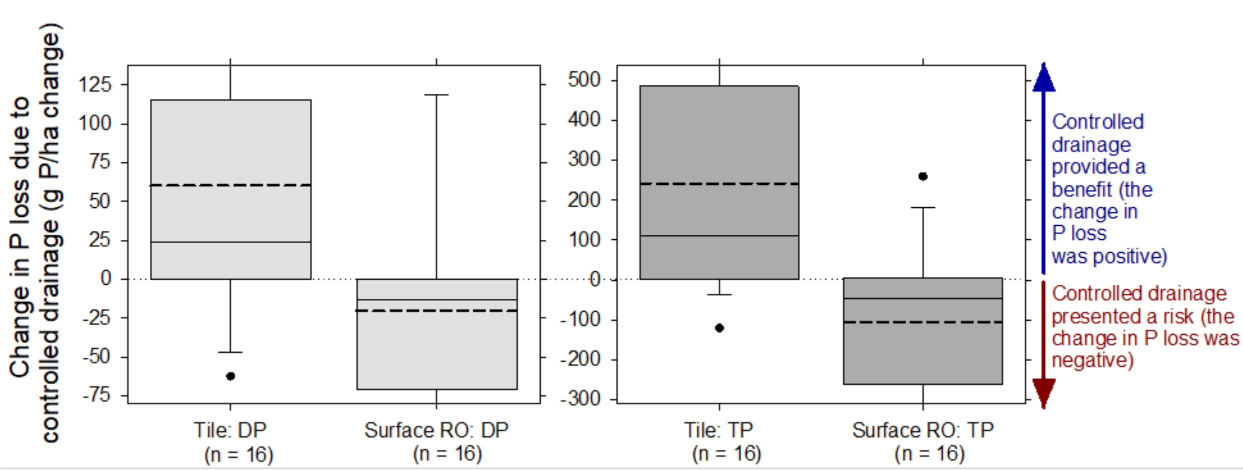

Figure 64. Dissolved P (DP, left) and total P (TP, right) loss impacts due to the use of controlled drainage (versus free drainage) for tile drainage and surface runoff losses of. A positive value indicated controlled drainage reduced the loss and a negative value indicated the controlled drainage increased the loss. The mean is the dashed line; median is the solid line; boxes edges are the interquartile range; dots are the outliers. Data were from (King et al., 2022; Tan and Zhang, 2011; Zhang et al., 2017).

This quantitative review was built from the Indiana Science Assessment database which did not include crop yield; thus, the corn yield impact of controlled drainage was not assessed here. However, the recent review by Youssef et al. (2023) provides a comprehensive evaluation of crop yield across the Transforming Drainage dataset. Overall, there was no significant crop yield benefit provided by controlled drainage relative to free drainage (means: 10.62 and 10.53 Mg ha, respectively; Youssef et al., 2023). Within this generality, there were crop yield impacts based on differences in annual precipitation. For example, controlled drainage boosted corn yield during dry years (4-14% grain yield increase) but created a yield detriment in wet years (4-10% penalty; Youssef et al., 2023).

Controlled drainage has been estimated to have an area-based cost of $50 to $100/ha for retrofitting and generally $220 to $255/ha for a new system (Cooke et al., 2008; Deutschman and Koep, 2022; Frankenberger et al., 2023b). This practice has a cost efficiency of approximately $1 to $11/kg N reduced, assuming no yield benefit (Frankenberger et al., 2023b). However, there is a positive return to the investment if a yield benefit is achieved, and the importance of even the possibility of a yield benefit to a land owner should not be underestimated. It is also interesting to consider that using drainage structures to inundate low areas over the winter could attract waterfowl which may be a benefit for some landowners (e.g., for hunting, Mitchell et al., 2023).

One of the biggest limitations for implementation of controlled drainage is that it is most cost effective on flatter slopes (e.g., up to 2%, generally less than 1%; Cooke et al., 2008). Structural drainage practices that require permanent infrastructure may also face barriers due to nuances of land rental (i.e., lack of landowner buy-in; Dentzman et al., 2022). Lastly, the necessary adjustment of the structures several times a year is a potential barrier to adoption (i.e., required labor). Simply, "*leaving structures open*" all year does not provide the nutrient loss reduction intended by this practice. However, relatively new automated control structures (e.g., Smart Drainage System®, AgriDrain) usher in a new age of sophistication for more real-time drainage water management (e.g., Kumar et al., 2023; Miller et al., 2022). Use of automated control structures has been approved for payment within the NRCS Environmental Quality Incentives Program in Minnesota (CPS 587; USDA NRCS, 2024).

The two largest knowledge gaps in understanding the water quality impacts of the practice of controlled drainage deal with the overall water balance (e.g., runoff and seepage) and P dynamics (King et al., 2022; Ross et al., 2016). Reported nutrient loss reductions could be overestimated due to a variety of factors including: flow monitoring limitations (i.e., discharge measurement during submerged conditions); lack of surface runoff monitoring in tandem with drainage monitoring; and uncertainties about seepage (Cooke and Verma, 2012; David et al., 2014; Frankenberger et al., 2023b). Chandrasoma et al. (2022b) attempted to quantify seepage at free and controlled drainage plots in Illinois. They found that considering seepage decreased the N reduction benefits slightly from 41-93% N loss reduction (without considering seepage) to 36-73% (considering seepage).

The possibility for water balance tradeoffs with above- and below-ground pathways could be compounded by the relatively large fraction of annual runoff that occurs as snowmelt in Minnesota. Additional field data are needed to better understand these conditions (Luo et al., 2010). Moreover, while many controlled drainage studies have focused on N loss reduction, higher-frequency sampling would be helpful to better estimate this practice's impact on P losses (Frankenberger et al., 2023b; Williams et al., 2015b). In summary, while these limitations present tangible opportunities to improve the robustness of controlled drainage studies, the nutrient loss reductions this practice provides are likely still substantial where landscape conditions allow controlled drainage to be performed.

Saturated buffers

The intersection of the 3.3 million tile-drained hectares in Minnesota and the 98% compliance achieved for Minnesota's Buffer Law (BWSR, 2019; State of Minnesota, 2023) results in a significant opportunity for the practice of saturated buffers in the state. In tile-drained areas, a saturated buffer reconnects the hydrology at the edge of the field by distributing drainage water underground in a non-cropped buffer where nitrate can be treated via natural processes (Jaynes and Isenhart, 2014; Johnson et al., 2023; MN NRCS, 2021b). Denitrification is generally considered the dominant N removal process (Groh et al., 2019), but the vegetative component also matters because established buffers appear to provide more N removal than recent buffer retrofits (Jaynes and Isenhart, 2019b). The term "*saturated buffer*" is used here, although this practice has also been called: "*saturated riparian buffer*" (Jaynes and Isenhart, 2019b), "*saturated buffer zone*"; and "*vegetated subsurface drain outlet*" (NRCS Interim CPS 739). An "*integrated buffer zone*" as studied in Europe is distinct from the practice assessed here (Carstensen et al., 2020; Stutter et al., 2022; Zak et al., 2019) and is not included.

Saturated buffers are a relatively new practice, and although there have been few studies of this practice to date (e.g., Carstensen et al., 2020), several studies provided a number of site-years (Table 39). Much of the work assessing saturated buffers has been done in Iowa (e.g., Jaynes and Isenhart, 2014; 2019b) but data from other locations is starting to become available (Illinois: Chandrasoma et al., 2022a; Eldridge et al., 2024; Ohio: Jacquemin et al., 2024; Jacquemin et al., 2020). Beyond N removal, previous saturated buffer studies have also provided context on groundwater hydraulics (Akrofi, 2024; Sahad, 2022; Streeter and Schilling, 2021), design processes (Abdalaal and Ghane, 2023; Dickey et al., 2021; McEachran et al., 2020; McEachran et al., 2023), and mechanistic functioning (Davis et al., 2019; Groh et al., 2019).

Table 39. Saturated buffer nitrate removal studies in this review

Study	# of site-years	# of sites	Location	Method notes
Jaynes and Isenhart (2019b)	19	6	Iowa	Also see Jaynes and Isenhart (2014)
Indiana Science Assessment	3	1	Iowa	Transforming Drainage dataset, Chighladze et al. (2021)
Streeter and Schilling (2021) *	4	2	Iowa	Flow was not measured; N reduction efficiency (%) was not reported nor calculable
Schilling and Streeter (2022) *	1	1	Iowa	Flow was not measured; drought; N reduction efficiency (%) was not reported nor calculable
Chandrasoma et al. (2022a)	10	3	Illinois	None
Eldridge et al. (2024)	4	2	Illinois	Standard and pitchfork designs; Extracted data "*without days containing backflow*"
Utt et al. (2015) *	22	12	Iowa, Illinois, Indiana	Original sites not designed according to the CPS 604
Utt et al. (2015)	5	3	Minnesota	Original sites not designed according to the CPS 604; included to ensure MN data
Indiana Science Assessment	1	1	Minnesota	Transforming Drainage dataset, Chighladze et al. (2021)
Jacquemin et al. (2020) *	1	1	Ohio	N reduction efficiency (%) was not reported nor calculable

* not used to develop the recommended average N loss reduction efficiency (%)

Two studies provided the majority of saturated buffer sites-years for this review (n = 29; Table 39; Chandrasoma et al., 2022a; Jaynes and Isenhart, 2019b). In addition to those, several sites compiled through the Transforming Drainage project (Chighladze et al., 2021) were used along with several newer studies (Eldridge et al., 2024; Jacquemin et al., 2020; Schilling and Streeter, 2022; Streeter and Schilling, 2021). Flow monitoring differed across some of these studies, but they were included nevertheless because the methods did not change the results. The full 27 site-years reported by Utt et al. (2015) were used in the Indiana Science Assessment, but a number of these early site-years were excluded here (following Johnson et al., 2023) because these sites were designed prior to the existence of a Conservation Practice Standard and thus deviated from the standard. However, to make this review as relevant as possible for the state of Minnesota, the original Minnesota sites from that early study were included.

The edge of field N loss reduction efficiency for saturated buffers was 43 ± 26% for this Minnesota-focused review (Figure 65c, left/green bar; median: 35%). The average annual N loss reduction was 9.0 ± 6.8 kg N/ha-y (median: 8.3 kg N/ha-y removed; n = 43; Figure 65a) and the average N load reduction was 98 ± 136 kg N/y (median: 62 kg N removed/y; n = 44; Figure 65b).

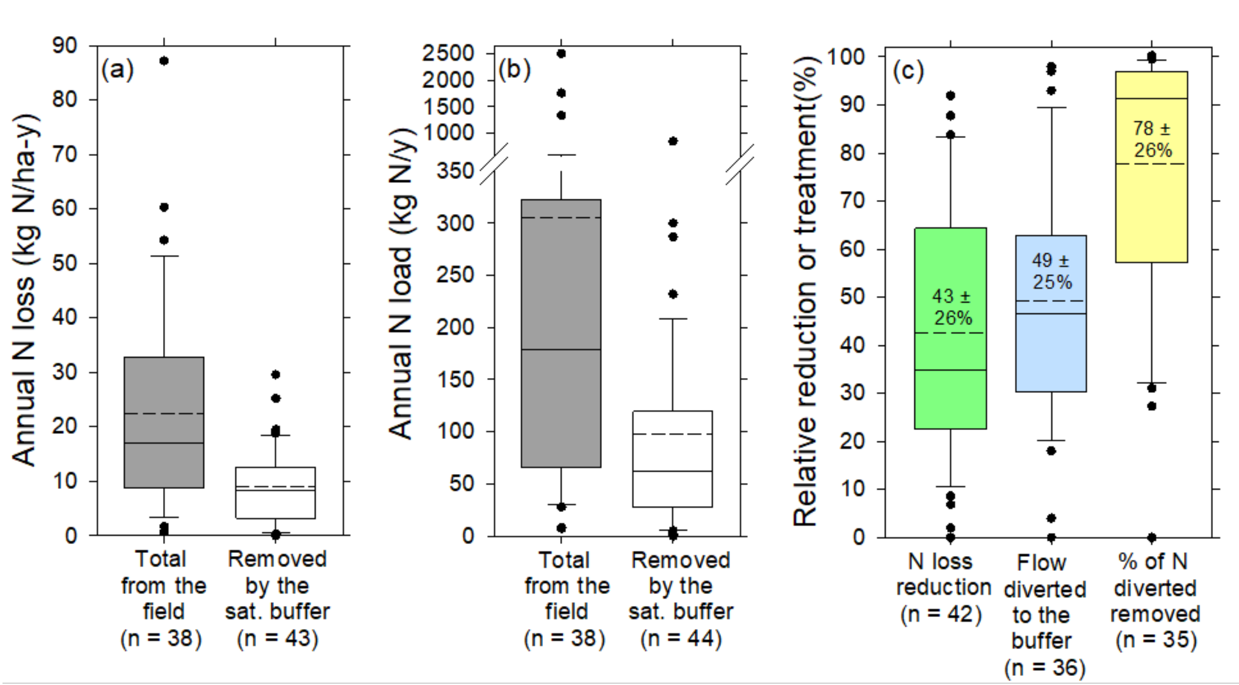

Figure 65. Annual N loss (a) and N load (b) from the field and removed in the saturated buffer as well as the fraction of water treated, N removed from that treated water and overall N loss reduction efficiency considering the total loss from the field (c). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 39.

The recommended mean of 43% annual N removal is similar to the recent review by Johnson et al. (2023) who reported a mean of 46 ± 24%. Saturated buffers were included in the Iowa Nutrient Reduction Strategy as part of their 2017 update at a value of 50 ± 13% N reduction (IDALS, 2017), which was recently revised to 45 ± 24% (IDALS, 2024). The MPCA Watershed Pollutant Load Reduction Calculator uses a default value of 45% N reduction for tile drainage water (MPCA, 2022).

The overall recommendation here is slightly lower than these earlier values to more conservatively account for the lack of Minnesota data for this practice.

Generally, half of the drainage water from the field is routed into a given saturated buffer for treatment (49 ± 25%; Figure 65c middle/blue bar). The vast majority of nitrate in that treated water is removed based on sampling of monitoring wells in the buffer (i.e., 78 ± 26%; Figure 65c right/yellow bar). Simply put, saturated buffers are very effective at removing nitrate that is routed into them. Thus, the overall mass load reduction provided by this practice is limited by the amount of flow treated (Chandrasoma et al., 2022a; Johnson et al., 2023). This amount of flow (i.e., hydraulic loading) is a function of the drainage area (Figure 66a) and distribution pipe length (Figure 66b) in addition to soil and hydraulic properties (e.g., hydraulic gradient across the buffer; Johnson et al., 2023; McEachran et al., 2020). The annual N load reduction provided by saturated buffers correlates with the product of the drainage area and distribution pipe length (Jaynes and Isenhart, 2019b). This correlation in the current dataset is not as strong as that shown in the original work (R^2: 0.228 versus 0.479, respectively), but is still significant ($p < 0.001$; data not shown).

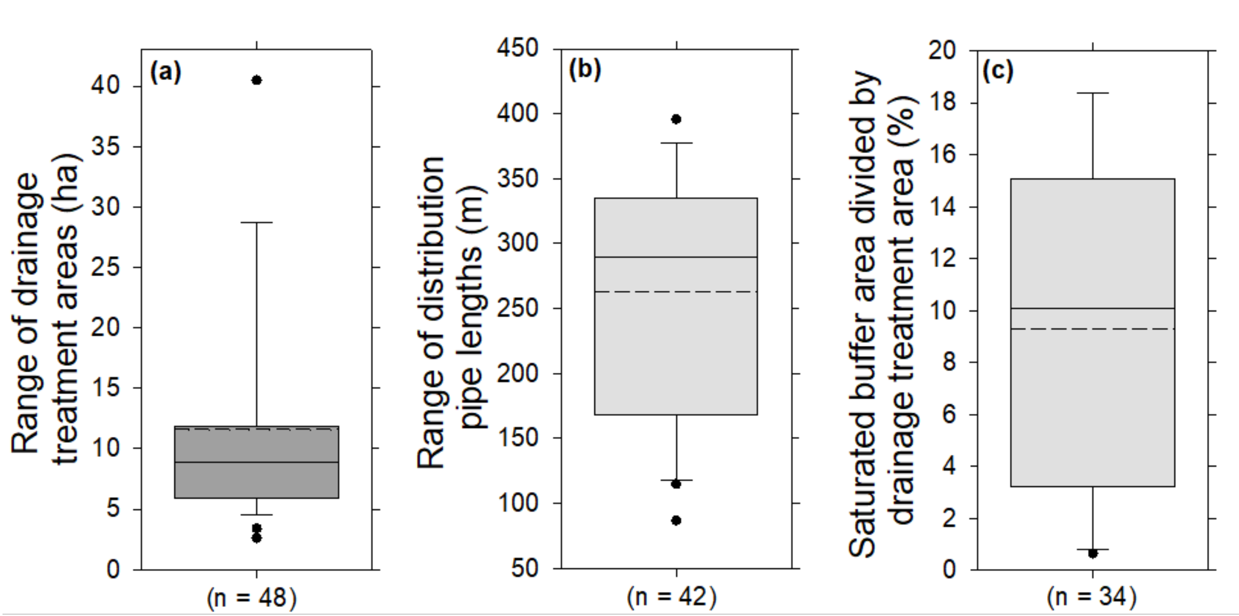

Figure 66. Design description (drainage treatment area, a; distribution pipe length, b; treatment area ratio, c) for saturated buffers assessed in Figure 65. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers.

Data on P removal at saturated buffers was insufficient to develop a recommended value in this science assessment (e.g., Carstensen et al., 2020). Chandrasoma et al. (2022a) did not observe any impact on leachate dissolved reactive P concentrations across saturated buffers in Illinois, but Jacquemin et al. (2020) reported an 80% reduction in orthophosphate concentrations at a saturated buffer in Ohio. In terms of surface runoff, a saturated buffer would likely provide similar benefits to an unsaturated buffer (Johnson et al., 2023). Land conversion to perennials (that is, perennials that might be used in a buffer) resulted in a 46% reduction in surface runoff total P in this current review but tended to increase P leaching losses (Table 4). It would be useful to assess surface runoff and possible P tradeoffs at saturated buffers in future studies given the soil may be relatively more saturated than an unsaturated buffer and thus infiltration benefits may differ. Better understanding of the possibility of tradeoffs is especially important in locations where annual runoff is dominated by snowmelt effects (Kieta et al., 2018).

Saturated buffers are thought to be widely applicable across the US Midwest (Chandrasoma et al., 2019; Tomer et al., 2020). According to the Conservation Practice Standard 604, suitable sites must have: (1) soils with high organic matter content (>1.2% in the top 76 cm); (2) no preferential flow paths due to high conductivity soils or pipes; and (3) a deep restrictive soil layer (MN NRCS, 2021b). Using criteria from the practice standard, Tomer et al. (2020) estimated roughly 30-70% of stream banks in Iowa were suitable to host a saturated buffer. Chandrasoma et al. (2019) reported 3.8 million ha of Midwestern drained land could be routed to a saturated buffer.

The potential application of this practice in the state of Minnesota may differ from other states due to Minnesota's Buffer Law (State of Minnesota, 2023). With a reported 98% compliance, this law has successfully established 15 m wide buffers along public waterways and 5 m wide buffers along drainage ditches (BWSR, 2019). It follows that tile main outlets from seemingly every tile drained field in the state should route through an edge of field buffer area. The minimum width for a saturated buffer is 9.1 m (MN NRCS, 2021b), although narrower widths may be effective at some sites (McEachran et al., 2020). Admittedly, the soils in a given buffer area in Minnesota may not be appropriate to host a saturated buffer, but at the very least, the barrier of land conversion required for a saturated buffer has greatly been reduced.

It is worth considering that the practice of a saturated buffer provides two distinct benefits within the context of this water quality review: (1) land use conversion and (2) treatment of drainage water from the cropped area. Retrofitting cropped land to create a new perennial buffer was assessed at a 94% N leaching reduction in this review (e.g., a CRP buffer, Table 4). On average, the saturated buffer area is roughly 10% of the drainage treatment area (Figure 66c). Thus, the land conversion component of a saturated buffer retrofit from cropped land would provide a 9.4% N leaching reduction (94% N reduction X 10% of the area). The remainder of the area would benefit from the 43% N loss reduction provided by saturated buffer treatment (43% N reduction X 90% of remaining area = 39% reduction from the total area). This equates to a 48% reduction across the entire area (94% N reduction X 10% of the area + 43% N reduction X 90% of the area). Because the two practices are applied to different land areas (buffer area versus cropped area), the area-weighted efficiencies are additive. Thus, this is not truly an example of practice "*stacking*." Feyereisen et al. (2022) provided a more nuanced description of these calculations for stacked and combined practices (e.g., see example c in their Figure 2).

Nevertheless, it is most appropriate to recommend the N reduction value of 43% for saturated buffers rather than the overall 48% calculated in the example above. This is because tracking of these two benefits would be done separately. For example, satellite imagery may be used to detect the land use change at the edge of the field whereas the treatment component may be tracked using NRCS data for implementation of Conservation Practice Standard 604. It is also worth noting that the comparison baseline for land uses is 1980-1996 for the purposes of the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force (Hypoxia Task Force; USEPA, 2017). Thus, cropped areas converted to buffers (saturated or not) prior to that date would not get credited with the 94% N leaching reduction because they were in that land use prior to the baseline period. As to how these temporal effects are accounted for by state agencies is beyond the scope of this review, but of course, buffers provide benefits to the environment regardless of if they are "*credited*" in state nutrient strategy processes.

The saturated buffer area averaging 10% of the drainage treatment area is a relatively small edge-of-field conversion compared to a total land use conversion (e.g., Chapter 5), but this does represent a 10% crop yield penalty. This may be a relatively moot point in Minnesota where the existence of the Buffer Law has already required this land conversion. However, crop yield penalties are generally a major consideration for landowners. Case in point, two of the three saturated buffers studied by Chandrasoma et al. (2022a) were re-converted to cropland once the CRP contract expired. The water quality benefit of a saturated buffer that is re-converted to crop land is unknown but would depend on continued management of the control structure.

Despite any limitations, the capital cost of saturated buffers is very reasonable compared to other edge-of-field practices ($1500 to $5000 to treat field-sized areas, e.g., Figure 66a; Jaynes and Isenhart, 2014; Kjaersgaard, 2021). The cost efficiency of saturated buffers is also very competitive ($1.20-$9.2/kg N removed; Johnson et al., 2023). It is possible there may be income benefits from buffers that are harvestable for animal feed or bioenergy (Johnson et al., 2023).

The novel "*Batch and Build*" approach, which is described in the middle of the above "*Conservation drainage background and database overview*" section, has accelerated adoption of saturated buffers and bioreactors in Iowa. This new program has also illustrated the importance of planning for these practice's engineering costs, which have historically been borne by the NRCS (Maxwell et al., 2024). As edge-of-field practices become increasingly necessary to progress toward water quality goals in tile-drained areas (Feyereisen et al., 2022), innovative conservation delivery mechanisms that disrupt traditional barriers (e.g., paperwork burden, lengthy lead times) need to be seriously considered. Approaches like the "*Batch and Build*" can increase efficiency and reduce public expenditure for these practices and thus should be further evaluated in additional areas (Kult, 2022).

Finally, as a relatively new conservation practice, there is still much to learn about the design, operation, and functioning of saturated buffers. **This Minnesota review highlights the need for more field-scale studies documenting N removal performance at saturated buffers in Minnesota that have been designed according to the CPS 604.** Deeper assessment of P dynamics in drainage and surface runoff water (including snowmelt) at saturated buffers would also be useful across many states. Saturated buffer design criteria are still being refined, with for example, the use of backflow-minimizing check valves a useful recent addition to the design toolbox (Eldridge et al., 2024). Assessment of novel saturated buffer designs will continue to move this field forward (Christianson, 2021; Jaynes and Isenhart, 2019a).

Denitrifying bioreactors

Denitrifying bioreactors, termed "*bioreactors*", use a solid organic carbon source to enhance denitrification of nitrate-laden water (MN NRCS, 2021a; Schipper et al., 2010). Bioreactors can be used to treat nitrate in subsurface drainage (as discussed here), shallow groundwater (Manca et al., 2020; Schipper and Vojvodic-Vukovic, 1998; Schmidt and Clark, 2012), and agricultural, municipal, and mining wastewaters (Suozzo et al., 2022; von Ahnen et al., 2018; Wadnerkar et al., 2022). Woodchips are the most common fill (Christianson et al., 2021a) although alternatives continue to be explored (Law et al., 2023). A key limitation of edge-of-field drainage practices is that they cannot address peak flow events, meaning some water will bypass untreated. Thus, results for saturated buffers and bioreactors are presented as "*edge of field*" reductions (EOF%) to represent the impact across the entire drainage treatment area which contributes both treated flow and untreated bypass flow.

In this review, 116 bioreactor studies were evaluated. The assessment of this conservation practice was more developed than for some others to meet objectives of an additional project (Christianson, 2021). Results from the larger database are presented elsewhere (e.g., large-scale bioreactors, design and operation comparison; Christianson, 2025 in preparation). Ultimately, 21 references that reported annual nitrate or P reduction at full-size bioreactors treating subsurface drainage were used (Table 40).

Bioreactors provided 30 ± 21% N loss reduction when assessed at the edge of the field (Figure 67c, left/green bar; median: 23%). The average annual N loss reduction was 5.5 ± 4.8 kg N/ha-y (median: 4.2 kg N/ha-y removed; n = 57; Figure 67a) and the average N load reduction was 79 ± 75 kg N/y (median: 52 kg N removed/y; n = 60; Figure 67b). The average nitrate removal rate for these site-years was 8.3 ± 8.7 g N removed/m^3 bioreactor-d (median: 5.0 g/m^3-d, n = 63; data not shown).

The recommended mean and the noted median (30% and 23%) bracketed the N efficiencies used for bioreactors in Iowa and Illinois of 24 ± 17% and 25%, respectively (IDALS, 2024; IDOA, 2015). The recommended mean in the current work was higher than the 22% Total N reduction used for bioreactors in the MPCA Watershed Pollutant Load Reduction Calculator (MPCA, 2022). It is possible the N loss reduction provided by bioreactors in Minnesota may be more on the order of 22-25%, but barring additional data, the mean ± standard deviation was recommended to be consistent with the reviews for the other practices. Regardless, it is recommended the 13% used for bioreactors in the original Minnesota Nutrient Reduction Strategy be increased (MPCA, 2013).

Using only data from Minnesota and immediately surrounding states resulted in an edge of field efficiency of 43 ± 22% (median 44%; *n* = 21 from MN, IA, and SD, the first six rows in Table 40) which was notably higher than the final recommended value of 30%. This higher reduction efficiency was similar to a global review by Christianson et al. (2021a) who reported an average of 40 ± 26% for bioreactors (median: 46%). However, this Minnesota-specific value of 43% seemed too high to recommend given the values used in the Iowa and Illinois strategies. Thus, it was deemed appropriate to include bioreactor performance data from Illinois, New York, and Ontario to better constrain reasonable expectations for bioreactors designed using current design processes. Despite this necessary data filtering process, it bears to note that the first six rows of Table 40 demonstrate that bioreactors do work in Minnesota's northerly climate (e.g., 43% N reduction).

Water temperature importantly influences denitrification in bioreactors with many small-scale experiments showing less nitrate removal under cooler conditions (Cameron and Schipper, 2010; Feyereisen et al., 2016; Hoover et al., 2016). How these results translate to impacts on annual N removal performance in the field needs to be better assessed at full-size bioreactors over the long term. Using latitude as a proxy for climate here illustrated that bioreactors perform better in warmer climates (Figure 68; North American studies only). This unsurprising correlation was not strong but was significant (R^2: 0.149, p < 0.0001).

Table 40. Denitrifying bioreactor field-scale nitrate and phosphorus removal site-years for both the drainage water treated ("Bio%") and edge of field effectiveness which considers untreated bypass water ("EOF%"). The Phosphorus removal column refers to treated water only and notes the form that was reported.

Study	# of nitrate site-years		# of P site-years	# of sites	Location	Method notes
	Bio%	EOF%				
Benes et al. (2016)	2	0	4 (TP conc.)	2	Minnesota	North and South sites; Additional site-years were monitored but sampling was sparse; Used TP concentrations as loss proxies
Ranaivoson et al. (2012)	2	2	---	1	Minnesota	Summed reported snowmelt and growing season results
S. Matteson (per. comm.)	2	2	2 (PO_4)	2	Minnesota	North and South sites
Nelson, (2013) *	1 (conc.)	0	---	1	Minnesota	Nitrate concentrations only, no load reported
Christianson et al. (2012)	7	7	---	3	Iowa	Four sites studied, but one was considered pilot-scale due to size
Thapa et al. (2023)	0	10	---	4	South Dakota	---
Woli et al. (2010)	0	2	---	1	Illinois	---
David et al. (2016)	3	0	3 (DRP)	1	Illinois	---
Christianson et al. (2024 in preparation)	34	34	22 (DRP)	10	Illinois	P data was in Sanchez-Bustamante Bailon et al. (2024)
R. Cooke (per. comm.)	9	0	---	6	Illinois	---
Maxwell et al. (2022) *	3	0	---	1	Illinois	Subsurface drainage diverted from a ditch
Israel et al. (2023)	3	0	---	1	New York	Loads estimated using modeling
Hassanpour et al. (2017)	7	0	---	3	New York	One multi-year mean reported for each woodchip-only bioreactor
Van Driel et al. (2006) *	3 (conc.)	0	---	2	Ontario	Nitrate concentrations only, no load reported
Robertson et al. (2000)	6	0	---	1	Ontario	One multi-year mean reported
de Haan et al. (2010) *	1	1	---	1	Netherlands	Drainage from vegetables
Gosch et al. (2020) *	3	0	---	1	Germany	---
Plauborg et al. (2023) *	6	0	6 (TP)	3	Denmark	Only used data from bioreactors treating ≤ 80 ha (field-scale)
Audet et al. (2021) *	6	0	---	2	Denmark	Only used data from full-sized bioreactors (not replicated pilots)
Rivas et al. (2020) *	2	0	2 (DRP conc.)	1	New Zealand	Used for P analysis only; used concentrations as loss proxies
Bock et al. (2018) *	0	1	1 (TP conc.)	1	Virginia	Used for P analysis only; used concentration as a loss proxy

* Not used to develop the recommended average N loss reduction efficiency (%) for Minnesota but some of these were used in the Northern European or phosphorus assessments, as appropriate (Figure 69, Figure 70)

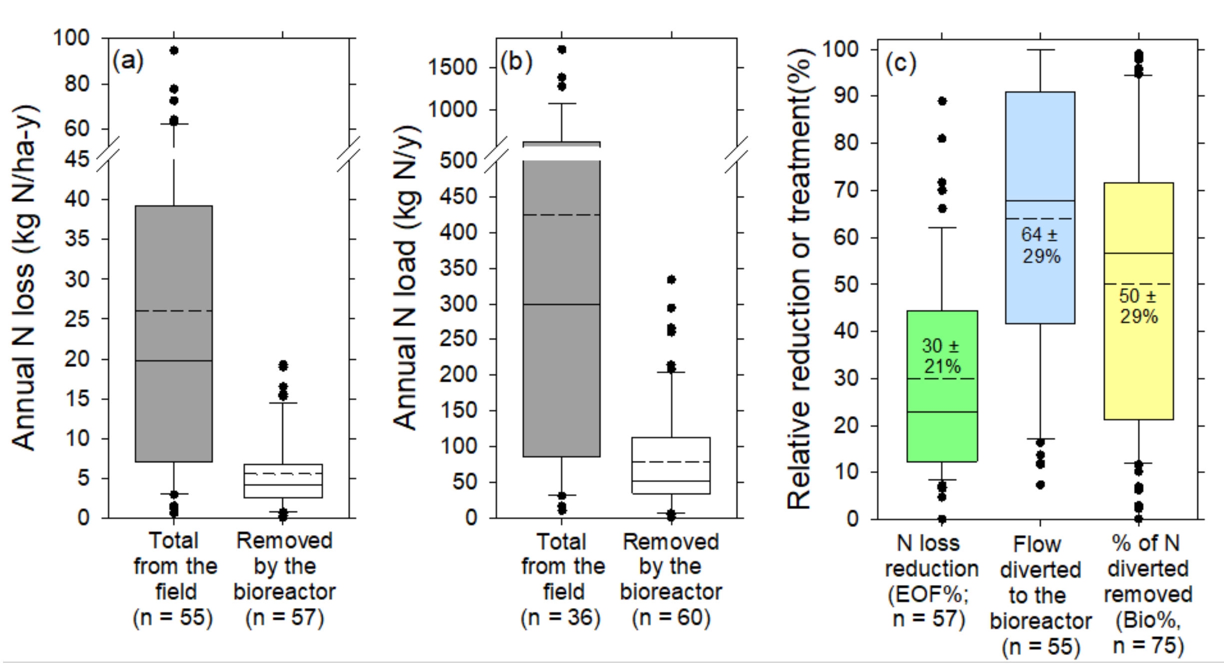

Figure 67. Annual N loss (a) and N load (b) from the field and removed in bioreactors as well as the fraction of water treated, N removed from that treated water ("Bio%") and overall N loss reduction efficiency considering the total loss from the field ("EOF%") (c). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 40.

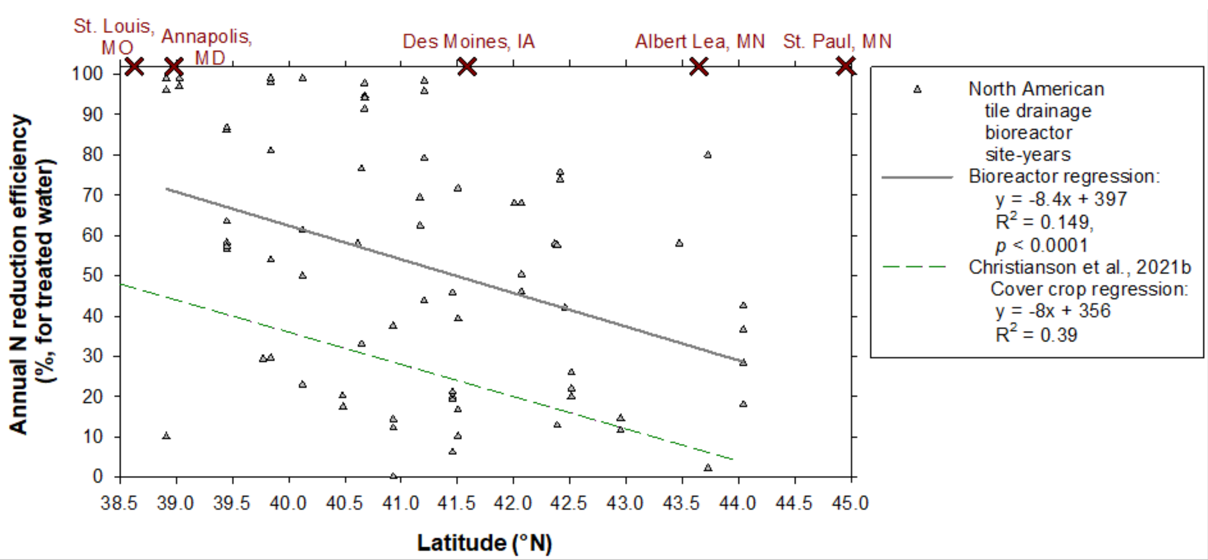

Figure 68. Correlation between the latitude of North American bioreactor locations and annual N removal efficiency (for treated water, "Bio%") with a similar previously reported correlation for cover crops. Latitudes for several reference cities are listed along the top. Data were sourced from studies listed in Table 40.

A benefit of this comprehensive science assessment was that it allowed practices to be viewed in context of each other. Christianson et al. (2021b) developed a similar relationship to assess the impact of latitude on N removal performance of cover crops (see discussion in "*Cover crops in short season cropping systems*"). The regression slopes for bioreactors and cover crops were surprisingly similar with both resulting in approximately 8% lower effectiveness for every additional degree north in latitude (Figure 68). Taken all together, these data demonstrate many conservation practices perform differently in cool climates and all the practices in Minnesota's conservation toolbox will be necessary to advance toward water quality goals.

Assessment of bioreactor performance in northern Europe provided additional climate context for this Minnesota-focused review (Table 40: Netherlands, Germany, Denmark). On a mass basis, these European bioreactors removed more N annually than the North American sites (Figure 69d; means: 149 vs 79 kg N/y). The design approach for bioreactors may differ in Europe, however. European bioreactors in this assessment tended to be placed on larger drainage areas (Figure 69a) and correspondingly tended to be larger than their North American counterparts (Figure 69b). Nevertheless, all bioreactors had a relatively small footprint when expressed as a percentage of the drainage area (Figure 69c; e.g., generally <0.15%). The latitudes of these European sites ranged from approximately 51°N to 56°N, with the farthest north bioreactors identified in this review located at above 67°N to treat nitrate at iron mines in Sweden (Herbert et al., 2014; Nordström and Herbert, 2018).

Overall, these European data illustrated an important concept: bioreactors are a viable N-removal technology even in cold climates. Professor Richard Cooke from the University of Illinois summarized this well: "*Bioreactors work as long as the water is in the liquid form*" (personal communication). Design processes may need to be further refined for specific climates, but this is an achievable research and engineering task.

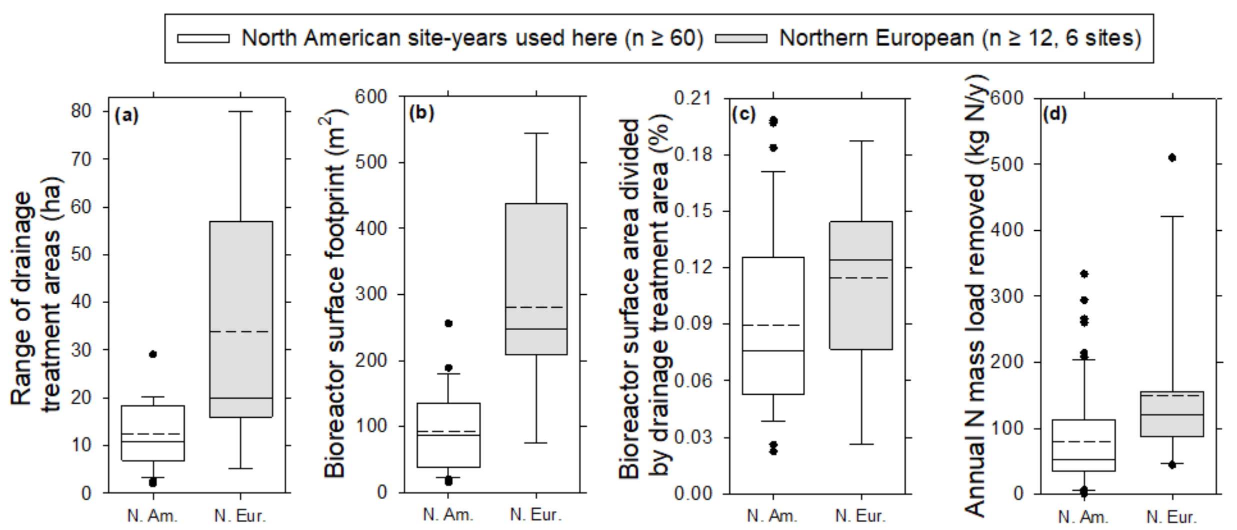

Figure 69. Design description (drainage treatment area, a; bioreactor surface footprint, b; treatment area ratio, c) along with annual N load removal (d) for bioreactors from North America and northern Europe (from Table 40). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers.

Denitrifying woodchip bioreactors are specifically intended for nitrate removal and thus should not be expected to provide P removal. However, P can leach from woodchips so assessing this possible tradeoff is merited. Few studies reported bioreactor P dynamics, and those that did often reported a comparison of bioreactor inflow versus outflow (i.e., Bio% for P rather than EOF%). These limitations were addressed using a multi-step approach. Firstly, several annual P concentration reduction efficiencies were used as proxies for P loss reductions. This was justifiable because these lined bioreactors could be assumed to have a relatively closed water balance, thus changes in annual mean concentration across a bioreactor should be very similar to changes in annual load or loss across that bioreactor. Dissolved P concentration reductions reported by Rivas et al. 92020) at a bioreactor treating pasture drainage in New Zealand and total P concentration reductions reported by Bock et al. (2018) at a bioreactor in Virgina and by Benes et al. (2016) at two bioreactors in Minnesota were handled in this way (Table 40).

A second necessary assumption was needed to extrapolate the P removal efficiencies reported for just the treated water to the entire discharge from the field. To do this, the average percentage of flow treated of 64% from the N reduction analysis (Figure 67c middle/blue bar) was applied to the relative changes in P loss across the bioreactor. An additional nuance was that this was one of the practices where the mean (negative) and median (positive) impacts on P loss markedly differed (Figure 70a). The mean dissolved and total P efficiencies were -104 and -195% whereas the medians showed a benefit of 2 and 35% reduction, respectively. Thus, the 64% average of flow treated was applied to the median value rather than the mean. Assuming a conservation of P mass in the water that bypassed the bioreactor resulted in a dissolved P reduction of 1% (0.02 X 0.64). Applying the same fraction of flow treatment to the mean total P reduction resulted in an edge of field reduction of 22% for total P (0.35 X 0.64). Note, there were few total P site-years (n = 11; Figure 70) so the extrapolated value for total P was not advanced to the final set of recommendations (Table 36).

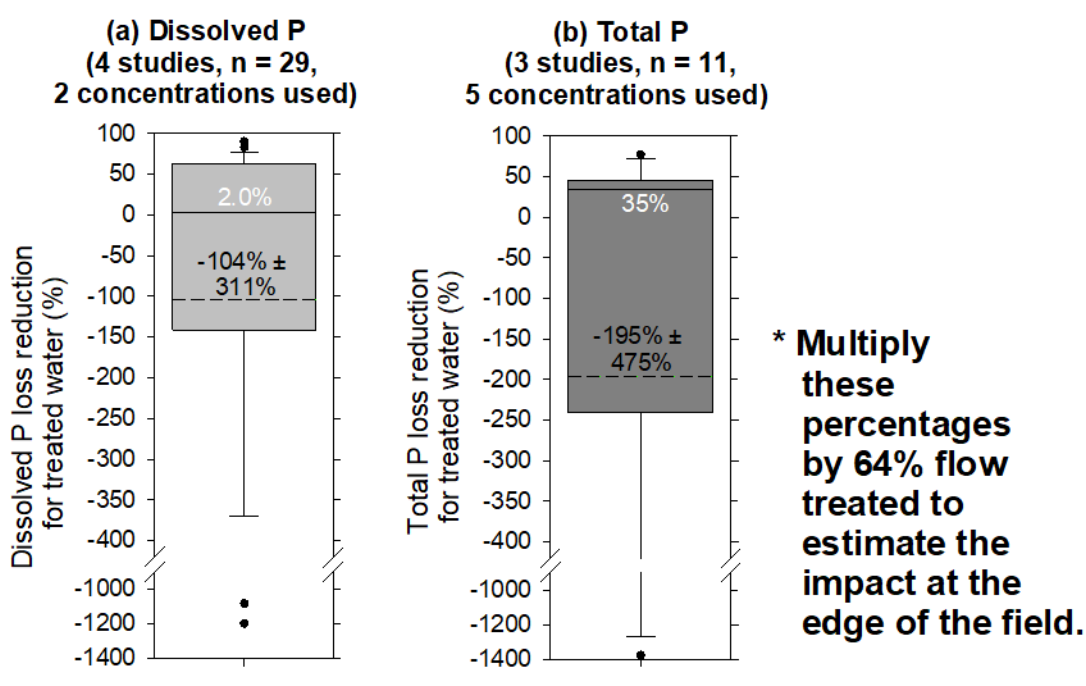

Figure 70. Dissolved (a) and total (b) phosphorus reductions (%) for water treated in a bioreactor. The reduction applies to only the treated water (that is, Bio%, not EOF% for P). Two and five annual concentrations were used as proxies for dissolved and total P, respectively. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 40.

Sanchez-Bustamante Bailon et al. (2024) recently attempted to address if bioreactors were a net source or sink for P in drainage. They reported changes in dissolved P loads across ten bioreactors in Illinois were small, and while the bioreactors tended to provide a P removal benefit, the risk of P release events negated the possibility of developing a P removal credit for this practice.

Their data significantly informed the analysis here (22 of 29 site-years), so the current findings and recommendations are similar. In the current work, just over half of the dissolved P site-years showed removal rather than release (15 of 29, 52%) and 64% of the total P site-years showed a removal benefit (7 of 11 site-years). Improved understanding of P removal mechanisms and the fate of P in bioreactors (e.g., after 10-15 years) is needed.

Field-scale bioreactors, such as those that would be designed consistently with the Conservation Practice Standard 605, were the focus of this review but scaling this technology to a drainage district- or small watershed-scale is suggested for future analysis. The average drainage treatment area for bioreactors in this analysis was only 12.4 ha (Figure 69a). Research is increasingly demonstrating that maximizing the annual N load removed by these edge of field practices involves maximizing the N load they receive (Christianson et al., 2024 in preparation; Jaynes and Isenhart, 2019b; Maxwell et al., 2024). Thus, these practices should be increasingly trialed on relatively larger drainage areas.

One of the most notable bioreactors in Minnesota (and possibly, in the US) is the large three-bed bioreactor near Blue Earth, Minnesota that treats drainage from 249 ha (Feyereisen et al., 2023a). Feyereisen et al. (2023a) documented that there is a learning curve with scaling bioreactor technology; for example, an automated sediment exclusion system was an important addition to counteract the sediment in surface runoff received from the catchment. Nevertheless, their N load removal of 713 kg N/y was nearly ten times greater than the mean 79 kg N/y removed across the current dataset (Figure 67b). Researchers in Denmark have set a precedent for the use of bioreactors to treat drainage areas exceeding 100 ha (Plauborg et al., 2023). Those large-scale bioreactors were not assessed in this quantitative review (see Christianson, 2025 in preparation), but even the smaller field-scale bioreactors studied in Denmark tend to treat larger drainage areas and have greater annual N load removals than those in North America (Figure 69). Given that Minnesota farmers have indicated that limited capacity at the main is a major challenge for drainage in the state (Carlson, 2014), linking large-scale drainage improvements with large-scale, targeted water quality practices warrants serious consideration.

Finally, this review illustrated that **the biggest research need for bioreactors in Minnesota is to better establish baseline N removal performance over multiple years at full-size bioreactors operating in the field.** A similar recommendation for baseline research was made for several other practices (cereal rye cover crop, saturated buffer, drainage water recycling). Novel approaches to improve bioreactor performance in cold climates, such as dosing with labile carbon and bioaugmentation (Feyereisen et al., 2023b; Moghaddam et al., 2023; Zhang et al., 2024), usefully advance the science for bioreactors. Better understanding of field-scale bioreactor performance is a necessary first step to facilitate comparison with these more sophisticated approaches.

Alternative drainage design (shallow drainage)

The practice of shallow drainage consists of the installation of drainage pipes at a shallower average depth (for example, 30 to 42 inches below the soil surface) compared to the traditional 36 or 48 inch installation depth (Christianson et al., 2016). Shallow drainage maintains a slightly wetter soil profile while still meeting crop drainage needs. This practice is another way to adhere to the Golden Rule of Drainage: "*Drain what is necessary for good crop growth and trafficability and not one drop more*" (attributed to R.W. Skaggs; Lewandowski, 2010). A minimum drainage depth must still be observed (e.g., at least 24 inches) to protect the pipe's structural integrity from deflection or damage by machinery.

A total of 17 drainage design studies were reviewed, and appropriate data focused on shallow drainage were extracted from seven (Table 41). Studies by Kladivko et al. (2004) and Jaynes et al. (2008) were excluded from analysis because they focused on drain spacing alone (not depth) or the unique treatment of deeply placed drains (i.e., 1.8 m deep). Several other studies modelled the impact of shallow drainage and did not provide original monitoring data (Davis et al., 2000; Luo et al., 2010; Singh et al., 2007; Zhao et al., 2000). No P leaching or surface runoff data were extracted for this practice.

A range of drainage depths and spacings were used across the studies in this review (Table 41). Nine site-years provided information about very shallow drainage where pipes were placed < 0.65 m from surface (i.e., 2 feet or closer to the soil surface; Cooke et al., 2002; Gordon et al., 2000; Schwab et al., 1980). These were excluded from the primary analysis because this shallow placement was not consistent with how the practice would be done in Minnesota.

Of the remaining five studies, a reduction in the depth of the drain placement coincided with narrower drain spacing to maintain a consistent drainage intensity (Skaggs and Chescheir, 2003). For example, the mean drain depth and spacing for the "*control*" were 1.2 m and 18.9 m whereas the "*treatment*" averaged 0.80 m depth and 13.4 m spacing (Figure 71). Reducing the spacing increases the cost of a drainage system (Strock et al., 2011) although, at the same spacing, (Gordon et al., 2000) reported shallow drainage was less expensive than deeper placed drain pipes.

Assessed across these 20 site-years, the practice of shallow drainage provided a 41 ± 24% N loss reduction (Figure 72a; median: 32%). One limitation of these data was that while there were five studies used, data were generated at only three locations and 14 of the 20 site-years were from one site (Crawfordsville, Iowa). The study by Maas et al. (2022) merits a note because the N loss reduction of those five site-years ranged from 59-80%, which increased the overall mean by 10%.

The practice of shallow drainage was not included in the original Minnesota Nutrient Reduction Strategy but was assessed at 32 ± 15% N loss reduction in the original Iowa strategy (IDALS, 2014). As noted above, one study significantly influenced the 41% recommended here, and excluding that newer work resulted in a very similar average to that used in Iowa (31 ± 19%, median: 30%, n = 15; excluding Maas et al., 2022).

The reduction in N loss provided by the practice of shallow drainage was driven by the reduction in drainage discharge (Figure 72c: discharge reduction 38 ± 19%; Cooke et al., 2002; Helmers et al., 2012; Sands et al., 2008; Schott et al., 2017). However, the earliest work in this review reported the shallow drainage treatment removed as much soil water as the deeper drain placement (40 versus 95 cm depths; Fausey and Brehm, 1976). This may have been because of the relatively shallow placements compared in that early work. Data extracted for this review indicated relatively less reduction in drainage discharge when comparing very shallow versus shallow drainage (Figure 72d versus c). Nevertheless, there were few site-years with very shallow drainage and differences may have also been related to a different climate or cropping system (e.g., very shallow drainage studies were mainly from Ohio, Nova Scotia).

Table 41. Studies of the impact of drainage design (particularly shallow drain placement) on nitrate loss.

Study	# of site-years	Control system		Treatment system		Location	Notes
		Spacing (m)	Depth (m)	Spacing (m)	Depth (m)		
Helmers et al. (2012)	4	18	1.2	12.2	0.76	Iowa	Taintor and Kalona silty clay loams; Corn-soybean split plot
Schott et al. (2017)	5	18	1.2	12.2	0.76	Iowa	Same as Helmers et al. (2012)
Maas et al. (2022)	5	18	1.2	12.2	0.76	Iowa	Same as Helmers et al. (2012)
Singh et al. (2007) *	NA	30	1.2	10-50	0.75	Iowa	DRAINMOD modeling
Sands et al. (2008)	5	12, 24	1.2	9, 18	0.90	Minnesota	Webster silty clay loam, Nicollet clay loam; Corn-soybean
Luo et al. (2010) *	NA	12, 24	1.2	9, 18	0.9	Minnesota	DRAINMOD-NII modeling
Zhao et al. (2000) *	NA	28	1.2	14, 28, 56	1.2	Minnesota	DRAINMOD-N modeling; spacing only
Davis et al. (2000) *	NA	27	1.2	15-200	0.9, 1.2, 1.5	Minnesota	ADAPT modeling
Cooke et al. (2002)	2 (1 used)	30	1.2	15, 30	0.91 and 0.61 (very shallow)	Illinois	Drummer silty clay loam; soybean-corn
Schwab et al. (1980) *	5	12	1.0	6	0.50 (very shallow)	Ohio	Toledo silty clay; Alfalfa-grass, corn-soybean-oat; also see Fausey and Brehm (1976); Schwab et al. (1985)
Gordon et al. (2000) *	3	12	0.80	12	0.50 (very shallow)	Nova Scotia	Debert and Queens soil groups; continuous corn
Strock et al. (2011) *	NA	NA	NA	NA	NA	NA	General review
Kladivko et al. (2004) *	NA	NA	NA	5, 10, 20	0.75	Indiana	Spacing study only
Jaynes et al. (2008) *	NA	30.5	1.2	30.5	1.8	Iowa	Deep tile at 1.8 m but with the outlet at 1.2 m
Skaggs and Chescheir (2003) *	NA	NA	NA	0 - 200	0.75 – 1.50	North Carolina	DRAINMOD modeling

* not used to develop the recommended average N loss reduction value

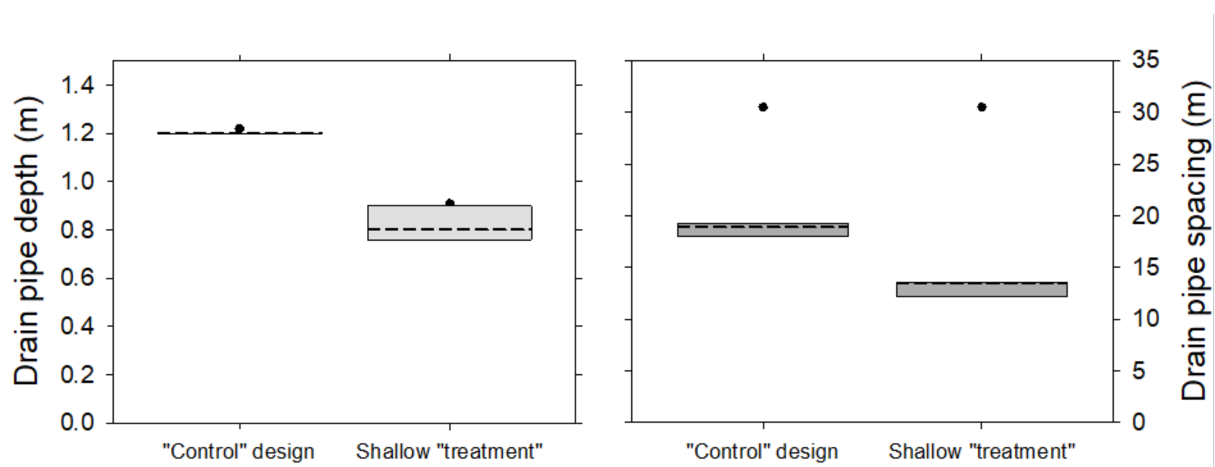

Figure 71. Range of drain placement depth (left) and spacing (right) for this shallow drainage assessment (n = 20 for each bar, from Table 41). The lack of variation in the control depth (1.2 m) was because all the site-years used that depth.

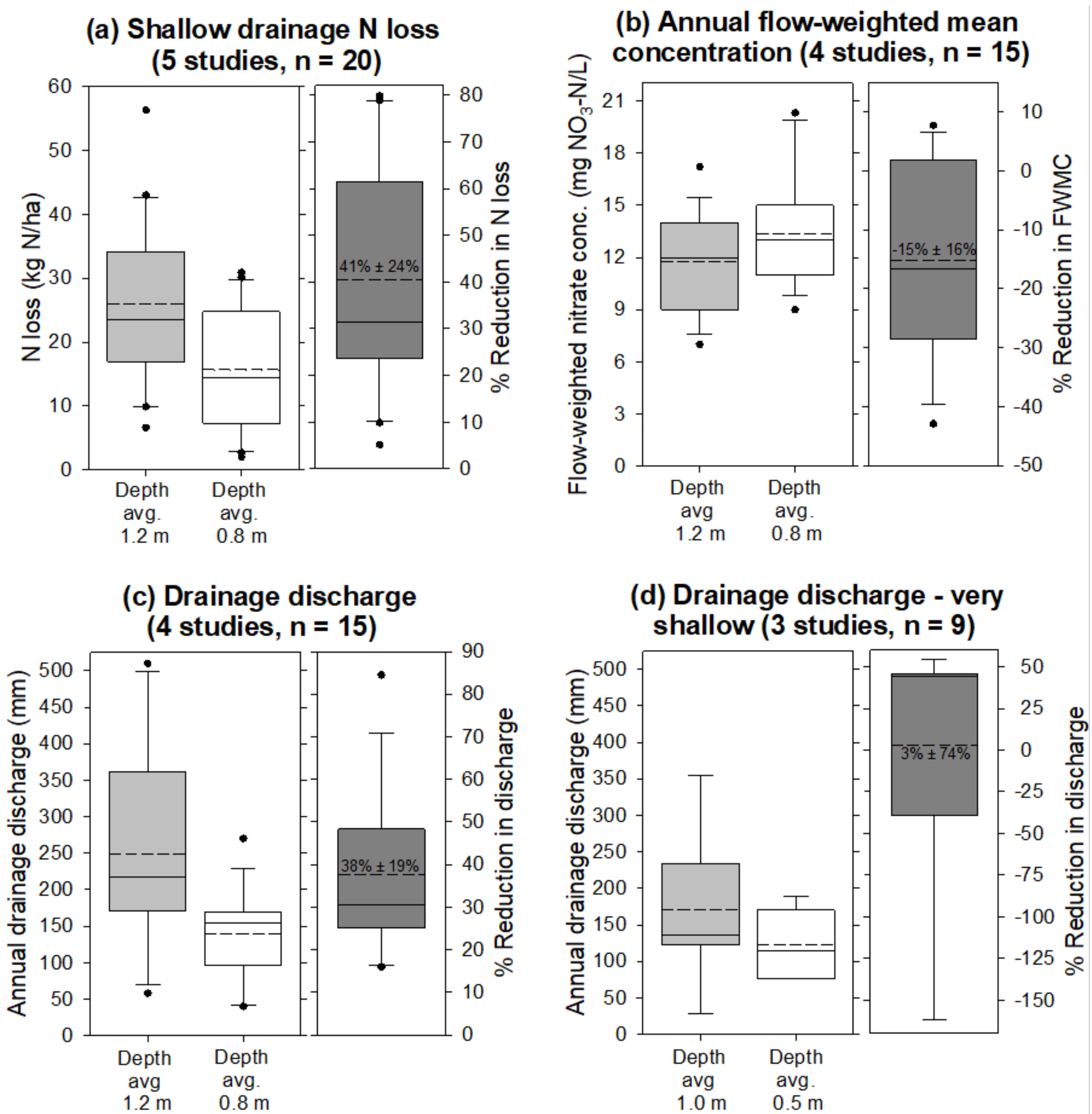

Figure 72. Nitrate-N loss and loss reduction (a), annual flow-weighted nitrate concentration (b), and drainage discharge (c) as impacted by the practice of shallow drainage. Panel (d) provides a comparison for drainage discharge from very shallow drainage (less than 2 ft from the surface; Cooke et al., 2002; Gordon et al., 2000; Schwab et al., 1980). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 41.

The practice of shallow drainage is generally thought to not impact the nitrate concentration of the drainage water (Cooke et al., 2002; Gordon et al., 2000; Sands et al., 2008). However, across the data compiled here, there was a slight increase in annual flow-weighted nitrate concentration due to this practice (Figure 72b: mean 15 ± 16% increase relative to deeper drainage depths). The deeper versus shallow placement had mean concentrations of 11.8 ± 2.8 versus 13.4 ± 3.3 mg NO_3-N/L. The concentration was numerically higher in 11 of the 15 site-years where concentration was reported, but 9 of those 11 were from one location (Crawfordsville, Iowa). Thus, these results may have been overly influenced by the Iowa site where it was hypothesized that shallow drainage may reduce the time for denitrification to occur in the soil (Schott et al., 2017).

Sands et al. (2008) pointed out that reductions in drainage discharge, and thus N loss, were provided by the practice of shallow drainage in all five years of their study (numerically, not significantly). This consistency is an important benefit of all the conservation drainage practices (e.g., Figure 13). Modeling of continuous corn over various drainage designs at Waseca, Minnesota by Davis et al. (2000) presented a contrasting view. They concluded that N fertilizer rate reduction would have a greater impact than modifying the drain spacing or depth. This current science assessment indicated that a rate reduction from 125% of the MRTN to the MRTN (from 242 to 194 kg N/ha for continuous corn) would result in a 21% N leaching reduction. That benefit was roughly half of the mean 41% calculated here for the practice of shallow drain placement. This comparison is nuanced, however, because the benefit of N rate reduction is a function of the initial and final rates.

The N leaching benefit of shallow drainage is thought to be less than that provided by controlled drainage (Helmers et al., 2012; Luo et al., 2010; Schott et al., 2017), and this was true in the current dataset (e.g., Table 36: 41% vs. 45%, respectively). Like controlled drainage, modeling has shown shallow drainage risks increasing surface runoff; there were insufficient data to analyze that possible effect in this review. Shallow drainage does generally have an advantage over controlled drainage in that it is widely applicable in situations where drainage is newly being installed and is not as restricted by land slope (Skaggs and Chescheir, 2003). It also doesn't require the management necessary for controlled drainage (Skaggs and Chescheir, 2003).

Shallow drainage presents a risk of crop yield reduction (Gordon et al., 2000; Singh et al., 2007). Across the relatively few crop yield site-years here, there was an average reduction of 9 ± 17% due to this practice (0.3 ± 0.4 Mg/ha reduction; *n* = 12; Figure 73). Three of these 12 crop yield site-years were from Nova Scotia where yields averaged approximately 2 Mg/ha across all treatments (Gordon et al., 2000). Despite these corn yield declines, the practice of shallow drainage improved yield-scaled N losses compared to placement of drains at more conventional depths (means: 3.3 vs. 3.8 kg N lost/Mg grain produced, respectively; Figure 73).

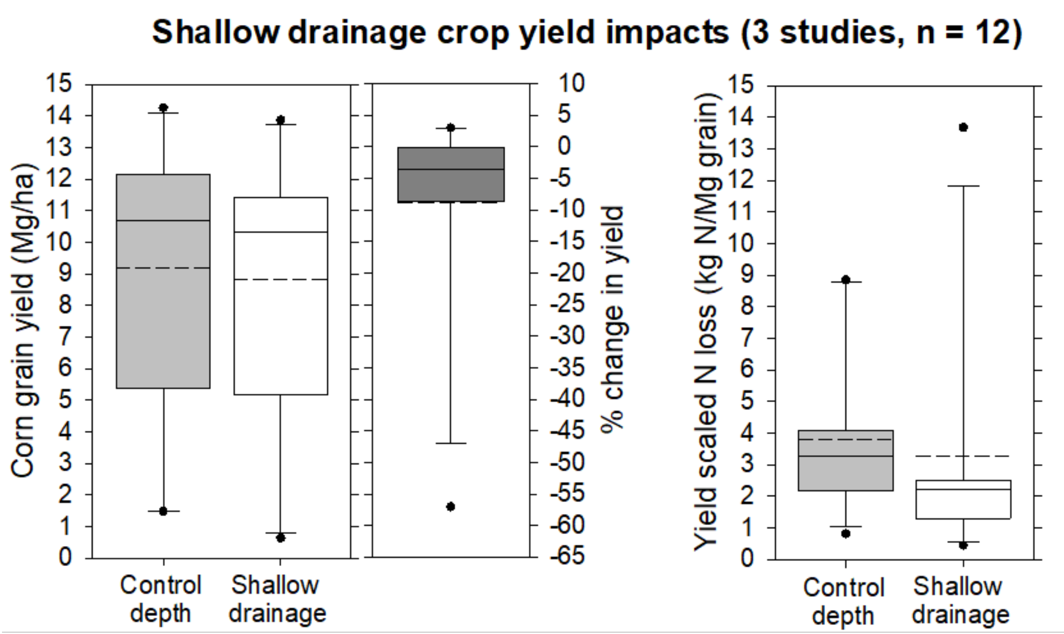

Figure 73. Corn yield impact and yield-scaled N losses of the practice of shallow drainage. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from Gordon et al. (2000), Helmers et al. (2012), and Schott et al. (2017).

Alternative underground inlet (gravel or blind inlet)

Flooding of low depressional areas which are common in many Midwestern fields will increasingly become a concern if the frequency and/or size of storm events increases in the future. There are two apparent options in these agricultural landscapes: (1) heavy crop yield penalties in these low areas will be borne (ideally leading to conversion of the area to a wetland or a perennial planting) or (2) the drainage will need to be improved using a surface inlet (AKA, surface intake, tile riser). Surface inlets are a direct physical connection from ponded water in a closed depression on the surface to the drainage system (Blann et al., 2009). Although tile drainage water tends to have relatively low sediment loads, surface inlets are the primary way sediment and associated nutrients enter a drainage system (Blann et al., 2009). There are tens of thousands of surface inlets across the US Midwest (Feyereisen et al., 2015) and these conveyance points are an underappreciated source of phosphorus loading (Tomer et al., 2010). Improved recognition of these contributions and better management of these areas could notably benefit water quality (Hall et al., 2023).

Li et al. (2017) and Feyereisen et al. (2015) both described four types of surface inlets: open inlets, perforated pipe tile risers, rock or gravel inlets, and blind inlets. This quantitative review considered both gravel inlets and blind inlets together using the general term "*alternative underground inlet*". This practice consists of an underground structure that replaces a tile riser for the purpose of reducing sediment and phosphorus transport through tile drainage systems (Ohio State University Extension, 2024). Gravel and blind inlets have similar underground piping but a gravel inlet has gravel visible at the surface whereas a blind inlet has soil at the surface. A gravel inlet could functionally become a blind inlet over time. Additional variations of alternative inlets include use of grass filter strips or a filter sock around an inlet (Shipitalo and Tomer, 2014) or including P-sorbing media or woodchips within the media bed (ILF, 2023; Wilson et al., 2024). The practice of "*side inlets*" is discussed separately below.

The NRCS describes "*underground outlets*" as conduit systems built underground to convey surface water to an outlet to reduce erosion and manage ponding (MN NRCS, 2021c). The NRCS Conservation Practice Standard 620 is not just for gravel and blind inlets placed in depressional areas of a field, however. It also applies to draining water from terraces and water and sediment control basins. Note, the terminology from the NRCS focuses on the outlet whereas terminology used here refers to the inlet (i.e., underground outlet vs gravel inlet). It should also be noted that there is no specific NRCS standard designed exclusively for alternative tile intakes. While the 620 standard can be applied to side inlet controls, the broader implementation of these practices often depends on funding from state grants, NGOs, or local drainage authorities. Local drainage authorities may incorporate these practices into their systems, allowing them to be maintained and repaired over time through the established drainage benefits process.

One important nuance is that the practice of alternative underground inlets is only considered a conservation practice if an existing surface inlet or tile riser is being retrofitted. This is because simply adding an alternative inlet to a drainage system may increase the overall sediment and nutrient loss compared to not having an inlet structure. In other words, any kind of inlet structure provides a more direct path to the outlet than through a soil profile. In this quantitative review and assessment, a perforated riser (e.g., Hickenbottom) was considered the "*control*" and a gravel or blind inlet was the "*treatment*".

Early assessments of alternative inlets tend to report 50-90% reduction of total P (Feyereisen et al., 2015; Lenhart et al., 2017; Smith and Livingston, 2013), whereas more moderate total P reductions of around 20% have followed (Ranaivoson and Moncrief, 2019; Wilson et al., 2024). The impact of alternative inlets on dissolved pollutants is even more variable. Ranaivoson and Moncrief (2019) and Williams et al. (2020) both found that soluble P and nitrate increased across a blind inlet or when comparing to a tile riser control. However, retrofitting 24 open inlets to gravel inlets in west central Minnesota resulted in a 35% decrease in soluble P concentrations (based on medians; Feyereisen et al., 2015). Most recently, Wilson et al. (2024) added a layer of woodchips to the bottom of blind inlet and was able to achieve nitrate concentration reductions of more than 40%. Surface and subsurface hydrology are inexorably linked in closed depressions which complicates the assessment of nutrient loss reduction provided by a practice like a blind inlet (Williams et al., 2020).

In terms of this science assessment, Williams et al. (2020) highlighted that most of the water quality research on blind inlets has been performed at two fields in Indiana (termed "*ADE*" and "*ADW*", Table 42; Feyereisen et al., 2015; Gonzalez et al., 2016; Smith and Livingston, 2013). This was an important gap in this review. In the most recent study at these Indiana sites, Williams et al. (2020) detected a confounding effect of lateral flow under one of the fields. This led them to conclude that the previously reported nutrient reductions for this site "*should be interpreted with caution*" (i.e., previous estimates should be broader: 0-80% nutrient reductions). Thus, the following reviewed findings, most of which predate the 2020 study, should be viewed through this cautionary lens.

In this review, 22 information sources were reviewed that covered topics of depressional areas and gravel, blind, and side inlets. Five of those contained extractable nutrient data, when both losses and concentrations were considered (Table 42). However, quantitative assessment was challenged by the inconsistent methods (e.g., concentrations versus losses reported; growing season versus annual values) as well as design variations (e.g., including woodchips), not to mention the caution that early performance reports may have been overestimated (Williams et al., 2020). So, given that reduction in flow volume at a blind inlet could provide a reduction in nutrient losses (Williams et al., 2020), several concentration reductions were used as conservative proxies for loss reduction. This was done for the Iowa study that did not report losses (Wilson et al., 2024) and for some limited concentration data from the Indiana site (n = 2 from Williams et al., 2020).

Total P removal provided by an alternative inlet averaged 41 ± 31% for the 14 site-years assessed here (including 3 concentration proxies; Figure 74a). Without the three concentration proxies, the average total P loss reduction was 54 ± 15% (n =11 from three studies). The practice of blind inlets was included in a 2017 revision of the Iowa Nutrient Strategy at a recommended P load reduction of 57 ± 28% (IDALS, 2017). Given the 2017 date of that revision and the more moderate performance reported in newer studies, a lower value as recommended here may be more conservative until additional research is performed.

Alternative inlets resulted in an average dissolved P reduction of 43 ± 29% (Figure 74b). Some caution is merited for use of this value because snowmelt periods can present extra risk for dissolved P in depressional areas (Ginting et al., 2000), and studies assessed here often only monitored during the growing season. Feyereisen et al. (2015) reported that gravel inlets across a field in Minnesota still provided SRP reduction during snowmelt periods, but the benefit was less than during the other parts of the year.

Table 42. Field studies documenting water quality at blind or gravel inlets. Conc. is concentration. Dissolved P was reported as soluble reactive P (SRP) or orthophosphate (ortho-P).

Study	# of site-years		Nutrients studied	Location	Method notes
	Loss	Conc.			
Smith and Livingston (2013)	2	0	SRP, TP, NO_3-N	Indiana	ADE and ADW sites in 2009 and 2010; May be some duplication with Feyereisen et al. (2015)
Feyereisen et al. (2015)	8	0	SRP, TP	Indiana	ADE and ADW sites in 2006-2013; Growing season, not annual; One value presented representing 8 years
Williams et al. (2020)	0	20	NO_3-N, SRP, TP	Indiana	ADE and ADW sites in 2006-2015; Growing season, not annual; Some duplication with Feyereisen et al. (2015); One median was presented representing 10 years for 2 sites, used as *n* =2 to avoid duplication;
Gonzalez et al. (2016) *	NA	NA	Pesticides only	Indiana	ADE and ADW sites in 2008-2011, 2013;
Feyereisen et al. (2015)	3	3	SRP	Minnesota	Represented retrofit of 24 inlets; Before-after study difficult to compare, used 3-y average loss three times
(Ranaivoson and Moncrief, 2019)	1	1	TSS, SP, PP, TP, NH_4-N, NO_3-N	Minnesota	Duplicated with Ranaivoson et al. (2002)
(Wilson et al., 2024)	0	1-2	NO_3-N (2); TP, ortho-P (1)	Iowa	Growing season, not annual; Included woodchip layer

* not used to develop the recommended average loss reduction values

Blind inlets are not a recommended practice for nitrate treatment. On average, alternative inlets increased nitrate losses by 60 ± 93% (or, -60% reduction) across these relatively limited 5 site-years which included 2 concentration proxies (Figure 74c). These data excluded the two site-years from Iowa where an enhanced denitrification layer was used (Wilson et al., 2024). Nitrate concentration reductions exceeded 40% in both of those years, which highlighted the importance of further design assessments to enhance denitrification and/or add P-sorbing media for this relatively new practice.

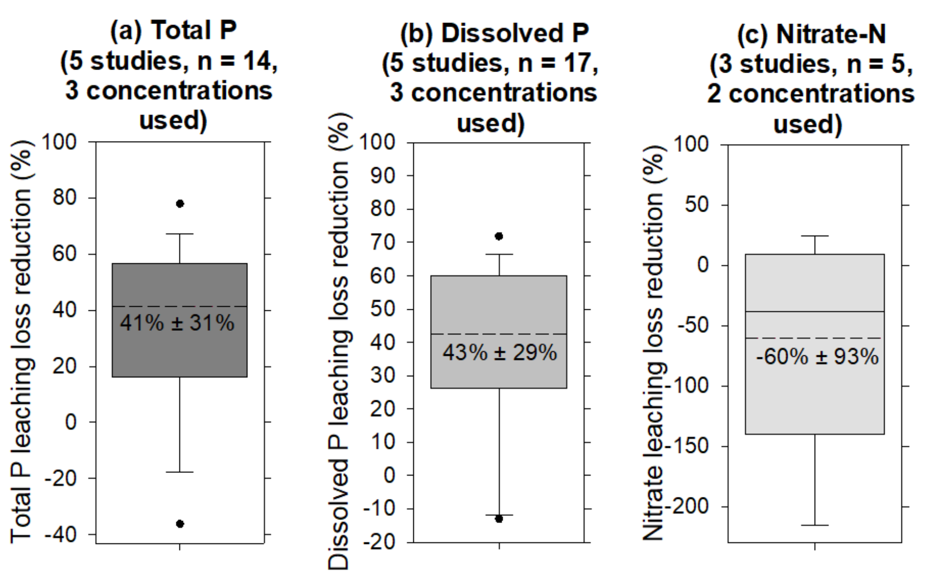

Figure 74. Relative total phosphorus (a) dissolved phosphorus (b), and nitrate (c) loss reductions (%) due to an alternative inlet (a gravel or a blind inlet). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 42.

An obvious benefit of gravel or blind inlets is that they don't need to be farmed around like a traditional tile riser (ILF, 2023; Ohio State University Extension, 2024). Thus, this practice is a practical retrofit for conventional tile risers. Installation tends to be relatively inexpensive (e.g., $500; ILF, 2023; Lenhart et al., 2017) and their service life is on the order of 10 years (Ranaivoson et al., 2002; Williams et al., 2020). Concerns about water ponding and associated crop impacts in the depressional area are some of the main barriers for adoption of this practice (Williams et al., 2020). Smith and Livingston (2013) reported that although water may be ponded for longer compared to using a traditional surface inlet, there was no negative crop impact during their study in Indiana. Further crop yield assessments for alternative inlets will be helpful.

Gravel/blind inlets are a difficult practice to effectively monitor in the field, and there are significant data gaps that need to be addressed for this practice. Firstly, annual studies are needed to document alternative inlet performance, and it would be helpful to report individual years rather than multi-year averages. For Minnesota, annual studies should specifically include snowmelt periods to capture any tradeoffs between nutrient forms. Ginting et al. (2000) was one of the first to assess the impact of ponding and surface inlets in depressional areas in cold climates. They reported settling of particulates was an important benefit, but prolonged ponding allowed release of dissolved nutrients.

Secondly, studies should be designed to monitor loads or losses in addition to concentrations because of confounding subsurface + surface hydrology in depressional areas (e.g., Williams et al., 2020). Finally, it would be useful to have a broader discussion about how to best develop robust and transparent experimental designs for this practice that include an effective control. The Indiana experimental configuration (ADE, ADW) was very sophisticated but was also complicated. Before-after experimental designs (Feyereisen et al., 2015) and above-below sampling designs (Ranaivoson and Moncrief, 2019) also both have benefits and limitations.

One final note is that reducing sediment losses (and associated nutrient losses) starts well up-gradient of depressional areas and surface inlets. Oolman and Wilson (2003) used modeling to show the most effective way to reduce sediment loss through a surface inlet in Minnesota was to practice no tillage compared to conventional tillage.

Side inlets

The practice of a side inlet is a surface inlet that drains small depressions directly through a streambank or side-of-ditch berm. These drainage features also may be referred to as "*drop pipes*" or simply as "*culverts*" (Peterson, 2013). Side inlets are classified by the NRCS as a type of grade stabilization structure (MN NRCS, 2022b). Krider et al. (2014a and 2014b) nicely illustrated the use of a side inlet, but there may be confusion about this term in practice (Lewandowski, 2010). The terminology of "*side inlet*" appears to be relatively limited to the state of Minnesota; in this review, only studies from Minnesota used this term.

From a water quality perspective, side inlets can be designed and placed to provide temporary runoff storage and sedimentation (Peterson et al., 2014). The benefit provided by a given side inlet depends on the ponded surface area to watershed area ratio which should be at least 1% (Peterson et al., 2014). Krider et al. (2014b) compared several side inlet configurations (flush pipe, Hickenbottom riser, rock inlet, coil inlet, and the rock weir) at Lamberton, Minnesota and found that sediment removal was linked with retention time. Side inlet studies have focused on hydrologic modeling and sediment retention; nutrient data were insufficient to review this practice further in this science assessment.

Drainage water recycling

Drainage water recycling was one of the newest conservation drainage practices in this science assessment. This practice involves on-farm storage of drainage water in a pond or reservoir and then use of that stored water via irrigation later in the season (Frankenberger et al., 2017). This provides a crop production benefit as well as a water quality benefit (Hay and Helmers, 2023). A variety of configurations can be used for drainage water recycling, for example use of a center pivot versus subsurface irrigation ("*subirrigation*") or variations with reservoir design and pumping systems (Hay et al., 2021). Thus, "*drainage water recycling*" may most appropriately be considered a systems approach to water management rather than a specific prescribed practice per se.

The colloquialism "*rain makes grain*" (Temple, 2023) aptly summarizes the benefit of crop production in the humid Midwest where supplemental irrigation has typically not been necessary. However, the timing and amount of rainfall in this region does not always coincide with the timing of crop water needs (Frankenberger et al., 2017), and accordingly the amount of irrigated land has increased in recent decades. In Minnesota, irrigated acres have increased by more than 30% over the past 15 years (from less than 500,000 reported acres in 2007 to more than 650,000 acres in 2022; IAM, 2023). Irrigation expansion is predominantly occurring on coarse textured soils (i.e., not tile drained areas), but this nevertheless illustrates growing knowledge, infrastructure, and technical capacity for this type of water management.

The prevalence of subsurface drainage in the state (i.e., 8.25 million acres) combined with increasingly hot summers, variable precipitation patterns (Reinhart et al., 2019), and growing irrigation capacity creates an opportunity for the practice of drainage water recycling. The NRCS recognized this growing potential across humid areas, and in 2020, renamed their irrigation tailwater practice standard to include recovery of drainage water (CPS 447: Irrigation and drainage tailwater recovery; USDA NRCS, 2023). At this time, this practice standard has not been adopted in Minnesota.

Sixteen information sources covering the practice of drainage water recycling were reviewed for this science assessment, and data were extracted from five (Table 43). These studies documented the water quality benefit of this practice in two ways: (1) by comparing the practice to a free drainage control, or (2) by assessing the nutrient balance across the water recycling reservoir. The older studies from Ontario did not directly refer to this practice as "*drainage water recycling*," but in fact, did recycle drainage water through a controlled drainage-subirrigation system and compared that treatment to a free drainage control (Drury et al., 2009; Drury et al., 1996; Tan and Zhang, 2011; Tan et al., 2007). These early plot-scale studies provided the benefits of replication as well as assessment of both surface runoff and drainage water quality. In this quantitative review, nutrient losses in both runoff and drainage from those studies were summed into a total nutrient outflow, and those total nutrient losses were compared for free drainage versus the drainage water recycling practice (termed "*controlled drainage-subirrigation*" in the original studies; Table 43).

Meyer (2023) recently noted those early studies didn't assess the nutrient impact of the water storage system itself. To address this gap, the newer studies from Iowa and North Carolina used the "*reservoir nutrient balance*" approach (Hay and Helmers, 2023; Meyer, 2023; Moursi et al., 2023). Because drainage water recycling is a newer practice, there is still some discussion about how to best develop these reservoir-centric water and nutrient balances. Meyer (2023) considered seepage around the reservoir whereas Moursi et al. (2023) did not (Table 43).

Table 43. Field studies documenting water quality at locations where drainage water was recycled.

Study	Nutrient site-years		Controlled drainage?	Irrigation method	Method	Location	Notes
	Nitrate or TN	DP, TP					
Drury et al. (1996)	12 (NO_3^-)	0	Yes	Subirrigation	Replicated plots: free drainage control	Ontario	Summed nutrient losses in drainage and runoff; four tillage treatments x 3 years
Drury et al. (2009)	8 (NO_3^-)	0	Yes	Subirrigation	Replicated plots: free drainage control	Ontario	Summed nutrient losses in drainage and runoff; two fertilizer treatments x 4 years
Tan and Zhang (2011)	0	4 (DRP), 4	Yes	Subirrigation	Replicated plots: free drainage control	Ontario	Summed nutrient losses in drainage and runoff; Tan et al. (2007) presents NO_3^- but only for drainage, not runoff
Meyer (2023)	4 (TN)	4 (TRP), 4	No	Center pivot	Reservoir inflow – (outflow + seepage)	Iowa	Load reported, normalized by irrigated area; 3 sites; also see Hay and Helmers (2023)
Moursi et al. (2023)	2 (TN)	2 (orthoP), 2	Yes	Subirrigation	Reservoir inflow – outflow	North Carolina	Load reported, normalized by area of irrigated plot

In addition to methodological differences between these studies, the sites also differed in context and application. For example, the drained and irrigated areas at the some of the Iowa sites did not directly overlap (in contrast to the Ontario plot studies) and some of the reservoirs received water pumped from drainage ditches (i.e., not direct recycling of on-farm drainage water). Reservoir size was strongly correlated with the irrigated area across all sites (R^2: 0.83; regression not shown, Figure 75a), but the ratio of reservoir: irrigated area was ≤ 7% for the larger-scale studies and higher for plot-scale studies (Figure 75b). Moreover, while most of the studies recycled the stored water using subirrigation, the three Iowa locations used center pivots. Use of center pivots would most likely be the context for this practice in Minnesota.

The practice of drainage water recycling provided a 51 ± 21% N loss reduction when averaged across all site-years in the available studies (Figure 76a; median: 51%). This was lower than the 72% average developed for the six site-years where the reservoir nutrient balance method was used (Figure 76c). That method will likely be the preferred method in future studies because it allows assessment of drainage water recycling separately from the practice of controlled drainage. However, five of those six site-years were noted by the authors to be relatively dry. The water quality benefit of this practice is a function of the quantity of water retained and used (Hay et al., 2021), and reservoir outflows were relatively low in these dry years. Thus, the 72% average (Figure 76c) may be skewed high due to weather conditions this limited dataset.

No other state nutrient strategy has included a recommended N reduction value for the practice of drainage water recycling yet. However, other states have established procedures to add new practices to nutrient strategy documents as data becomes available (e.g., IL EPA, 2023). Those periodic updating procedures do not exist in Minnesota. Therefore, it was important to use this review to establish drainage water recycling as a legitimate practice within Minnesota's conservation toolbox even though this review represented a mixture across methods and contexts.

While the recommended 51% N reduction for the practice of drainage water recycling may be premature (necessarily), these data provided significant evidence of the ability of this practice to reduce N loss. One of the benefits of this science assessment comprehensively reviewing a number of conservation practices is that the individual practices can be viewed in context of each other. For example, it is likely that over the long term, drainage water recycling would provide a higher N reduction efficiency than N management practices (i.e., 4-10%, Table 4) but less than prairie conversion (94%, Table 4). Nevertheless, the recommended 51% for drainage water recycling, along with all other recommended values developed here, should be periodically reviewed and revised as necessary.

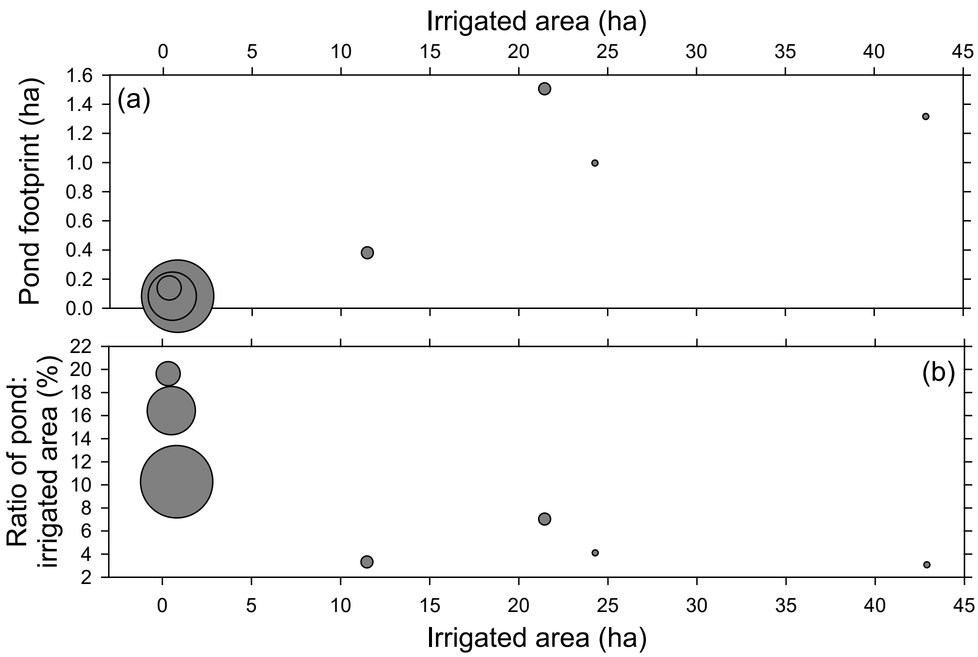

Figure 75. Design details from the drainage water recycling sites (from Table 43). The bubble size indicates the relative number of site-years at a given site; the number of site-years is also shown in Table 43 (n = 1-12). Note Meyer (2023) studied three sites. The irrigated area for the replicated plot studies was the plot size times the number of irrigated plots.

Drainage water recycling provided a 22 ± 57% total P loss reduction when averaged across all available site-years (Figure 77b; median: 27%). There was no notable benefit to dissolved P (-1 ± 80%, median: -7%, Figure 77a). The impact of drainage water recycling on P in water has been reported to be variable (Meyer, 2023), but these data confirmed that, on average, there was a total P benefit.

Digging deeper into these water quality benefits, larger reservoirs relatively increase nutrient load reductions and provide more storage for irrigation (Reinhart et al., 2019). The benefits of drainage water recycling will also differ across the Midwest (and year-to-year) due to variation in the timing of monthly precipitation and drainage flow relative to crop water use (Reinhart et al., 2019). The timing within the year of water storage could also matter for nutrient treatment within the reservoir itself. Moursi et al. (2023) observed nitrate concentrations in reservoir water in North Carolina decreased due to the summer's longer retention times and warm temperatures. It is unknown how those results may translate to Minnesota's climate.

Drainage water recycling's main agronomic benefit of crop yield enhancement due to supplemental irrigation is complemented by the co-benefits of improved yield stability and resilience (Nelson et al., 2021). Across the Transforming Drainage multi-state project, this practice improved corn yield 64% of the time (Willison et al., 2021) and the average increase was 1.2 Mg/ha (i.e., 19 bu/ac above the 147 bu/ac mean for free drainage equates to 13%; Nelson et al., 2021). As expected, yield benefits were greater in drier years (Tan et al., 2007; Willison et al., 2021). Further development of irrigation management guidelines is needed because irrigation on poorly drained soils with shallow water tables fundamentally differs from irrigation in arid climates (Hay et al., 2021; Reinhart et al., 2019). There may also be a N rate effect when water limitations are removed using this practice (Drury et al., 2009). Fine tuning water management and N management, in tandem, are important for irrigated crop production in Minnesota (Sharma et al., 2023).

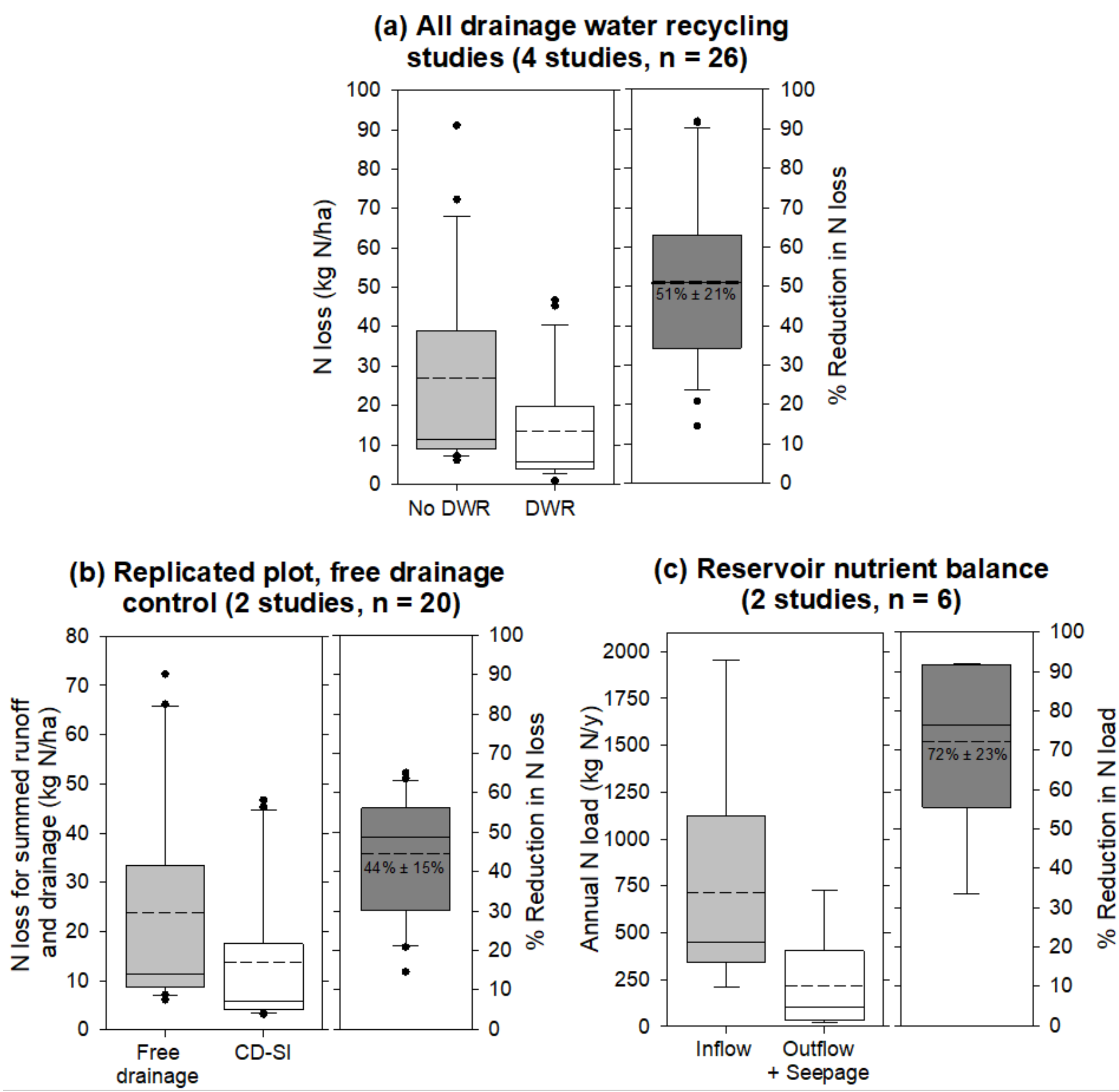

Figure 76. N loss and loss reduction from all drainage water recycling (DWR) site-years (a); N loss and reduction from plot-scale studies of controlled drainage-subirrigation (CD-SI) (b); and N load and reduction from reservoir nutrient balance studies (c). The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 43.

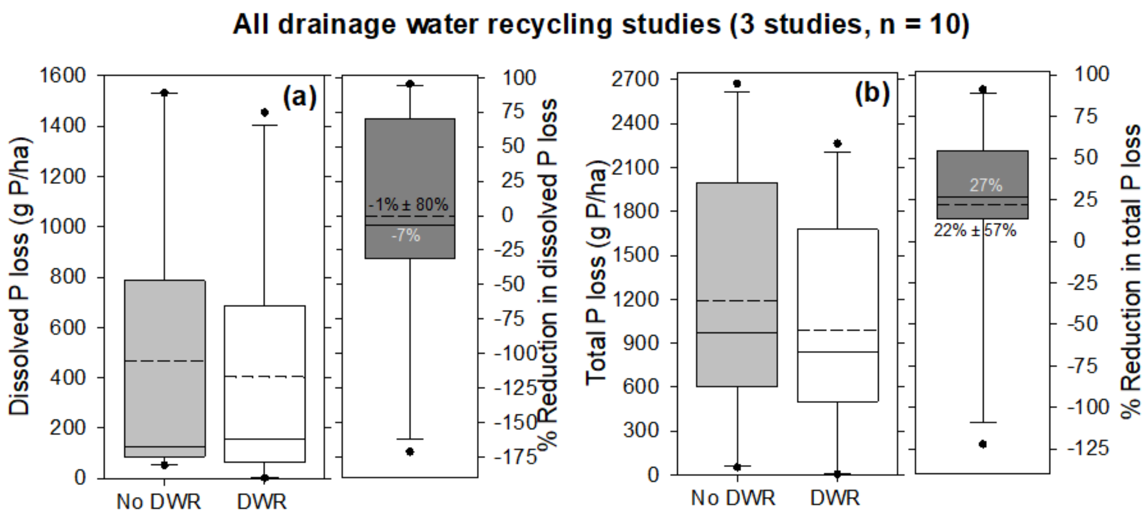

Figure 77. Dissolved (a) and Total P (b) loss and loss reduction from all drainage water recycling (DWR) site-years. The solid and dashed lines are the median and mean, respectively; box edges are the interquartile range; dots are outliers. Data were sourced from studies listed in Table 43. Loads reported in the reservoir balance studies were normalized by irrigated area.

The obvious barrier for the practice of drainage water recycling is its large capital expense (Frankenberger et al., 2017; Reinhart et al., 2019). Permitting may also be required (Frankenberger et al., 2017). However, this practice has a relatively long design life of 25-50 years and thus the benefits would be long-lasting (Reinhart et al., 2019).

Water storage provided via the practice of drainage water recycling has the potential to reduce downstream flooding (Frankenberger et al., 2017; Mitchell et al., 2023). However, a more thorough assessment of the potential for this benefit is needed. An on-farm reservoir would need to be managed using strategic drawdowns to provide storage at critical times. It is also likely that ponds would be needed throughout the watershed to provide a benefit at scale (Frankenberger et al., 2017). Other environmental benefits such as wildlife habitat around the reservoir or energy production (e.g., floating solar panels; Hay et al., 2021) also merit more consideration. One final exciting note is that supplemental irrigation could provide new possibilities for double- and relay-cropping and integration of perennials into the Midwestern landscape (Reinhart et al., 2019). Baker et al. (2012) eloquently outlined this vision where increased water storage linked with irrigation in the Midwest could help stabilize food production, support more resilient cropping systems, improve water quality, and mitigate climate risk.

Monitoring at more drainage water recycling sites, across wet and dry years, using consistent methods (e.g., Abendroth et al., 2022) is needed to refine understanding of the water quality and agronomic benefits of this practice. Understanding the impact of this practice on stream flow (Hay et al., 2021) and streambank stability (Gupta, 2024) will require field experiments coordinated with landscape-scale computer modeling. Additional suggested research topics for this practice include development of irrigation guidance for Midwestern climates and soil, monitoring of greenhouse gasses at drainage storage reservoirs, and more detailed economic analyses (Hay and Helmers, 2023; Meyer, 2023).

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Supplementary Information

Constructed Wetlands:

Constructed wetlands are widely recognized as an effective edge-of-field practice for mitigating nitrate-nitrogen losses to water resources. The focus of this section is on quantifying their efficiency in Minnesota, where nutrient reduction strategies are a priority to protect water quality. The review aimed to compile data solely from Minnesota over the past decade. However, due to the limited availability of localized studies, data from neighboring Midwest states were included to provide a more comprehensive analysis.

Although research by Lenhart et al. (2016) and Gordon et al. (2021) reported high nitrate-nitrogen reductions in Minnesota wetlands, 68 and 67%, respectively, much of this reduction occurred through seepage into the underlying groundwater rather than remaining within the wetland itself. Consequently, these findings were not included in the dataset for this review.

The wetlands studies assessed here were located in agricultural areas dominated by corn-soybean cultivation. The reduction efficiencies reported in these studies were estimated by comparing the amounts of nitrate-nitrogen in inflow and outflow water. The median was used here because the data ranges tended to be skewed, making it a more appropriate measure of central tendency. The criteria for inclusion of studies included: 1) primarily studies conducted in Minnesota or surrounding states in the Midwest region; 2) studies that have measured year-round nitrate-nitrogen in water; and 3) studies that have information on catchment-wetland area ratio.

Three studies met these criteria: two from Illinois (Groh et al., 2015 and Lemke et al., 2022) and one from Iowa (Crumpton et al., 2020). The wetland-to-catchment area ratios for these studies ranged from 3.3% to 4.0% for Groh et al., 3.0% to 6.0% for Lemke et al., and 0.25% to 3.5% for Crumpton et al. Illinois had a greater mean efficiency of 49 ± 7.4%, which might be attributed to the greater wetland-to-catchment area ratios or other site-specific factors such as hydrology and nutrient loads. A greater wetland-to-catchment area ratio generally enhances the ability of wetlands to intercept and process nutrient loads effectively, as it increases its capacity relative to the drainage area (Forgrave et al., 2024). The reduction efficiency in Iowa was lower with 35%, which may reflect its lower ratios likely combined with high nitrate loads. Combined, the three studies covered a total of 109 wetland-years (i.e., more than 1 wetland was assessed in some years) and resulted in an overall median nitrate-nitrogen reduction of 41.7 ± 6.4%, with reductions ranging from 35% to 57%, This range is higher than the 30% reported in Iowa (IDALS, 2024) and includes the 50% efficiency reported in Illinois (INLRS, 2023). These findings reveal that, even within the same region, variability in wetland performance exists, which might be influenced by site-specific factors such as wetland design, hydrology, nutrient load levels, and wetland-to-catchment area ratios.

The inclusion of data from Illinois and Iowa provides valuable insights, expanding the understanding of constructed wetlands and providing the potential for this strategy to mitigate nitrate losses in Minnesota. However, this assessment demonstrates the importance of increasing Minnesota wetland-specific research to develop tailored strategies for nutrient reduction. While the combined dataset offers a robust regional perspective, it also emphasizes the need for localized studies to address gaps in knowledge and optimize wetland performance under Minnesota's specific environmental and agricultural conditions.

Table 1. Details of Constructed Wetland Studies for Nitrate-Nitrogen Reduction Efficiency: Authors, Publication Years, and Results.

				Reduction (%)
Study	First author	year	location	NO_3^--N
Nitrogen Removal and Greenhouse Gas Emissions from Constructed Wetlands Receiving Tile Drainage Water	Groh et al.	2015	IL	57
Nitrogen and phosphorus removal using tile-treatment wetlands: A 12-year study from the midwestern United States	Lemke et al.	2022	IL	42
Illinois overall median ± standard error				49.1±7.4
Water quality performance of wetlands receiving nonpoint-source nitrogen loads: Nitrate and total nitrogen removal efficiency and controlling factors	Crumpton et al.	2020	IA	35.0
Overall median ± standard error				41.7±6.4

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Science Assessment of Cropland Practices for Minnesota's Nutrient Reduction Strategy: Part 2 Phosphorus

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List of abbreviations:

BMP, Best Management Practice
P, Phosphorus
TP, Total Phosphorus
DRP, Dissolved Reactive Phosphorus
NRS, Nutrient Reduction Strategy
STP, Soil Test Phosphorus
APP, Ammonium Polyphosphate
DAP, Diammonium Phosphate

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Executive Summary

Phosphorus (P) losses from agricultural lands continue to be a leading cause of water quality degradation in Minnesota and the broader Upper Midwest. The Minnesota Nutrient Reduction Strategy (NRS), first released in 2014, established a target of a 45% reduction in phosphorus loads to the Mississippi River Basin. This updated review focuses on evaluating and synthesizing the latest scientific evidence on the effectiveness of field-based best management practices (BMPs) for reducing total phosphorus (TP) and dissolved reactive phosphorus (DRP) losses from agricultural systems.

Objective and Approach

The review aimed to reassess and update phosphorus BMP reduction efficiencies using peer-reviewed field studies, with a preference for research conducted in Minnesota and surrounding agroclimatic regions. A screening process was used to include only studies that met specific criteria: treatments with comparable initial soil test P levels, measured (not simulated) runoff data, and a minimum of seasonal or annual P loss reporting. Studies based solely on simulated rainfall were excluded from the primary synthesis; however, their results are summarized separately (Chapter 2) to provide context for the differences between controlled and natural rainfall conditions. In total, 52 studies were reviewed, and 21 were included in the quantitative synthesis.

Updated Reduction Efficiencies

The updated analysis produced revised reduction efficiency estimates for previously recognized strategies and introduced new ones not assessed in the original NRS. Recommended statewide efficiencies are based solely on field-based studies; simulated rainfall results (Chapter 2) and constructed wetlands (Chapter 4, Item 7) are presented only for context and are not included in Table I. Notably, P drawdown, spring P application, subsurface P application, and cover crops emerged as the most consistently effective strategies for reducing both TP and DRP losses. In contrast, the use of organic (manure-based) P sources increased losses of both P forms, while ridge till, no-till systems, and soil P build-up showed a trade-off: reductions in TP were accompanied by increases in DRP, highlighting the balance between erosion control and soluble P mobilization.

Table I. Summary of recommended P reduction efficiencies developed in this review. Negative values refer to increased response to BMPs.

Strategies	Reduction efficiency	
	TP	DRP
Drawdown	25.5 ± 31.4%	60.5 ± 4.4%
Source (organic vs inorganic)	-50.5 ± 34.4%	-53.6 ± 33.2%
Timing (spring vs fall)	18.4 ± 2.02%	33.6 ± 11.1%
Placement (subsurface vs surface)	40.2 ± 18.9%	58.8 ± 28.8%
Conservation tillage: ridge till vs moldboard plow	47.0 ± 3.63%	-180 ± 226%
No-till vs conventional (i.e., chisel plow and ridge till)	5.40 ± 0.18%	-72.9 ± 82.0%
Cover crops (terminated/suppressed in spring before planting)	21.5 ± 16.8%	7.93 ± 31.6%
Soil P Build-Up on Low-P Soils*	21.9 ± 14.7%	-175 ± 129%

* P loss reduction is expected to occur only during the build-up phase, when soils transition from low to optimum P levels

Trade-Offs and Yield Impacts

The report includes a preliminary yield impact assessment, based only on studies that also evaluated P losses, to help stakeholders understand potential agronomic trade-offs. While some strategies, such as P drawdown and manure-based P applications, showed positive yield effects, others, like no-till and cover crops, were associated with modest yield declines. However, a more complete evaluation would require reviewing all studies comparing these strategies, regardless of whether they measured P losses, to better estimate their true impact on crop yields.

Key Findings

- BMP effectiveness is highly dependent on site-specific conditions, including initial soil test P (STP), hydrology, and cropping history.
- Many BMPs that reduce TP also increase DRP, particularly where surface P accumulation occurs.
- Constructed wetlands show variable long-term P retention and may become net P sources, so they are included for context only and not assigned a statewide reduction efficiency.
- Most studies are short-term and regionally limited.

Research Needs

- Year-Round and Long-Term Assessments: Existing BMP evaluations rarely incorporate both rainfall and snowmelt conditions. It is crucial for future assessments to explain BMP performance differences across years.
- Comparative and Multi-Site Studies: Most current field studies lack direct comparisons to conventional systems or are limited to one location. Investments in paired, regionally distributed research trials are needed.
- Modern Tillage Systems: Ridge-till evidence may not reflect current practices. Research is needed to evaluate P losses under strip tillage and other low-disturbance systems.
- Infrastructure for Monitoring: Limited field plots with tile drainage and flow monitoring systems restrict comprehensive data collection.
- A more balanced investment in phosphorus research, comparable to the support for nitrogen, is essential to progress on both nutrient fronts.

Conclusion

The revised synthesis of phosphorus BMPs demonstrates both progress and limitations in achieving the goals of the Minnesota Nutrient Reduction Strategy. While some practices show high potential, none are universally effective, and trade-offs, especially between TP and DRP, are common. A more integrated, long-term, and regionally coordinated approach will be necessary to meet water quality goals while supporting productive agriculture.

Chapter 1: Overview and Approach

This review was developed to synthesize and assess the effectiveness of phosphorus (P) best management practices (BMPs) in reducing P losses from agricultural fields, with a primary focus on surface runoff pathways. The process began with a targeted and intensive search for peer-reviewed literature that evaluated BMP impacts on total phosphorus (TP) losses. When available, dissolved reactive phosphorus (DRP) data were also extracted and included in the analysis to provide a more complete picture of P dynamics in agricultural systems.

This review prioritized field-based studies that measured year-round P losses, including those from snowmelt events, which are particularly relevant in cold-climate regions such as the U.S. Upper Midwest. In some cases, studies that did not include snowmelt but assessed P losses throughout the full growing season were also accepted. Notably, this review excluded studies that reported P losses based solely on simulated rainfall experiments. While such studies provide valuable insight into P mobilization and process-level responses to conservation practices (Timmons et al., 1973; Andraski et al., 1985; Edwards and Daniel, 1994; Bundy et al., 2001; Zhao et al., 2001; Andraski et al., 2003; Tabbara, 2003; Daverede et al., 2004; Allen and Mallarino, 2008; Kovar et al., 2011; Gilley et al., 2013; Jokela et al., 2016; Schuster et al., 2017; Sherman et al., 2021), they typically assess short-term runoff events under controlled conditions. As a result, they may not capture the hydrologic variability and cumulative impacts of BMPs over an entire growing season or calendar year. However, due to the useful insights these studies can provide, they will be summarized in a separate section of the report for contextual reference and comparison. In addition to field-based BMPs, constructed wetlands were included in the broader review as an edge-of-field treatment practice; however, because their hydrologic function and monitoring requirements differ from in-field studies, their evaluation is summarized separately in Chapter 4 (Item 7).

Studies were included only if they reported P loss values derived from measured runoff, either surface runoff, tile-drained water, or both. Most studies included surface runoff, with a smaller number reporting losses from subsurface (tile-drained) systems. While the focus was on TP due to its broader regulatory and environmental relevance, studies that provided DRP data were also summarized where available.

To ensure valid comparisons, studies were excluded if there were substantial differences in initial soil test P levels between BMP and conventional treatments. This criterion was critical to avoid confounding treatment effects with baseline soil fertility conditions. Only studies that compared treatments under comparable soil P levels were retained, allowing for more accurate attribution of P loss differences to management practices rather than inherent soil variability.

To maintain consistency, studies were screened based on several criteria:

1- The comparison must involve a more conservation-oriented practice (e.g., no-till, spring P application, cover crops) defined as the BMP, evaluated against a conventional or less conservation-oriented system.
2- P losses must be measured under field conditions, not simulated.
3- P loss must be quantified over a growing season or full year.
4- Studies must provide enough detail to extract or calculate mean annual P loss values per treatment.

Constructed wetland studies were screened separately because they evaluate catchment-scale inflow-outflow P loads rather than plot-scale BMP comparisons; therefore, they are included as a distinct practice category in Chapter 4 (Item 7).

The literature search prioritized studies conducted in Minnesota and the broader U.S. Midwest, given the regional focus of this report. However, due to climatic and agronomic similarities, studies from adjacent Canadian provinces (e.g., Manitoba, Ontario, and Saskatchewan) were also considered. In one case, a study from Pennsylvania was included because its design closely aligned with regional conditions and provided long-term field data consistent with the inclusion criteria.

As of this review, 52 studies have been identified and reviewed. Of these, approximately 40% (n = 21) were used directly in the quantitative analysis of BMP effectiveness (Figure 1). The remaining studies were referenced for context, included in background discussion, or excluded due to unmet inclusion criteria (e.g., insufficient data, simulation-based estimates, non-equivalent comparisons, differences in initial soil P levels, snowmelt runoff data only, DRP from tile-drained water only) (Schuman et al., 1973; Hansen et al., 2000; Brye et al., 2002; Thoma et al., 2005; Tiessen et al., 2010; Daigh et al., 2015; Hoover et al., 2015; Williams et al., 2016; Duncan et al., 2017; Singh et al., 2018; Wiens et al., 2019; Trentman et al., 2020; Sherman et al., 2021; King et al., 2023; Bender & Lenhart, 2024).

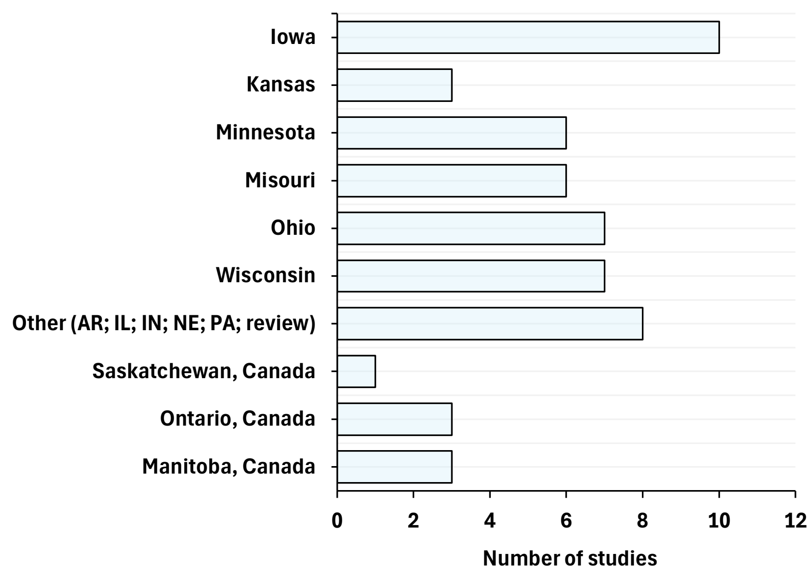

Figure 1. Number of reviewed studies by location.

From a disciplinary perspective, six studies were conducted in Minnesota, with Iowa contributing the most studies (n = 10), more details on Figure 1. The Journal of Environmental Quality accounted for the largest number of published studies (22 out of 53), followed by Soil Science Society of America Journal (SSSAJ) and Transactions of the ASABE, each contributing 4 studies, and the Journal of Environmental Management and Journal of Soil and Water Conservation, which each contributed 3 studies (Figure 2).

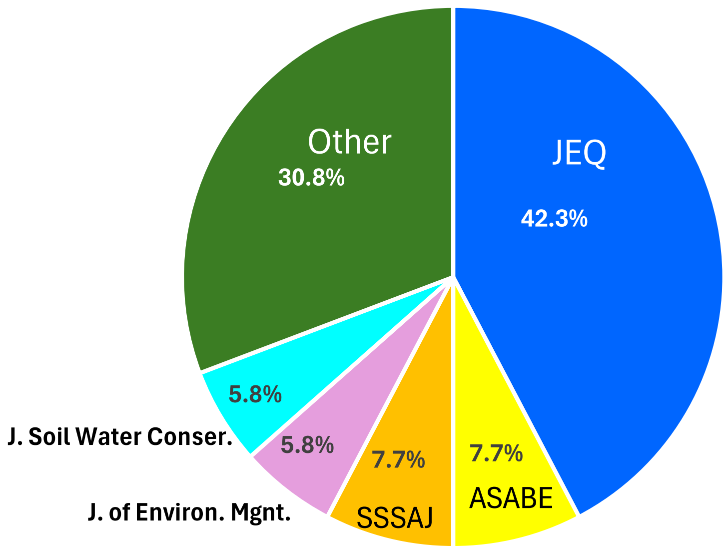

Figure 2. Percentage distribution of reviewed studies by journal source.

Estimation of Reduction Efficiencies

For each study, reduction efficiency was estimated using the following formula:

$$\text{Reduction Efficiency (\%)} = \left(\frac{\text{Conventional loss} - \text{BMP loss}}{\text{Conventional loss}}\right) \times 100$$

This calculation was applied to the mean annual P loss load values (kg ha^{-1} yr^{-1}) reported for each treatment. When more than one site or year was reported within a study, the treatment means were averaged before calculating the reduction efficiency. For constructed wetlands, reduction efficiencies were calculated using inflow and outflow P loads, consistent with wetland performance literature, rather than BMP vs. conventional comparisons used for field practices.

To derive an overall estimate across studies, site-year weighted means were calculated:

$$\text{Weighted mean} = \frac{\sum (E_i \times N_i)}{\sum N_i}$$

Where:

- E_i = Effect size (e.g., % TP reduction) in study *i*
- N_i = Number of site-years in study *i*

Table 1. Hypothetical example:

Study	Site-years (N_i)	TP Reduction (%) (E_i)	$E_i \times N_i$
A	4	10	40
B	6	20	120
C	10	35	350
Total	**20**		**510**

$$\text{Weighted mean} = \frac{\mathbf{510}}{\mathbf{20}} = \mathbf{25.5\%}$$

This ensured that studies contributing more years of data had greater influence on the final estimate. Similarly, the standard deviation (SD) was estimated using a site-year weighted approach to capture the variability in reduction effectiveness across locations and conditions.

In cases where yield impacts of BMPs were also reported (e.g., for corn grain), the same approach was applied to estimate BMP yield reduction or gain relative to conventional practices. Although yield data were available in only a subset of studies, this information is essential to evaluate potential agronomic trade-offs and balance environmental and production outcomes.

Chapter 2: BMP Performance Under Simulated Rainfall: Relevance and Comparison to Natural Conditions

Background and Rationale

This section complements the phosphorus nutrient reduction strategies (P-NRS) assessment presented in this report, which primarily focused on field studies conducted under natural rainfall conditions (see Chapter 3). Those field-based studies were prioritized due to their relevance to the local climate and hydrology and the ability to evaluate year-long P losses through edge-of-field monitoring. These natural rainfall datasets are used in Chapter 3 to produce realistic, field-representative P reduction efficiencies for BMPs across the Midwest.

During the development of this report, reviewers and stakeholders raised an important question: Why were studies based on simulated rainfall excluded? Their concern may have originated from the perception that these studies, despite their limitations, provide valuable, controlled insight into how BMPs function under worst-case runoff scenarios. Furthermore, simulated rainfall studies are commonly cited in literature to demonstrate high P reduction potential for certain BMPs, and omitting them could be interpreted as a gap in evidence.

In response, we conducted a focused review of simulated rainfall studies evaluating total phosphorus (TP) and dissolved reactive phosphorus (DRP) runoff losses. Results were categorized according to event timing and compared with the natural rainfall reduction efficiencies presented later in Chapter 3. The purpose of this comparison (Table 2; Figure 3) is to assess whether including simulated rainfall data would meaningfully change the field-based P reduction efficiencies used for nutrient reduction planning in Minnesota, and whether it would be appropriate to integrate these datasets.

Structure and Categorization of Simulated Rainfall Data

We categorized phosphorus loss reduction estimates into two temporal groupings:

1) Initial simulated rainfall: Refers to data collected from the first rainfall simulation following P application, typically within 30 days. This captures the “worst-case” or “peak-loss” scenario, often used to assess short-term BMP efficacy under high-risk conditions.

2) Cumulative simulated rainfall: Refers to data aggregated across two or more rainfall simulation events over the growing season. These events occurred between 1 week and 6 months after P application and aimed to represent seasonal trends in P loss dynamics.

This classification allowed us to examine the extent to which high P losses in initial events skew overall BMP efficiency estimates in simulated rainfall studies. For context, natural rainfall-based efficiency estimates (presented in Chapter 3) were included for comparison wherever applicable (Table 2).

Phosphorus Application Placement: Surface vs Subsurface

The first BMP assessed was phosphorus placement, comparing surface applications (broadcast without incorporation) to subsurface placement (including banding, injection, and incorporation). This is one of the most widely studied BMPs for P loss mitigation.

Eight studies were reviewed: Tabbara et al. (2003), Allen and Mallarino (2008), Daverede et al. (2004), Jokela et al. (2016), Gilley et al. (2013), Shuster et al. (2017), Sherman et al. (2021), and Yuan et al. (2018). Not all studies had sufficient data to contribute to both initial and cumulative categories. Cumulative estimates were derived from studies reporting ≥2 runoff events (all except Tabbara et al. 2003), while initial estimates were based only on studies specifying the first event within one month of P application. Gilley et al. (2013) and Yuan et al. (2018) were excluded from the initial category due to lack of timing detail.

The results show a clear pattern: simulated rainfall studies overestimate initial P reduction efficiencies relative to simulated cumulative efficiencies and efficiencies under natural conditions (see Chapter 3). Initial TP and DRP reduction efficiencies were 71.1 ± 22.8% and 80.4 ±18.6%, respectively. For cumulative events, these decreased to 44.2 ± 22.0% for TP and 72.1 ± 14.6% for DRP. In contrast, natural rainfall studies produced more conservative estimates of 40.2 ± 18.9% for TP and 58.8 ± 28.8% for DRP (Table 2; Figure 3).

Table 2. Summary of P reduction efficiencies based on simulated rainfall conditions, with natural rainfall efficiencies (from Chapter 3) included for comparison. Negative values indicate increased P losses associated with the BMP. Dash indicates data not available.

Strategies	Natural rainfall efficiencies (%)		Simulated rainfall efficiencies (%)				
			Initial[§]		Cumulative[‡]		Studies
1. P management	TP	DRP	TP	DRP	TP	DRP	Studies
Placement (surface vs subsurface[¶])	40.2 ± 18.9%	58.8 ± 28.8%	71.1 ± 22.8%	80.4 ± 18.6%	44.2 ± 22.0%	72.1 ± 14.6%	9
2. Tillage intensity:							
Ridge till vs moldboard[£]	47.0 ± 3.63%	-180 ± 226%	54.4 ± 6.26%	24.7 ± 59.3%	25.7 ± 48.3%	-108 ± 173%	2
Chisel till vs moldboard	-	-	63.6 ± 35.6%	16.7 ± 106%	52.3 ± 24.4%	-63.9 ± 109%	3
No-till vs conventional[Σ]	5.40 ± 0.18%	-72.9 ± 82.0%	88.0 ± 15.0%	-31.4 ± 200%	57.6 ± 24.9%	-91.5 ± 207%	7
3. Cover crops	21.5 ± 16.8%	7.93 ± 31.6%	33.0 ± 29.3%	26.1 ± 44.7%	26.8 ± 30.3%	4.29 ± 43.4%	1 Limited data
4. Contour buffer strip	-	-	-	-	56.1 ± 12.2%	33.1 ± 29.4%	1 Limited data

[¶] Broadcast and incorporation, injection, and banded P application were considered subsurface placement.

[§] Initial rainfall: Refers to P loss data collected after the first simulated rainfall event following P application. These events typically occurred within 30 days of the application.

[‡] Cumulative rainfall events: Refers to P loss data summed across two or more simulated rainfall events occurring over the growing season. Events ranged from 1 week to 6 months after P application.

[£] Ridge tillage systems were more common when many of the classic tillage-P loss studies were conducted. In present-day Minnesota, ridge till has largely been replaced by strip tillage and other low-disturbance systems, indicating a need for updated research on P losses under current tillage practices.

[Σ] Any tillage was considered conventional, regardless of intensity.

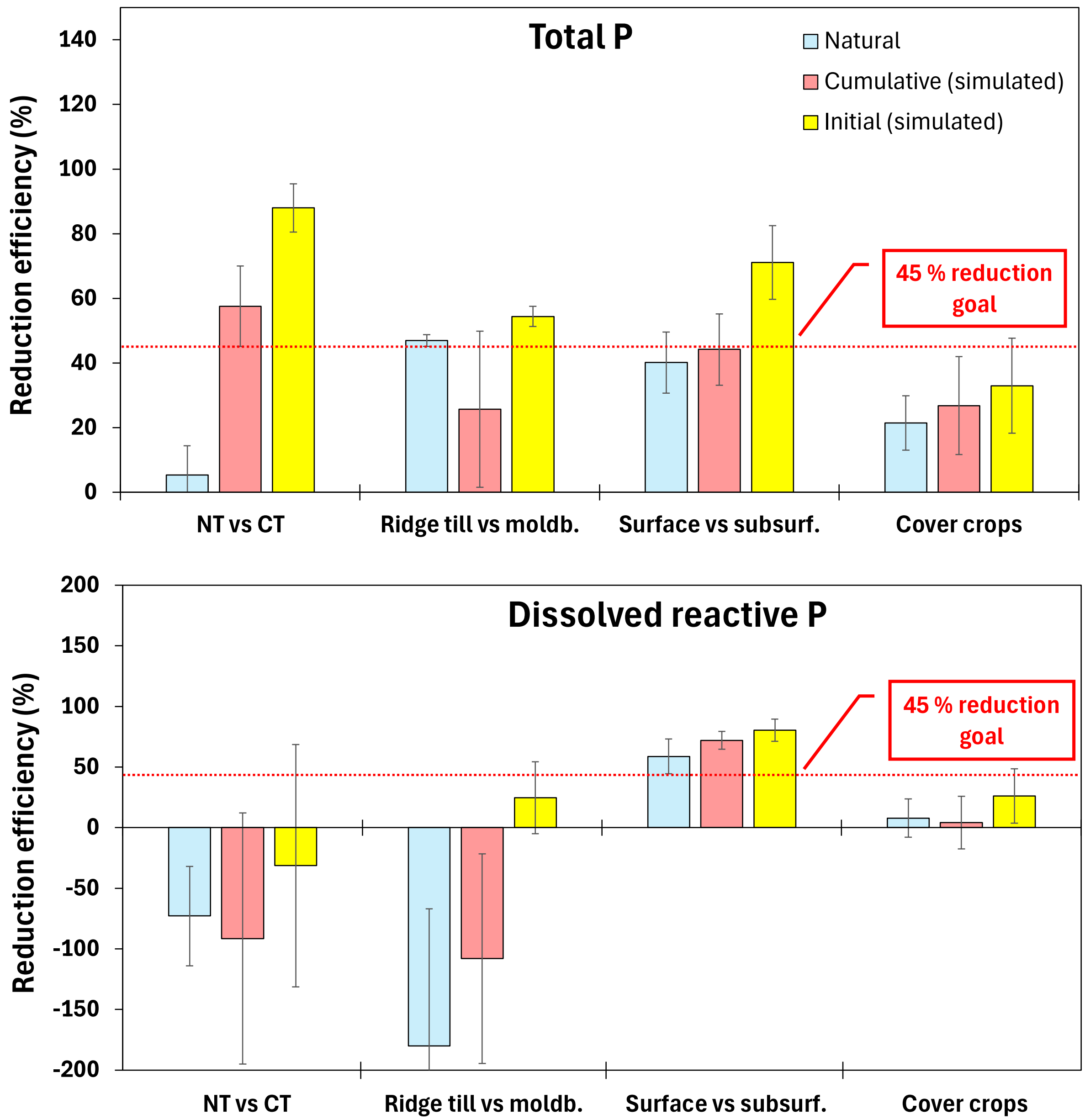

Figure 3. Percent change in total (TP) and dissolved reactive phosphorus (DRP) losses associated with various phosphorus (P) management strategies. Positive values indicate reductions in TP losses, while negative values indicate increases. Bars represent the mean percent change, and error bars denote the standard deviation of the mean for each strategy. *NT, no tillage; CT, conventional tillage.

This discrepancy likely reflects the intensity and timing of simulated events. Simulated rainfall was typically applied within days of P application and at medium to high intensities, conditions that favor maximum loss of surface-available P. In contrast, natural rainfall-runoff in the Midwest is less frequently driven by such high-intensity events and often does not occur shortly after P application, especially in the fall, allowing time for phosphorus to become less mobile.

Tillage Intensity Comparisons

We next examined the influence of tillage intensity on P losses, comparing different tillage systems: ridge till vs moldboard plow, chisel plow vs moldboard, and no-till vs conventional tillage (where "conventional" refers to systems with soil disturbance, including chisel or moldboard plowing). Studies included were Mueller et al. (1984), Laflen and Tabatabai (1984), Andraski et al. (1985, 2003), Bundy et al. (2001), Zhao et al. (2001), Eghball et al. (2000), and Yuan et al. (2018). Not all studies contributed to every tillage category due to differences in comparison treatments or data availability.

Ridge Till vs Moldboard

Only two studies (Zhao et al. 2001; Andraski et al. 1985) reported data for this comparison. Initial simulated rainfall efficiency was 54.4 ± 6.3% for TP and 24.7 ± 59.3% for DRP. Cumulative values dropped to 25.7± 48.3% for TP and –108 ±173% for DRP. Natural rainfall data, presented in Chapter 3, showed 47.0 ± 3.6% for TP and –180 ± 226% for DRP, indicating high variability and even increases in DRP losses in both natural and simulated conditions.

Chisel Till vs Moldboard

This comparison, not evaluated in the natural rainfall analysis of Chapter 3, was supported by three studies: Mueller et al. (1984), Laflen and Tabatabai (1984), and Andraski et al. (1985). Initial simulated reductions were 63.6 ±35.6% for TP and 16.7 ± 106% for DRP, while cumulative results were 52.3 ± 24.4% and –63.9 ± 109%, respectively. These results highlight that chisel tillage can reduce particulate-bound P losses relative to moldboard, but may not consistently reduce DRP, especially under cumulative or lower-intensity runoff conditions.

No-Till vs Conventional

This comparison yielded the most data points (7 studies). Initial simulated efficiencies were 88.0 ±15.0% for TP and –31.4 ±200% for DRP, while cumulative values were 57.6 ±24.9% and –91.5 ±207%. Natural rainfall values were much lower: 5.4 ± 0.2% for TP and –72.9 ± 82.0% for DRP (presented in Chapter 3). These results reaffirm that no-till effectively reduces soil erosion and TP losses, but may increase DRP losses, likely due to surface stratification of soluble P in undisturbed soils. Importantly, simulated rainfall tends to inflate the initial benefit for TP reduction and potentially underrepresents the longer-term DRP trade-offs that become evident under natural rainfall.

Additional BMP Categories: Cover Crops and Contour Buffer Strips

Only one study was found for each of the following categories, so they are flagged as “limited data” and interpreted with caution.

Cover Crops

Kovar et al. (2011) was the only simulated rainfall study assessing cover crops. Initial reduction efficiencies were 33.0 ± 29.3% for TP and 26.1 ± 44.7% for DRP. Cumulative values were slightly lower: 26.8 ± 30.3% and 4.3 ± 43.4%, respectively. These are generally consistent with natural rainfall results of 21.5 ± 16.8% for TP and 7.9 ± 31.6% for DRP (presented in Chapter 3), though more data are needed to confirm these trends.

Contour Buffer Strips

Eghball et al. (2000) assessed narrow grass hedges (switchgrass) as contour buffer strips. Only cumulative data were available, showing 56.1± 12.2% TP and 33.1± 294% DRP reduction. These results suggest substantial potential for TP mitigation, although the high variability in DRP highlights the need for additional studies to confirm consistency under varying conditions.

Limitations and Uncertainties in Simulated Rainfall Studies

One consistent finding across all reviewed simulated rainfall studies is the need for caution due to methodological limitations:

- Simulated rainfall events typically involve medium to high-intensity rainfall, which may not reflect regional rainfall patterns where runoff can occur under low to moderate intensities.
- These studies consistently lacked critical metadata necessary for interpreting P loss outcomes. Missing or inconsistently reported details included: the interval between P application and rainfall, whether runoff loads were summed or averaged, initial soil P concentrations, residue management, and crop presence. These factors directly affect P mobilization and runoff potential, making it difficult to standardize or compare efficiencies across studies.

These flags are summarized in Table 3 and should be considered when interpreting results.

Table 3. Notes and Limitations in Simulated Rainfall Studies.

Author (Year)	Location	Strategy category	Limitation/Note
Laflen and Tabatabai (1984)	IA	Tillage	Initial soil P not reported.
Eghball et al. (2000)	IA	Tillage and Buffer Strips	Switchgrass hedges (~0.75 m); rainfall timing not reported.
Tabbara (2003)	IA	placement	Different P rates used for each source.
Allen and Mallarino (2008)	IA	placement	Only highest P rate assessed; 24 h rainfall event used due to data limitations.
Kovar et al. (2011)	IA	Cover crop	P rates differed by injection method.
Daverede et al. (2004)	IL	placement	Results averaged across P sources and rates.
Yuan et al. (2018)	IL	Placement and Tillage	Runoff loads averaged across events, not summed.
Zhao et al. (2001)	MN	Tillage	Soil P differed between tillage and P treatments; ridge left more residue than moldboard.
Gilley et al. (2013)	NE	placement	Manure timing not reported; no-till vs disked treatments.
Shuster et al. (2017)	NE	placement	Five rainfall events (1–45 days post-application); crop presence not reported; manure rate based on N, P applied rate not specified.
Mueller et al. (1984)	WI	Tillage	1978 samples contaminated; manure not comparable across treatments; conventional till undefined.
Andraski et al. (1985)	WI	Tillage	Rainfall timing not reported.
Bundy et al. (2001)	WI	Tillage	Rainfall timing not reported; shallow till vs chisel plow with different residue levels.
Andraski et al. (2003)	WI	Tillage	Conducted on historically specific tillage systems.
Jokela et al. (2016)	WI	placement	Initial soil P not consistent between treatments.
Sherman et al. (2021)	WI	placement	Manure applied on triticale cover crop; no-till vs conventional not fully separated.

* Complete reference information is found in the reference section at the end of the report

Final Thoughts: Should Simulated and Natural Rainfall Studies Be Combined?

The evidence suggests that while simulated rainfall studies provide valuable insight into short-term BMP performance under controlled conditions, they tend to overestimate P loss reduction efficiencies, particularly for initial events following P application. This is especially true for surface-placed P, where early, intense rainfall can mobilize a disproportionately large share of the applied P.

In contrast, cumulative simulated rainfall efficiencies, derived from studies simulating multiple runoff events over time, are generally more aligned with natural rainfall results, though they remain more variable due to small sample sizes and methodological inconsistencies.

Based on this analysis, we recommend against combining simulated initial datasets and natural rainfall datasets when calculating nutrient reduction efficiencies for statewide or watershed-scale planning purposes. Instead, initial simulated rainfall data should be used to support mechanistic understanding, conduct sensitivity analyses, or inform risk assessment of worst-case scenarios. However, cumulative simulated rainfall data results from well-documented studies may be cautiously included as supporting evidence, particularly where natural rainfall data are scarce.

Overall, these findings support the continued prioritization of natural rainfall-based field studies for baseline nutrient reduction estimates, while recognizing the supplemental value of simulated studies in experimental design and mechanistic modeling contexts.

Chapter 3: Results Summary

This section summarizes field-based phosphorus (P) loss reduction efficiencies for selected best management practices (BMPs) as reviewed in the updated Minnesota Nutrient Reduction Strategy (NRS) analysis. The goal was to reevaluate these strategies using available studies and updated synthesis methods to provide more accurate and regionally relevant estimates. Emphasis was given on total phosphorus (TP) and dissolved reactive phosphorus (DRP) losses based on peer-reviewed literature across the Midwest and similar agroclimatic regions. The evaluation also revisits the original NRS (MPCA, 2014) estimates for context. These natural rainfall-based results serve as the primary field-scale P reduction values used in this report and are referenced in Chapter 2 when comparing simulated and natural rainfall outcomes.

Table 4 presents the updated TP reduction efficiencies alongside the original MPCA (2014) estimates. For example, while the 2014 strategy estimated cover crops would reduce TP losses by 29%, new data suggest a slightly lower average of 21.5% ± 16.8%. Conservation tillage was previously estimated to reduce TP by 63%; the revised estimate is 47.0% ± 3.63%. Other revisions include P drawdown (from 17% to 25.5% ± 31.4%) and subsurface P placement (from 24% to 40.2% ± 18.9%). Constructed wetlands information, an edge-of-field treatment practice, is included in Table 4 for context; however, because of uncertainty in long-term phosphorus retention, their effectiveness results are summarized separately in Chapter 4 (Item 7) and are not used in statewide reduction efficiency calculations.

The current assessment also incorporates new strategies not evaluated for P losses in the original report. These include the comparison of organic versus inorganic P sources, P application timing (spring vs. fall), no-till vs. conventional tillage, and the opportunity to reduce TP losses during the manure-based soil P build-up phase on low-STP soils.

Table 4. Summary of recommended P reduction efficiencies developed in this review. Negative values refer to increased response to BMPs. § indicates values from the original NRS report (MPCA, 2014).

Strategies	Reduction efficiency					
	§TP losses original NRS	TP	DRP	Site-years	Studies	Assessment
1. P management						
Drawdown	17%	25.5 ± 31.4%	60.5 ± 4.4%‡	18	1	Surface runoff
Source (organic vs inorganic)	---	-50.5 ± 34.4%	-53.6 ± 33.2%	51	4	Surface and tile drainage
Timing (fall vs spring)	---	18.4 ± 2.02%	33.6 ± 11.1%	15	3	Surface runoff
Placement (surface vs subsurface)	24%	40.2 ± 18.9%	58.8 ± 28.8%	6 Limited data	1	Surface runoff
2. Tillage intensity						
Conservation tillage: ridge till vs moldboard plow	63%	47.0 ± 3.63%	-180 ± 226%	2 limited data	1	Surface runoff
No-till vs conventional¶	---	5.40 ± 0.18%	-72.9 ± 82.0%	34	2	Surface runoff
3. Cover crops	29%	21.5 ± 16.8%	7.93 ± 31.6%	22	7	Surface and tile drainage
4. Constructed wetlands	---	Estimates for context only*		50	4	Surface and tile drainage
Conditional opportunity for reducing P losses:						
Soil P Build-Up on Low-P Soils¥	---	21.9 ± 14.7%	-175 ± 129%	6 limited data	2	Surface runoff

‡ Based on flow-weighted mean concentration.
¶ Chisel and ridge till were assumed to be conventional for this comparison.
¥ Sub agronomic recommended P rate vs recommended or over application of P as manure.
P loss reduction is expected to occur only during the build-up phase, when soils transition from low to optimum P levels.
* The TP (26.1 ± 19.3%) and DRP (61.8 ±13.5%) reductions for constructed wetlands are reported in Chapter 4 (Item 7) for contextual comparison only and should not be interpreted as recommended statewide reduction efficiencies. Both Illinois and Iowa nutrient reduction strategies elected not to assign a phosphorus retention value for constructed wetlands due to limited evidence and high uncertainty in long-term consistency and representativeness (IDALS, 2025; ILEPA, 2023). Following guidance from MDA and MPCA, these values are provided solely for context and are not included in the final statewide summary reduction estimates.

Practices to reduce total phosphorus (TP) runoff losses vary in their effectiveness depending on site-specific factors and implementation context. As shown in Figure 4, some strategies, such as soil P drawdown, P subsurface placement, and ridge-till, achieved TP reductions near or above the 45% goal established by the Mississippi River/Gulf of Mexico Hypoxia Task Force, while others, including timing adjustments and cover crops, offered more modest benefits. Although ridge tillage is the system most commonly evaluated in the literature, it has largely been replaced in Minnesota by strip tillage. Because strip tillage involves less soil disturbance and does not create raised ridges that slow sheet flow, its TP reduction performance may differ. Current evidence is not sufficient to assume equivalent benefits, highlighting the need for updated research on modern strip till systems. These findings suggest that while individual practices contribute to P reduction, combining multiple BMPs may be necessary to meet regional water quality targets.

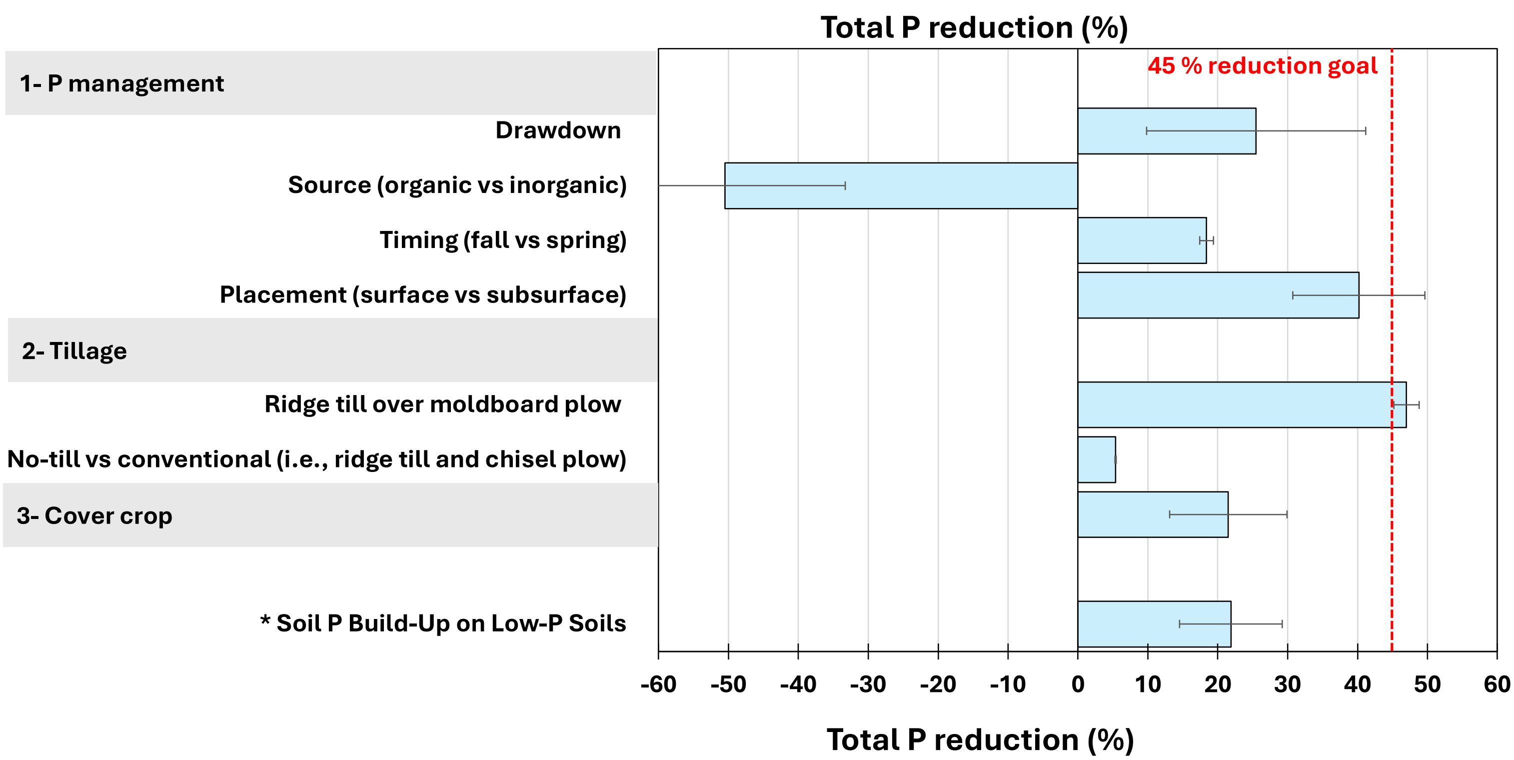

Figure 4. Percent change in total phosphorus (TP) runoff losses associated with various phosphorus (P) management strategies. Positive values indicate reductions in TP losses, while negative values indicate increases. Bars represent the weighted mean percent change, and error bars denote the standard error of the mean for each strategy. *Conditional opportunity for reducing P losses

Management practices also differ in their impact on dissolved reactive phosphorus (DRP), which is more immediately bioavailable and poses a greater risk to water quality. According to Figure 5, some strategies that effectively reduced TP, such as no-till or conservation tillage, also led to increased DRP losses. This underscores the importance of designing nutrient strategies that address both P forms and balance erosion control with soluble P management.

Understanding how these practices affect crop productivity is critical for encouraging widespread adoption. Yield impacts associated with each strategy are summarized in Figure 6, which illustrates that some practices, such as soil P drawdown and manure-based P sources, were associated with yield increases. In contrast, others like no-till showed modest declines. Although not a direct focus of the NRS, yield effects influence farmer decisions and must be considered when designing programs to improve both agronomic and environmental outcomes.

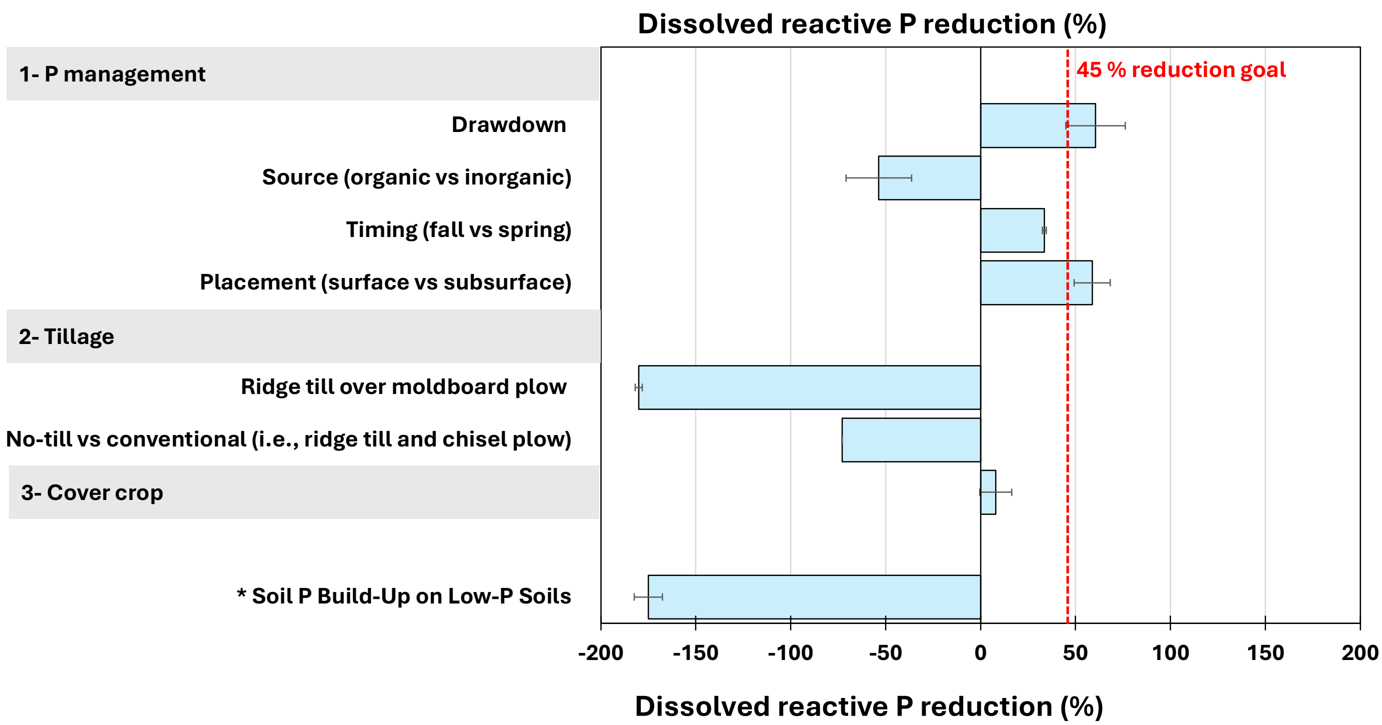

Figure 5. Percent change in dissolved reactive phosphorus (DRP) runoff losses associated with various phosphorus (P) management strategies. Positive values indicate reductions in TP losses, while negative values indicate increases. Bars represent the weighted mean percent change, and error bars denote the standard error of the mean for each strategy. *Conditional opportunity for reducing P losses

The updated P reduction efficiency estimates provided here (Part 2 NRS report) are recommended for future use in scenario development, economic assessments of BMP adoption, and estimates of potential impacts on P loads to the Mississippi River Basin and other regional watersheds. These field-based estimates may also help inform future revisions of the Minnesota Phosphorus Loss Index by providing more recent, regionally representative phosphorus loss data. The estimates were derived by synthesizing peer-reviewed field studies that primarily reported total edge-of-field P losses, combining both surface runoff and tile drainage pathways when available. When studies reported P losses separately by pathway, surface runoff and tile drainage contributions were summed to represent the total P loss from the field system, regardless of the hydrologic route. As a result, these updated estimates better reflect the overall P load reductions expected at the field edge and provide a more comprehensive basis for evaluating BMP effectiveness at watershed scales.

This approach differs somewhat from that used in the Part 1 NRS report, where P losses, including DRP contributions from agricultural drainage, were assessed separately by water pathway (e.g., surface runoff vs. tile drainage) for each BMP. Those estimates were rigorous and valuable, especially given the limited P loss studies available. However, for most BMPs, the updated Part 2 NRS synthesis offers combined pathway estimates that are more directly applicable to P load reduction goals.

An exception applies to two strategies: controlled drainage and gravel/blind inlets. For these practices, P loss reductions were estimated specifically based on tile drainage water because these BMPs are designed to manage subsurface water flow. While they can influence surface runoff by improving infiltration and reducing overland flow, their primary mechanism targets subsurface drainage. Therefore, the Part 1 NRS report estimates are retained: 30% ± 29% TP and 30% ± 40% DRP reduction for controlled drainage, and 41% ± 31% TP and 43% ± 29% DRP reduction for gravel/blind inlets.

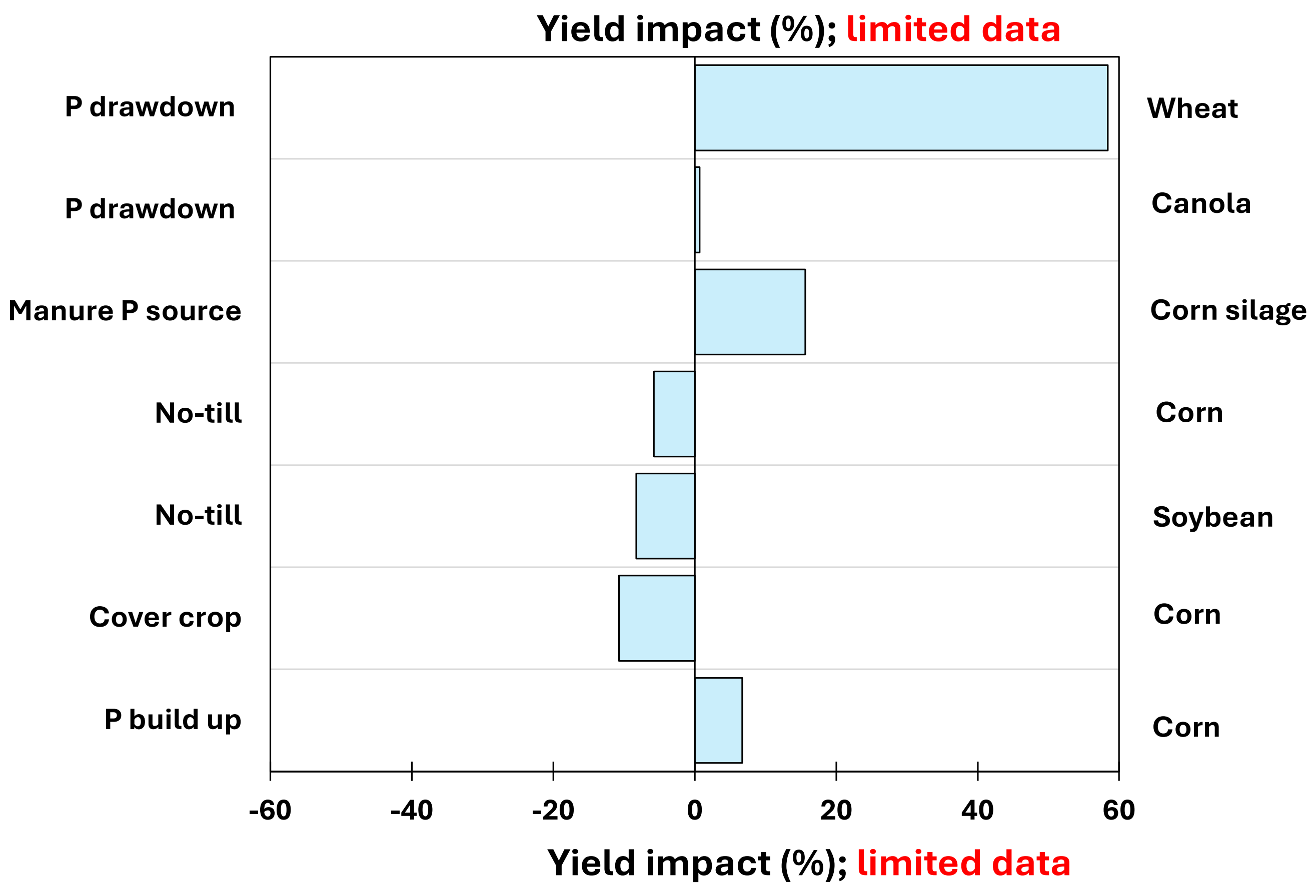

Figure 6. Yield impacts (%) associated with different phosphorus (P) management strategies. Negative values indicate yield reductions, while positive values indicate yield gains. Multiple entries for "No-till" and "P drawdown" reflect results from separate crops.

Key Takeaways:

- **Context matters**. BMP effectiveness depends heavily on initial conditions such as soil test P (STP), drainage system type, and crop management history. For example, drawdown is most effective in high-P soils, while manure-based build-up is only applicable to low-STP fields.
- **Trade-offs between TP and DRP are common**. Practices that reduce erosion (e.g., no-till, manure application) often lower TP losses but increase DRP due to surface P accumulation. Integrated systems like cover crops can help address this.
- **Few strategies consistently meet the 45% reduction target**. While ridge-till and P drawdown exceed or approach this threshold under certain conditions, most strategies fall short on their own and may need to be stacked or combined.

- **Current Minnesota assessments are geographically and methodologically limited**. Most studies are short-term, lack replication across regions, and do not consistently evaluate both runoff and tile drainage losses.
- **Infrastructure is lacking for comprehensive evaluations**. Paired plot drainage systems and regionally distributed field trials are essential to expand the capacity of the state for BMP research.
- **Constructed wetlands show high variability and uncertain long-term phosphorus retention.** Limited Minnesota data, regional differences with Illinois studies, and evidence that mature wetlands can become net P sources make it inappropriate to assign a statewide TP reduction efficiency; therefore, wetland estimates are included for context only.

Suggested Future Research

In developing an effective P reduction strategy, several critical research gaps must be addressed:

a) **Long-term, year-round assessments are essential**. Evaluations including rainfall and snowmelt-driven runoff will improve our understanding of seasonal variability in P loss. Over time, this will help explain shifts in BMP performance and clarify differences between historical and current NRS results.

b) **Comparative, multi-site BMP studies are lacking**. Most existing studies or assessments (e.g., Discovery Farms temporal assessments) are either site-specific or lack a conventional comparison. Expanding comparative research is necessary to quantify the actual benefits of BMP adoption over time.

c) **Modern tillage systems require updated evaluation.**
Much of the existing literature compares ridge till to conventional systems, even though ridge till is no longer widely used in Minnesota. Current practices such as strip tillage and other low-disturbance approaches lack equivalent field-based evaluations of P loss, representing a key research need.

d) **d) The Minnesota Phosphorus Index (MNPI) requires modernization.**
A recent evaluation of the MNPI (Reitmeier et al., 2024) confirmed that the tool remains directionally correct but has notable limitations in estimating particulate P, snowmelt losses, and runoff. However, the underlying model has not been updated since 2006. New field-based phosphorus datasets, including those synthesized in this report, could support a revision of the Index, which is also referenced in Minnesota's Chapter 7020 Feedlot Rules.

e) **Uneven investment in P vs. N research**. Much of the recent efforts in nutrient management have prioritized nitrogen, leaving phosphorus BMPs comparatively understudied. Rebalancing investments is essential to make meaningful progress.

Together, these research priorities reflect the need for a more comprehensive and long-term approach to phosphorus loss assessment in Minnesota and the broader Upper Midwest.

Chapter 4: Priority BMPs for Reducing Phosphorus Losses

1- Phosphorus drawdown strategy

The phosphorus (P) drawdown strategy aims to lower soil P levels over time to mitigate environmental risks. This approach targets reducing or eliminating P fertilizer applications, allowing crops to gradually deplete soil P reserves and decreasing P losses to surface waters. While maintaining soil P at agronomically sufficient levels is necessary for crop production, excessive soil P can lead to elevated P losses, particularly in regions with a high risk of runoff. The effectiveness of the drawdown strategy depends on initial soil P levels, crop P demand, and hydrological conditions, all of which influence the rate at which soil P declines and its potential impact on runoff P concentrations (Gatiboni et al., 2025).

A study by Liu et al. (2019) in southern Manitoba, Canada, assessed the impact of a long-term drawdown strategy on soil P test levels, P runoff losses, and crop productivity (Table 5). The study was conducted over 18 years, comprising nine years of conventional P fertilization followed by nine years of drawdown, during which crops were fertilized with mineral P fertilizers at reduced or zero application rates. The experiment included two agricultural fields with differing initial soil test P levels (33.8 mg kg^{-1} and 22.3 mg kg^{-1}, Olsen method). This difference in soil P level allowed for an evaluation of how initial P concentrations influenced the effectiveness of the drawdown strategy in reducing P losses.

Findings from this analysis revealed that soil P test decreased by 30% over the drawdown phase, demonstrating that reducing P inputs effectively lowered soil P reserves over time. This reduction in soil P was associated with a 25.5% ±31.4% decrease in total P (TP) losses, with more significant TP reductions observed in the field with a higher initial soil P level (47.7%) than in the field with the lower initial soil P level (3.33%). Although overall dissolved reactive phosphorus (DRP) loads were not reported, the study found a 60.5% ±4.4% reduction in flow-weighted mean DRP concentrations (mg L^{-1}), showing that the drawdown period resulted in less soluble P available in the runoff. These results support the hypothesis that higher initial soil P levels contribute more to runoff P losses and that the effectiveness of long-term drawdown strategies increases with higher initial soil P levels.

These findings demonstrate the importance of identifying soil P levels thresholds above which the risk of P loss to water increases. Fang et al. (2002), using soils representative of major corn-growing areas in the Minnesota River Basin, found that DRP concentrations in runoff increased sharply when Olsen P exceeded 40–55 mg kg^{-1} and Mehlich-III P exceeded 65–85 mg kg^{-1}. Although based on controlled conditions, these values provide useful reference points for environmental risk in high-P soils and demonstrate the need for drawdown strategies when approaching or exceeding such thresholds. However, it is important to point out that these values are higher than the initial soil test P levels in Liu et al. (2019), where significant runoff P losses were still observed. This suggests that field-based P losses may occur at lower STP levels than those predicted in laboratory studies, potentially due in part to P stratification in the upper soil layer under reduced tillage, reinforcing the importance of evaluating site-specific conditions.

The study of Liu et al. (2019) also assessed yield impacts, comparing wheat and canola productivity under conventional and drawdown management. Wheat yield rose by 58.4% and canola by 0.7%, suggesting that P drawdown did not impair, and in some cases improved, crop performance. However, due to limitations in site-specific data, standard deviations could not be estimated.

Overall, these findings indicate drawdown to be a potential strategy as a long-term approach for reducing P losses while maintaining or even increasing crop yield. The greater TP reduction observed in soils with higher initial P levels suggests that targeting high-P fields for drawdown to agronomically optimum soil test P levels could be a particularly effective strategy for mitigating P losses. However, the variability in TP reduction efficiency (SD ±31.4%) indicates that there are site-specific factors influencing the success of this approach. Given the significant reduction in flow-weighted DRP concentrations, drawdown strategies can also effectively reduce the risk of eutrophication in P-sensitive watersheds. Further research is needed to better understand how quickly and effectively soil P levels can be reduced, especially in fields with a history of manure-based P applications.

Table 5. Drawdown study reporting annual total and dissolved reactive phosphorus runoff losses.

Study	Site-years	Location	Loss pathway assessed
Liu et al. (2019)*	18	Manitoba, Canada	Surface runoff

* Complete reference information is found in the reference section at the end of the report

2- Phosphorus Source Strategy: Organic vs. Inorganic Fertilizers

The source of phosphorus (P) used in agricultural systems, whether organic (e.g., animal manures) or inorganic (e.g., triple superphosphate, monoammonium phosphate), can significantly influence the magnitude and form of P losses to water. These effects depend not only on the chemical composition and solubility of the P source but also on its interaction with soil characteristics and hydrological pathways.

To evaluate the effect of P source on P loss, results from four studies comparing organic and inorganic P sources, considering surface runoff and tile drainage pathways (Table 6). The timing of P applications varied among the studies: in Tomer et al. (2016), Wang et al. (2018), and King et al. (2022), both P sources were applied within the same season, Fall, Spring, and Fall, respectively, allowing direct seasonal comparisons. In contrast, King et al. (2018) aggregated data from multiple site-years regardless of application timing; for instance, each treatment (i.e., organic vs inorganic) included both fall and spring applications. Overall, manure-based P applications were associated with higher total phosphorus (TP) and dissolved reactive phosphorus (DRP) losses, increasing TP losses by 50.5% ±34.4% and DRP losses by 53.6% ± 33.2% when compared to mineral P sources. These estimates were weighted by site-years to ensure proportional representation, with King et al. (2022) alone contributing 33 site-years, giving substantial weight to the outcomes.

It is also important to note that while the current analysis found that manure increased overall P losses relative to inorganic fertilizers, this contrasts with findings from the Iowa P reduction strategy report (ISUST, 2025), which reported an average of 46% reductions in runoff TP from liquid swine, dairy, and poultry manure compared to commercial fertilizers. However, those values were based primarily on rainfall simulation studies conducted shortly after manure application and exhibited large variability (e.g., ± 45% and 96% SE). Here, the present analysis is based on year-round natural runoff data from field studies that captured seasonal dynamics and legacy P effects. This methodological difference likely explains the contrasting outcomes and highlights how the choice of hydrologic assessment influences observed P loss responses to different P sources.

Results also diverged depending on the pathway through which P loss was assessed. The studies by Tomer et al. (2016) and King et al. (2018), which evaluated P losses in surface runoff, found that manure significantly increased TP losses by 71.4% and 61.7% with manure, respectively. DRP losses were also elevated in King et al. (2018), with a 68.5% increase under manure treatments. Notably, these two studies reported that initial soil P test levels were higher in manured areas than in areas receiving mineral fertilizers, likely contributing to the elevated P losses observed in runoff.

In contrast, Wang et al. (2018) and King et al. (2022) assessed P losses from tile-drained water and found the opposite effect: manure reduced TP losses by 33.9% and 37.8%, respectively. DRP losses were also lower under manure in both studies, with reductions of 7% and 41.3%, respectively. Importantly, these two studies reported no significant differences in initial soil P test levels between treatments, suggesting that the observed reductions may be more directly attributable to P source differences rather than legacy P effects.

None of the studies reported grain yield differences. However, King et al. (2022) observed that corn silage yield was 15.6% higher under manure (41.4 Mg ha^{-1}) than under mineral fertilizer (35.8 Mg ha^{-1}). Although silage yield is not a direct measure of grain productivity, it does suggest potential agronomic benefits of manure over mineral P supply alone for grain production.

These findings highlight the complexity of P source effects on environmental outcomes. While manure may increase P loss risk in surface runoff, especially under high initial soil P conditions, it may reduce losses in subsurface tile-drained systems, demonstrating the need to consider P source and dominant hydrological pathway in P management decisions.

Table 6. Phosphorus source studies reporting annual total and dissolved reactive phosphorus losses in runoff and tile-drainage.

Study	Site-years	Location	Loss pathway assessed
King et al. (2018)*	33	Ohio	Tile-drainage
King et al. (2022)	2	Ohio	Tile-drainage
Tomer et al. (2016)	11	Iowa	Surface runoff
Wang et al. (2018)	5	Ontario, Canada	Surface runoff and tile-drainage

* Complete reference information is found in the reference section at the end of the report

3- Phosphorus Timing Strategy: Fall vs. Spring Application

The timing of phosphorus (P) fertilizer application is a critical management factor influencing the risk of P losses to surface waters. In temperate cropping systems, P is often applied in the fall for operational convenience. However, fall-applied P, especially when left on the soil surface, is vulnerable to losses via snowmelt and early spring rainfall. In contrast, spring application has the potential to synchronize nutrient availability with crop uptake better and reduce the potential for runoff losses.

Three studies were evaluated to assess the impact of P application timing on runoff P losses (Table 7). In Abel (2017), P was applied at a uniform rate of 36 kg P ha^{-1}, with fall-broadcast diammonium phosphate (DAP) compared to spring-injected ammonium polyphosphate (APP). Over one growing season, spring application reduced TP losses by 23.7% and DRP losses by 75.9%, demonstrating the potential water quality benefits of timely P application in the spring.

Carver et al. (2022) similarly evaluated the effect of P timing on runoff P losses under a corn–soybean rotation. The study compared spring-injected APP with fall-broadcast DAP, both applied at 26.6 kg P ha^{-1} and under the same initial soil P level (17 mg P kg^{-1}, Mehlich-III method). Across 4 site-years, spring application resulted in a 15.6% reduction in TP and a 34.1% reduction in DRP losses. Although placement was confounded with timing, so the reduction is likely due to stacked practices.

To strengthen the analysis, these results were integrated with findings from Liu et al. (2017), who conducted an 11-year field-scale study in Pennsylvania, evaluating the seasonal timing of manure applications on runoff P losses. Although the study was conducted outside the U.S. Midwest, it was included due to its long-term design, use of a corn-soybean rotation, and treatment structure closely aligned with Abel (2017) and Carver et al. (2022). Pennsylvania and the Midwest share a humid continental climate, similar cropping systems, and comparable seasonal runoff patterns, despite some differences in topography and winter severity.

Liu et al. (2017) reported that spring-applied manure reduced total phosphorus (TP) losses by 12 - 16% and dissolved reactive phosphorus (DRP) losses by 19 - 40% compared to fall or winter applications. Because there was no significant difference between the fall and winter treatments, the median of those two application times was used to represent fall application in the comparison with spring: a 14.0% reduction for TP and a 29.5% reduction for DRP. Although the study also included watershed-scale simulations, only the field-scale measured runoff data were used in this analysis.

The magnitude of DRP reduction varied among studies, likely due to differences in P source and placement method. Abel (2017) and Carver et al. (2022) used mineral fertilizers with spring subsurface injection, while Liu et al. (2017) used surface-applied manure, which poses a greater risk for DRP loss and may have reduced the magnitude of observed reductions.

When results from all three studies were combined and weighted by site-years, spring application reduced TP runoff losses by 18.4% ± 2.02% and DRP losses by 33.6% ± 11.1% compared to fall application. None of the studies reported yield response by application timing, leaving agronomic performance unassessed in this context.

These findings reinforce the importance of P application timing in minimizing environmental losses. While mineral and manure sources showed reductions in runoff P with spring application, the degree of reduction differed based on source and method. Nonetheless, the consistent trend across sources suggests spring application as a best management practice (BMP) for reducing both TP and DRP runoff losses in corn-soybean systems.

Table 7. Phosphorus application timing studies reporting annual total and dissolved reactive phosphorus runoff losses.

Study	Site-years	Location	Loss pathway assessed
Abel (2017)*	1	Kansas	Surface runoff
Carver et al. (2022)	11	Kansas	Surface runoff
Liu et al. (2017)	4	Pennsylvania	Surface runoff

* Complete reference information is found in the reference section at the end of the report

4- Phosphorus Placement Strategy: Subsurface vs. Surface Application

The placement of phosphorus (P) fertilizer in agricultural systems plays a critical role in determining its environmental fate. Surface-applied P is more susceptible to loss via surface runoff, particularly in no-till or residue-covered systems. In contrast, subsurface placement, through injection or incorporation, can reduce this risk by positioning P below the soil surface, closer to plant roots and less exposed to rainfall or snowmelt events (Wiens et al., 2019). However, the effectiveness of subsurface placement may vary depending on hydrologic pathways.

To evaluate the impact of P placement on P losses, results from Kimmell et al. (2001) were used (Table 8). The study assessed surface runoff P losses during sorghum production over two years, comparing broadcast (surface) versus injected (subsurface) P application across three tillage systems. Since both placement treatments were evaluated under each tillage system across two years, the total number of site-years was estimated at six. Subsurface injection reduced total phosphorus (TP) losses by 40.2% ± 18.9% and dissolved reactive phosphorus (DRP) losses by 58.8% ± 28.8%, indicating that subsurface placement effectively reduces both forms of P loss under surface runoff conditions. The study did not report initial soil test P values but noted that treatment plots were uniformly managed. Additionally, P loss data from soybean production were excluded from the estimation, as no P was applied to those plots.

These results support the understanding that subsurface P placement is an effective strategy to reduce TP and DRP runoff losses, particularly in systems prone to overland flow. By minimizing P exposure at the surface, this method helps reduce both particulate and dissolved P transport. However, further research across varied systems is needed to validate these findings and assess potential agronomic trade-offs.

Table 8. Phosphorus application placement studies reporting annual total and dissolved reactive phosphorus losses in runoff and tile drainage.

Study	Site-years	Location	Loss pathway assessed
Kimmel et al. (2001)*	6	Kansas	Surface runoff

* Complete reference information is found in the reference section at the end of the report

5- Tillage Intensity Strategy: Impact on Phosphorus Losses

Tillage intensity is critical in phosphorus (P) transport by influencing surface residue cover, soil structure, infiltration, and erosion processes. Conservation tillage systems that aim to retain crop residue, such as no-till, are promoted to reduce erosion and surface runoff.

To evaluate the effect of tillage intensity on P losses, data were evaluated from three field studies: Ginting et al.(1998), Shipitalo et al. (2013), and Kimmell et al. (2001) (Table 9). In Ginting et al. (1998), ridge-till system was compared to moldboard plowing on corn production, under the same soil P level (12 mg P kg^{-1}, Olsen method), with ridge-till retaining three times more crop residue. In Shipitalo et al. (2013), no-till and chisel plow systems were evaluated across 32 site-years on corn and soybean production, with baseline soil P levels of ~30 mg P kg^{-1} (Bray P1 method). Yield data from this study indicated a 5.81% reduction in corn yield and 8.33% reduction in soybean yield under no-till, which aligns with global evidence of modest yield declines under no-till when not combined with additional conservation practices (Pittelkow et al., 2015). Kimmell et al. (2001) evaluated no-till versus conventional till (either as chisel plow or ridge-till) under sorghum and soybean production across 2 site-years. All three studies assessed P losses exclusively through surface runoff.

For Shipitalo et al. (2013) and Kimmell et al. (2001), results were averaged across crops to provide an overall estimate. When combined, the three studies revealed that conservation tillage systems resulted in a 7.74 ± 9.63% reduction in TP losses but an increase of 78.8 ± 83.4% in DRP losses. This trade-off reflects reduced erosion but greater accumulation of soil P in the surface layer under conservation tillage.

When the data were separated by specific tillage comparisons, more specific trends were identified. Ridge-till compared to moldboard plowing (Ginting et al., 1998) reduced TP losses by 47% ± 3.63% and increased DRP losses by 180% ± 226%. In comparisons of no-till to conventional tillage systems (Kimmell et al., 2001; Shipitalo et al., 2013), no-till reduced TP losses by 5.40% ± 0.18% but increased DRP losses by 72.88% ± 82.0%.

Two additional long-term studies from Canada (Liu et al., 2019; Tiessen et al., 2010) were reviewed but not included in the synthesis. In those studies, conservation tillage systems had higher soil test P levels than conventional systems, making it difficult to isolate the effect of tillage intensity alone. These elevated soil P levels likely contributed to the increased TP losses observed under conservation tillage.

These findings indicate that reducing tillage intensity tends to lower TP losses by limiting erosion but increasing DRP losses, likely due to surface P stratification under residue-rich conditions. This trade-off shows the importance of integrating complementary practices, such as cover crops, to maintain soil conservation benefits while mitigating DRP losses.

Table 9. Tillage intensity studies reporting annual total and dissolved reactive phosphorus runoff losses.

Study	Site-years	Location	Loss pathway assessed
Ginting et al. (1998)*	2	Minnesota	Surface runoff
Kimmel et al. (2001)	2	Kansas	Surface runoff
Shipitalo et al. (2013)	32	Ohio	Surface runoff

* Complete reference information is found in the reference section at the end of the report

6- Cover Crop Strategy: Impact on Phosphorus Losses

Cover crops are widely promoted as a conservation practice to improve soil health, reduce erosion, enhance nutrient cycling, and mitigate nutrient losses to surface and subsurface waters. Their effectiveness in reducing phosphorus (P) losses depends on several factors, including cover crop species, timing, biomass production, and termination method.

To assess the effectiveness of cover crops in reducing P losses, eight studies were initially identified, including six peer-reviewed papers (Kaur et al., 2024a; Adler et al., 2020; Hanrahan et al., 2021; Carver et al., 2022; Zhang et al. 2017; Zhu et al., 1989; Kaur et al., 2024b) and one thesis (Abel, 2017) (Table 10). However, the study by Hanrahan et al. (2021) was excluded from the final analysis because the cover crop treatments had substantially higher initial soil P levels than the no-cover treatments (135 vs. 58 mg P kg^{-1}, Mehlich-III), confounding the attribution of P loss differences to cover crop effects alone.

The included studies involved mineral P sources, and most assessed P losses from surface runoff, except for Adler et al. (2020), which assessed losses in tile-drained water, and Zhang et al. (2017), who reported losses in surface runoff and tile-drained water. A wide range of cover crop species were included in these studies, such as rye, wheat, radish, turnip, rapeseed, hairy vetch, triticale, Canada bluegrass, common chickweed, and downy brome, reflecting diverse management approaches. Not all studies evaluated both

P forms: Adler et al. (2020) reported only TP losses, while Zhu et al. (1989) reported only DRP losses.

Reduction efficiency was estimated for each study and combined using a site-year weighted mean approach. For Zhang et al. (2017), P losses from surface runoff and tile-drained were summed to represent total edge-of-field losses. The analysis showed that cover crops reduced TP losses by 21.5% ± 16.8% and DRP losses by 7.93% ±31.6% across hydrologic pathways. While TP reductions were more consistent across studies, the effect on DRP was more variable, likely due to differences in soil and residue management, surface stratification of P, and biomass decomposition patterns.

Two studies (Adler et al., 2020 and Kaur et al., 2024b) also evaluated the impact of cover crops on corn grain yield, and results indicated an average 10.7% ± 9.73% reduction with cover crops. The yield data from Kaur et al. (2024b) were extracted from Kaur et al. (2023), which reported agronomic outcomes, not environmental losses. These yield declines may be linked to early-season soil moisture depletion or nitrogen immobilization caused by high-residue cover crops (late termination date), particularly under cool or wet spring conditions (Deines et al. 2022).

Together, these findings suggest that cover crops can be an effective strategy to reduce total P losses by protecting the soil surface and reducing erosion. However, their effect on DRP is more variable and may require additional management considerations, such as appropriate termination timing and nutrient balancing, to prevent trade-offs. Additionally, potential yield impacts should be considered in system-level decisions to optimize both environmental and agronomic outcomes.

Table 10. Cover crop studies reporting annual total and dissolved reactive phosphorus losses in runoff and tile drainage.

Study	Site-years	Location	Loss pathway assessed
Abel (2017)*	1	Kansas	Surface runoff
Adler et al. (2020)	3	Missouri	Tile drainage
Carver et al. (2022)	4	Kansas	Surface runoff
Kaur et al. (2024a)	4	Missouri	Surface runoff
Kaur et al. (2024b)	3	Missouri	Surface runoff
Zhang et al. (2017)	4	Ontario, Canada	Surface runoff and tile drainage
Zhu et al. (1989)	3	Missouri	Surface runoff

* Complete reference information is found in the reference section at the end of the report

7- Constructed Wetlands: Phosphorus Reduction

Constructed wetlands have also been evaluated as a strategy to mitigate phosphorus (P) losses from agricultural landscapes in the Midwest. The goal of this assessment was to quantify total phosphorus (TP) and dissolved reactive phosphorus (DRP) reduction efficiencies under regional conditions. The inclusion criteria were (1) studies conducted in Minnesota or the broader U.S. Midwest, (2) year-round monitoring of P losses, and (3) availability of catchment-to-wetland area ratios.

Two studies from Minnesota were initially considered. Lenhart et al. (2016) reported a 76% reduction in TP from tile drainage; however, subsequent review indicated that much of the apparent reduction was due to seepage into groundwater through a sand vein, rather than true wetland treatment performance. For this reason, the study was excluded from the overall efficiency calculation. Gordon et al. (2021) monitored TP year-round but concluded that TP removal was inconclusive. Both studies had catchment-to-wetland ratios of 1.1%, which are lower than many studies conducted across the region, potentially limiting their treatment capacity.

Given the limited availability of suitable Minnesota data, the estimation relied on studies from Illinois (Table 11), which provide the most comprehensive long-term records of wetland performance. Although studies from across the U.S. Midwest were considered, only those from Illinois met the criteria. Kovacic et al. (2000) included 9 wetland-years of tile drainage catchments with ratios ranging from 3.2% to 6.0%. Hoagland et al. (2001) reported 1 wetland-year with a 6% ratio. Kovacic et al. (2006) reported 4 wetland-years combining both tile drainage and surface runoff, with ratios ranging from 3.3% to 4.3%. Finally, Lemke et al. (2022) reported 36 wetland-years from tile drainage systems, with ratios of 3% to 9%. These higher ratios, compared with Minnesota studies, may partly explain the more consistent P retention reported in Illinois, as wetlands with a larger treatment area relative to their contributing catchments generally have greater capacity to retain nutrients. At the same time, caution is necessary when applying IL results to MN due to key regional differences. Minnesota typically experiences longer frozen periods and different hydrologic dynamics associated with snowmelt, as well as generally lower average rainfall than central Illinois. These climatic contrasts can alter both water flow and nutrient transport and consequently influence wetland treatment performance.

The reduction efficiencies were estimated by comparing inflow and outflow P loads. For each study, the mean reduction of TP and DRP was extracted, and variation across studies was expressed as standard deviation (SD). To account for study size, both means and SDs were weighted by the number of wetland-years, giving greater influence to longer-term or multi-wetland datasets. Across the compiled studies, constructed wetlands reduced TP losses by 26.1 ± 19.3% and DRP losses by 61.8 ±13.5%. These values highlight two important insights. First, wetlands were moderately effective at retaining TP, largely through sedimentation and particulate trapping. Second, they were substantially more effective at reducing DRP, suggesting that wetland processes such as sorption and plant uptake may play a particularly important role in DRP removal.

However, constructed wetlands can also exhibit negative or inconsistent TP performance over time. Phosphorus can accumulate in wetland sediments, and once these sediments become saturated, internal P release may occur under reducing conditions as iron-bound P is solubilized. Additionally, the accumulation and decomposition of organic matter can increase dissolved P concentrations, especially following drying-rewetting cycles or during low-oxygen periods. Without active management, such as sediment removal or vegetation harvest, constructed wetlands may gradually become a net P source. Particularly, in mature systems. These processes introduce uncertainty regarding the long-term stability of TP retention across years and sites.

Given these uncertainties, and consistent with Illinois and Iowa nutrient reduction strategies (IDALS, 2025; ILEPA, 2023), which both chose not to assign a P reduction efficiency for constructed wetlands due to high variability and limited evidence, we also do not provide a final efficiency value here. While the weighted mean estimate (26 ± 19.3%) offers useful context, it should not be interpreted as a recommended statewide value. Additional Minnesota-based research will be critical to determine whether constructed wetlands consistently reduce TP losses under local conditions and to quantify long-term retention potential.

Table 11. Constructed wetland studies reporting annual total and dissolved reactive phosphorus losses in runoff and tile drainage.

Study	Wetland-years	Location	Loss pathway assessed	Catchment to Wetland ratio (%)
Kovacic et al. (2000)*	9	Illinois	Tile drainage	3.2 to 6.0
Hoagland et al. (2001)	1	Illinois	Tile drainage	6.0
Kovacic et al. (2006)	4	Illinois	Tile + surface runoff	3.3 to 4.3
Lemke et al. (2022)	36	Illinois	Tile drainage	3.0 to 9.0

* Complete reference information is found in the reference section at the end of the report

8- Conditional opportunity for reducing P losses:

Soil P Build-Up Using Manure on Low-P Soils

Phosphorus (P) management plays a critical role in supporting long-term soil fertility and crop productivity. In Minnesota and other areas with low soil test P (STP), building up soil P reserves is often necessary to meet crop P requirements for future growing seasons. One strategy to achieve this involves repeated applications of manure on soils classified as low to medium STP levels. While this practice is primarily agronomic, aimed at elevating soil P to agronomic optimum levels, it may also offer an opportunity for reductions in total P (TP) losses, specifically during the build-up phase.

Two Minnesota-based studies (Ginting et al., 1998; Gessel et al., 2004; Table 12) evaluated manure applications on soils with initial STP levels ranging from 6 to 12 mg kg^{-1} (Olsen method), values within the low to medium fertility range according to state guidelines (Kaiser et al., 2023). These studies did not apply mineral P and compared treatments with increasing manure rates, including over-recommended rates, to evaluate runoff P losses and crop yield. In both experiments, manure applications were designed to increase soil P legacy, with higher manure-based P rates considered the best P management practices (BMPs) within this objective.

A site-year weighted mean approach was used to synthesize results. Higher manure-based P rates were associated with a 21.9% ±14.7% reduction in TP runoff losses compared to lower rates, while DRP losses increased by 175% ±129%. TP losses ranged from 0.55 to 2.6 kg ha^{-1}, and DRP losses from 0.07 to 0.39 kg ha^{-1}. Corn grain yield increased modestly by 6.7% ±0.8% under higher manure applications.

The reduction in TP losses under higher manure rates appears linked to enhanced infiltration and lower surface runoff, as observed particularly in Gessel et al. (2004). Additionally, increased residue cover and improved soil aggregation likely reduced erosion and particulate P transport (Gilley & Risse, 2000). At the same time, elevated STP and surface accumulation of manure P increased DRP losses, likely due to P saturation and solubilization effects (Duncan et al., 2017).

Importantly, these environmental effects, particularly the reduction in TP, are expected to occur only during the build-up phase, when soils transition from low to optimum P levels. Once optimum STP is reached and manure applications are reduced or ceased, the potential for TP reduction may diminish, while DRP risks may persist if surface P accumulation remains.

The modest yield increase observed suggests that corn may still respond to manure-based P applications above recommended rates; however, these gains are likely not economically justified and may contribute to P losses once soil P reaches agronomic sufficiency. As noted by the Mulla and Whetter (2020) report, which focuses on the Red River Basin region, yield benefits tend to plateau around 10 mg kg^{-1} Olsen P, and increasing STP beyond this threshold does not improve yield but significantly raises the risk of P losses. Further research is needed to evaluate the long-term agronomic and environmental implications of manure-based P build-up strategies.

In summary, building soil P using manure in low-STP soils may offer temporary water quality benefits by reducing TP runoff losses, though it also increases DRP and may negatively affect yields. This strategy should be used specifically in regions with low STP, with clear exit points once optimum levels are achieved, and with attention to DRP management.

Table 12. Phosphorus build-up studies reporting annual total and dissolved reactive phosphorus losses in runoff and tile drainage.

Study	Site-years	Location	Loss pathway assessed
Gessel et al. (2004)	3	Minnesota	Surface runoff
Ginting et al. (1998)	2	Minnesota	Surface runoff

* Complete reference information is found in the reference section at the end of the report

Chapter 5 References

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