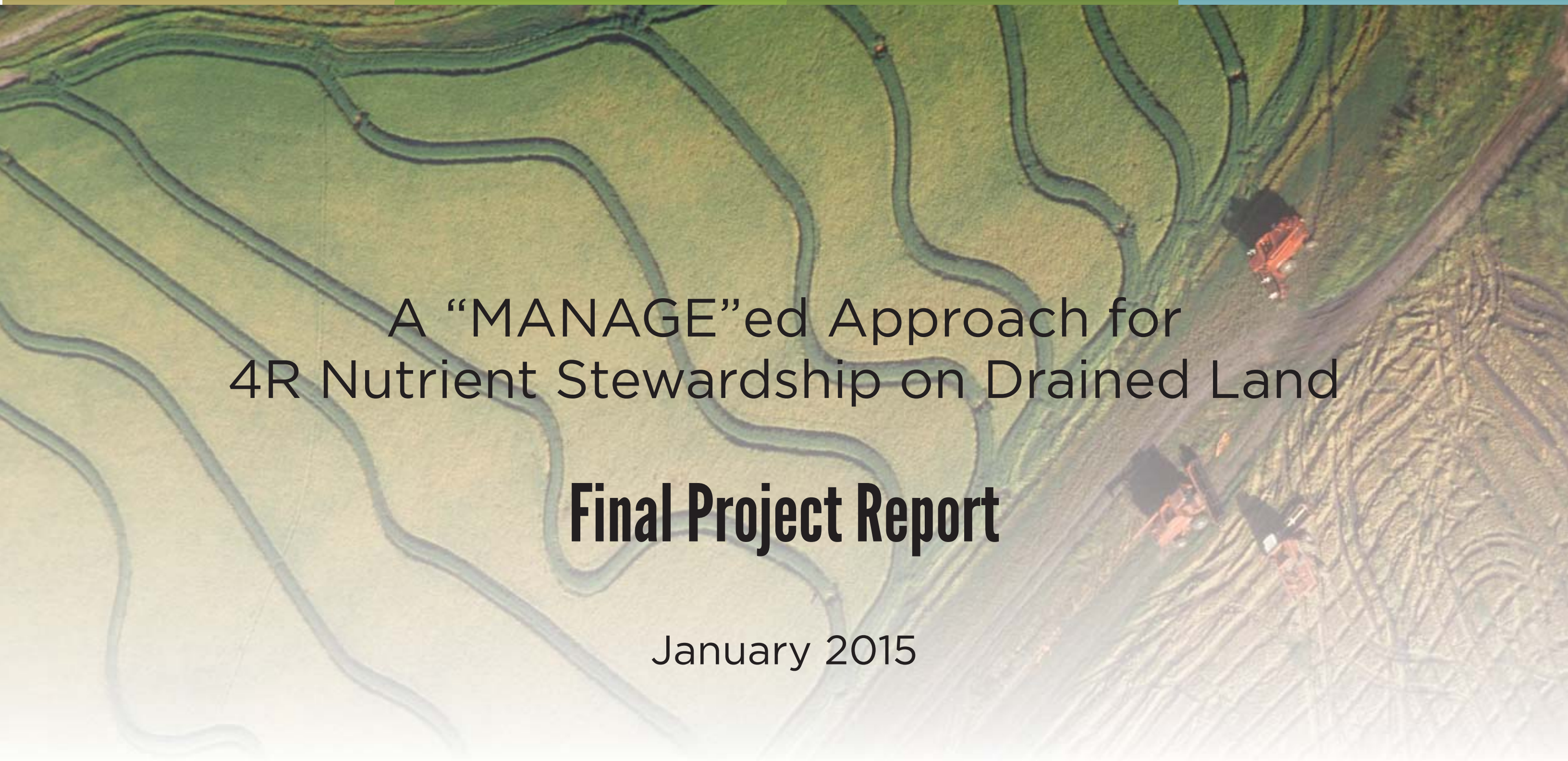

A "MANAGE"ed Approach for 4R Nutrient Stewardship on Drained Land

Final Project Report

January 2015

4R Research Fund

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THE CONSERVATION FUND

USDA United States Department of Agriculture Agricultural Research Service

A "MANAGE"ed approach for 4R Nutrient Stewardship on Drained Land

Laura E. Christianson, Ph.D., The Conservation Fund
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Executive Summary

As agriculture in the 21st century is faced with increasing pressure to reduce negative environmental impacts while continuing to efficiently produce food, fiber, and fuel, it becomes ever more important to reflect upon more than half a century of drainage water quality research to identify future paths towards increased sustainability. This work provided a quantitative review of the water quality and crop yield impacts of artificially drained agronomic systems across North America by compiling data from drainage nutrient studies into the "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database. Of the nearly 400 studies reviewed, 91 individual journal publications and 1279 site-years were included in the new MANAGE Drain Load table with data from 1961 to 2012.

The major outputs of this work included three manuscripts to be submitted for peer-review:

1. The MANAGE Drain Load database: Review and compilation of more than fifty years of drainage nutrient studies
2. 4Rs water quality impacts: A review and synthesis of forty years of drainage nitrogen losses
3. A quantitative review and synthesis of fifty years of drainage phosphorus losses

Journal submission is planned for February/March 2015. The new Drain Load table will be made publically available in the online version of MANAGE once the draft manuscripts are accepted for publication.

Major findings included that wetter years resulted in significantly greater drainage discharge and nutrient loads than dry years, and there was a notable seasonal impact within years (i.e., non-growing season nutrient transport poses a particular concern). The importance of drainage hydrology was a strong theme that emerged throughout the analyses which carries a special significance in the face of a highly variable climate. Major findings pertaining to the 4Rs strategies included:

Nutrient Application Rate

- Increasing nitrogen application rates both improved crop yields and increased dissolved nitrogen loads in drainage. As such, optimizing nitrogen rates will continue to receive primary research and regulatory focus. "Fine-tuning" these rates is clearly important from economic and environmental standpoints, but it would be short-sighted and unrealistic to focus solely on this practice.
- The order of magnitude difference between agronomic phosphorus application rates and phosphorus loadings that can cause ecological damage presents a serious environmental challenge, especially compared to nitrogen. Across the literature, generally less than 2% of applied phosphorus was lost in drainage in a given site year.

Nutrient Application Timing and Method

- The lack of significant differences between nitrogen application timing or application method treatments indicated neither should receive primary focus as dissolved nitrogen load reduction strategies. Nevertheless, the typically recommended practices such as applying at-planting or side-dressing had lowest median nitrogen losses (not significant), and injection of nitrogen sources, the most widely used nitrogen application method across studies, increased nitrogen loss risk in drainage over incorporation and surface application based on median values (not significant).
- The timing and method of phosphorus application are both known to be important for drainage phosphorus losses, but these conclusions could not be verified due to low site-year counts across the Drain Load database.

Nutrient Source

- Use of organic nitrogen and phosphorus sources could boost corn yields with potentially no increase in dissolved nutrient loads compared to inorganic fertilizer. Nevertheless, adherence to 4Rs strategies is vital regardless of the nutrient source, and accurate implementation of the 4Rs approach will require site-specific knowledge.

As drainage water quality research continues into the 21st century, this work provided an important opportunity to evaluate gaps in drainage nutrient research including:

- The scientific body of knowledge would benefit from more annual nutrient loading data in newly drained areas, ditch drained areas, and areas where surface intakes are specified.
- Historically, dissolved nitrogen loads in subsurface drainage have received much greater attention than phosphorus loads, and this presented challenges in terms of low site-year counts for phosphorus losses in this meta-analysis.
- There is a need to consider all seasons for quantification of annual drainage nutrient loads. It is increasingly clear that snowmelt drainage nutrient loads are significant in certain locations.
- Overall, more long-term drainage nutrient loss studies with coordinated controls across multiple sites and years would increase statistical power for more robust future comparisons.

Paths Forward

- MANAGE currently includes both load and concentration tables for runoff from agricultural and forested land, and the addition of a Drain Concentration table to supplement the new Drain Load table is suggested. This may be especially enlightening for better understanding agronomic management impacts on drainage phosphorus transport.
- A number of European drainage studies contributed to the literature review, but were not included in the Drain Load table due to MANAGE's existing North American focus. Expansion of the database to include European studies would be an exciting new frontier.
- An Urban Load and Concentration database should be incorporated into MANAGE to fully capture nutrient and sediment contributions across sectors.
- New drainage nutrient load studies, particularly those suggested in the "Gaps" section above, should continue to be added to MANAGE on a periodic basis.

Thank you for your thoughtful and generous funding of this research. We look forward to future opportunities to continue this important conversation relating the 4Rs and water quality.

A "MANAGE"ed approach for 4R Nutrient Stewardship on Drained Land

Laura E. Christianson, Ph.D., The Conservation Fund
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Interpretive Summary

As agriculture in the 21st century is faced with increasing pressure to reduce negative environmental impacts while continuing to efficiently produce food, fiber, and fuel, it becomes ever more important to reflect upon more than half a century of drainage water quality research to identify future paths towards increased sustainability. Here, nearly 400 drainage water quality studies were reviewed to create a publically available database that compiles site, climate, cropping, and agronomic management information across more than fifty years of drainage research. This new "Drain Load" table in the "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database was then used to compare drainage water quality and crop yield based on this pooled dataset. Unsurprisingly, wetter years resulted in more drainage and greater nitrogen and phosphorus loads in drainage waters. Nutrient loads also increased at increasing nitrogen and phosphorus application rates. However, increased application rates also improved crop yield across this large dataset which indicates that new approaches and trade-offs may be required to balance agronomic and environmental goals. Optimal nutrient application timing and placement are generally known to be important strategies for improving water quality, but these two approaches showed no significant impact here. This may have been complicated by a lack of drainage phosphorus studies. This work provided an important opportunity to evaluate gaps in drainage nutrient research. While more drainage studies focused on newly drained areas, ditch drained areas, and phosphorus in drainage water are needed, the most important call is for more long-term studies with coordinated controls across multiple sites and years. The MANAGE database primarily focuses on North America, but suggested future directions include addition of international drainage studies as well as creation of a Drain Concentration table to complement this new Drain Load table.

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Final Report Update Addendum
02 October 2015

Since final report submission in January 2015, the first chapter has been published as:

Christianson, L.E., Harmel R.D. (2015) The MANAGE Drain Load database: Review and compilation of more than fifty years of drainage nutrient studies. Ag. Wat. Man. 159:277-289. Available open access at: http://www.sciencedirect.com/science/article/pii/S0378377415300408.

Since final report submission in January 2015, the second chapter has been published as:

Christianson L.E., Harmel R.D. (2015 (doi:10.2134/jeq2015.03.0170; posted 23 Sept. 2015)) 4R water quality impacts: A review and synthesis of forty years of drainage nitrogen losses. J. Envir. Qual. Available as a First Look at: https://dl.sciencesocieties.org/publications/jeq/view/first-look/q15-03-0170.pdf.

Significant improvements have been made to the third chapter since January 2015, and it will be submitted for peer review by the end of 2015 as:

Christianson, L.E., R.D. Harmel, D. Smith, M.R. Williams, and K. King. (Under review). A quantitative review and synthesis of fifty years of drainage phosphorus losses.

The authors thank readers for their interest and encourage them to view these peer-reviewed journal manuscripts for the final version of project results.

The MANAGE Drain Load database: Review and compilation of more than fifty years of drainage nutrient studies

L.E. Christianson and R.D. Harmel

Abstract

As agriculture in the 21st century is faced with increasing pressure to reduce negative environmental impacts while continuing to efficiently produce food, fiber, and fuel, it becomes ever more important to reflect upon more than half a century of drainage water quality research to identify paths forward. This work provided a quantitative review of the water quality and crop yield impacts of artificially drained agronomic systems across North America by compiling data from drainage nutrient studies in the "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database. Of the nearly 400 studies reviewed, 91 individual journal publications and 1279 site-years were included in the new MANAGE Drain Load table with data from 1961 to 2012. Across site-years, the mean and median percent of precipitation occurring as drainage were 25 and 20%, respectively, with wet years resulting in significantly greater drainage discharge and nutrient loads. Water quality and crop yield impacts due to management factors such as cropping system, tillage, and drainage design were invested. This work provided an important opportunity to evaluate gaps in drainage nutrient research; In addition to the current analyses, the resulting MANAGE drainage database will facilitate further analyses and improved understanding of the agronomic and environmental impacts of artificial drainage.

Introduction

A strong "development ethos" in North America first demonstrated through early settlement water development projects set the context for the 21st century's widespread use of artificial agricultural drainage (Skaggs and van Schilfgaarde, 1999). Drainage legislations and works in the United States are reported as early as the mid-1700s even prior to the signing of the Declaration of Independence (Shirmohammadi et al., 1995). The first documented use of tile drainage occurred in 1835 by a New York farmer who imported "horseshoe-type drain tile" patterns from Scotland (Ritter et al., 1995). Since these early beginnings, artificial agricultural drainage has been a source of scientific study and policy debate (Madramootoo et al., 2007) and will likely continue to be so for years to come.

The economic benefit of improved artificial drainage is ultimately the most important driver for such installations. Improved drainage enhances crop growth and yield (Portch et al., 1968; Stout and Schnabel, 1994; Tan et al., 1993) and also reduces on-farm risk by increasing the number of days available for field activities (Fausey et al., 1995; Skaggs and van Schilfgaarde, 1999). Installation of subsurface drainage systems costs from $300-$600 per acre but can boost yields by 5-25% (Blann et al., 2009). Pavelis (1987) reported the average US replacement value of drainage was $370 per acre, and the total net value of drainage capital in the US was nearly $25 billion in 1985 dollars. With high crop and land prices in the late 2000's (Nickerson et al., 2012), the value of said infrastructure is doubtless much higher.

Despite the agronomic and rural economy benefits associated with artificial drainage, it does change natural hydrology and is a major conduit for nutrient transport (David et al., 2010). Foundational work by Randall and Goss (2008) identified controllable and uncontrollable factors that impact drainage water quality. Uncontrollable factors include amount and temporal distribution of precipitation, climate during the non-growing season, soil type and organic matter, the latter of which can be influenced by management practices which are controllable. Controllable factors pertain to human-induced choices such as cropping system, tillage practices, and nutrient management (Randall and Mulla, 2001). Drainage system design is also an important controllable factor not only for drainage efficiency but for water quality (Randall and Goss, 2008; Sands et al., 2008; Skaggs et al., 1994).

Nevertheless, artificial subsurface drainage can also benefit water quality. Installation of tile drainage can reduce surface runoff volume and peak outflow rates by providing storage capacity in the soil above the tiles (Ball Coelho et al., 2012a; Blann et al., 2009; Robinson and Rycroft, 1999). In areas where prioritization between pollutants is required, subsurface drainage may indeed be a strategy for reducing surface runoff-associated sediment and phosphorus (P) transport (Ball Coelho et al., 2012b; Bottcher et al., 1981; Fausey et al., 1995; Gold and Loudon, 1989). However, additional mitigation strategies for soluble pollutants, particularly nitrate-nitrogen, will be necessary if drainage is implemented to reduce sediment and particulate P loads. Conveniently, installation of subsurface systems allows opportunity to treat some dissolved pollutants through diversion of outflow and water table control *in situ* treatment (Fausey et al., 1995). Through good design, drainage systems can optimize both agronomic and environmental goals (Skaggs et al., 1994).

As policy debate and regulatory interest related to water quality continue to grow, it becomes important to reflect upon decades of drainage research in North America to create a future vision for drained agricultural lands. With increased computing power and more sophisticated hydrologic and biogeophysical modeling efforts, there is clearly a need for the large number of drainage nutrient studies to be compiled and analyzed to enhance understanding of the state of drainage science and to develop improved drainage models. Fortunately, an existing framework is available for such a drainage-oriented compilation. The "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database aims to "compil[e] measured annual nitrogen and phosphorus load data representing field-scale transport from agricultural land uses in the USA into a readily accessible, easily queried format" (Harmel et al., 2008). This free and publically available water quality database was developed in Microsoft Access by the United States Department of Agriculture, Agricultural Research Service, Grassland, Soil, and Water Research Laboratory in Temple, Texas (www.ars.usda.gov/spa/manage-nutrient). The agricultural runoff and forest-focused tables within MANAGE include over 1800 watershed-years from 300 nutrient load records (i.e., sites or plots) with database fields pertaining to study location, tillage type, conservation practice, soil type/group, fertilizer application, nutrient loss, and reference information (Harmel et al., 2006). This well-established database was the ideal platform for this work aimed at integrating and compiling water quality and yield information from drained landscapes in North America.

The major aim of this work was to review and analyze the water quality and crop yield impacts of artificially drained agronomic systems across North America. The specific objectives were to further develop the MANAGE database through addition of drainage studies and to analyze the resulting pooled information to investigate drainage trends and impacts during the past fifty years. This work was a part of broader efforts to evaluate the nutrient loading and economic impacts associated with the 4R nutrient management strategies.

Methods

Literature was reviewed between April and October 2014 for nitrogen (N) and phosphorus (P) drainage loads and crop yields, and site-years deemed acceptable were entered into a new "Drain Load" database table in MANAGE (Microsoft Access). First, surface runoff studies already in MANAGE that also reported drainage water quality were reexamined and entered into the Drain Load table, so identifiers and fields across the entire database were consistent. New sources of drainage nutrient loads were identified through web and journal searches and by tracing citations in relevant papers and review articles. A master literature review database was kept of all potential and sourced articles to aid in tracking: (1) articles yet to be sourced and/or reviewed, (2) suitable articles having undergone review, (3) articles needing further review, and (4) articles having undergone review and deemed unsuitable. Each sourced article was summarized in a supporting appendix to provide transparency on either why the dataset was deemed unsuitable for inclusion in MANAGE or where within the paper the MANAGE-relevant data were sourced. In total, 394 individual publications were reviewed.

Data in MANAGE are based upon a robust, previously peer-reviewed selection process (Harmel et al., 2006; Harmel et al., 2008). Suitable studies met the following criteria: peer-reviewed, from study areas of at least 0.009 ha with a homogenous land use within North America, not a rainfall simulation or lysimeter study, and include load data from at least one year. For the new Drain Load table, irrigation-drainage systems common in the western US and controlled drainage treatments were not included (i.e., only nutrient loads from free, unrestricted outlets were included). The most prevalent unsuitability reasons for the MANAGE Drain Load database were that (1) a given study did not contain an annual nutrient load (e.g., the study reported only nutrient concentrations or hydrology information, the study reported event-based sampling rather than annual values), (2) the study was not, in fact, a drainage study (e.g., a soil leaching or groundwater seepage study using porous cup samplers or lysimeters), (3) the study was from outside North America (Figure 1), or (4) the study was a review with no original data. There were a few notable drainage studies that were necessarily excluded from the Drain Load table. For example, work at the Waseca, MN research station used drainage plots of only 0.0055 ha (6.1 m x 9.1 m), smaller than the 0.009 ha threshold. Nevertheless, these studies and others were used to inform the text-based literature review (Randall and Vetsch, 2005; Randall et al., 2003). In at least 25 studies, it was necessary to extract data from published graphs and figures using Data Thief® software (Johnson and Curtis, 2001; Tonitto et al., 2006). In one case, it was necessary to contact the lead author to clarify the number of site-years and study details (Evans et al., 1995).

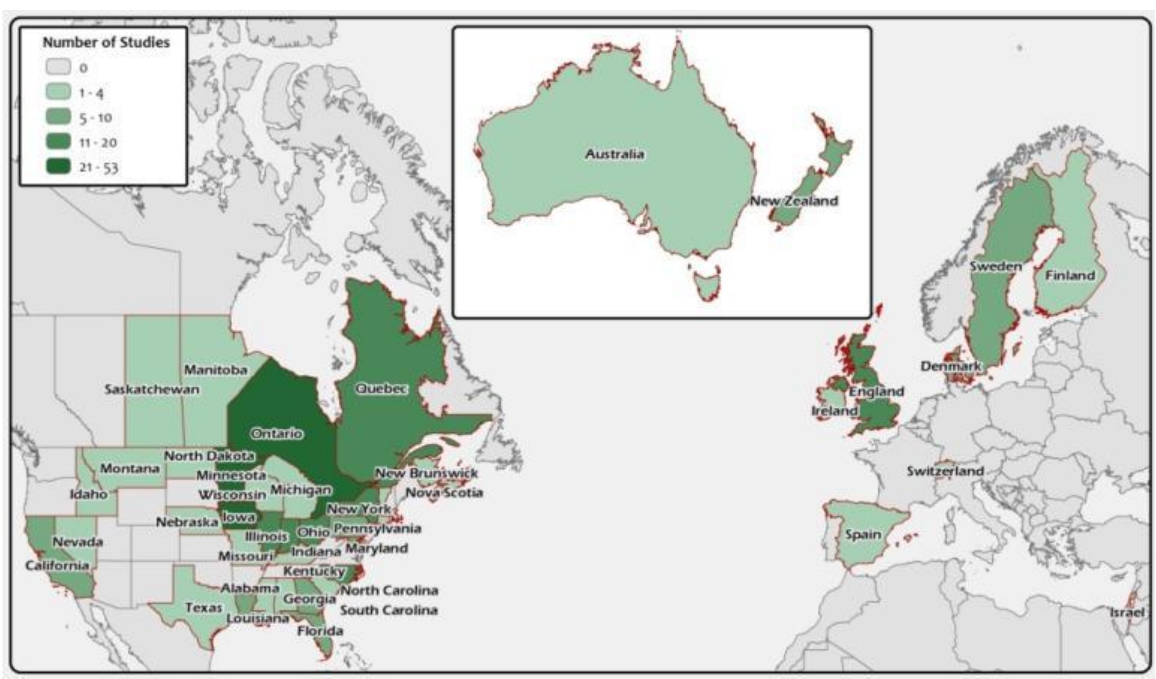

Figure 1: Locations of sourced studies for this review; n = 379 locations reported from the 394 publications reviewed

Data on dissolved, particulate, and total N and P loads were sought, and existing database fields in MANAGE's Ag Load table served as the template for the new Drain Load table. Recently, several new fields were added across all of MANAGE's tables to enhance ability to make 4R-related comparisons. These fields included both N and P crop uptake, fertilizer timing ("At Planting, within 1 week of plant", "Out of Season, > 2 months before plant", "Pre-Plant, 2 months-1 week before plant", and "Side/Top Dress, > 1 week after plant"), yield, and economic profit/loss. The newly created field of crop yield was used as the primary economic metric because few studies reported economic data. Specifically for the Drain Load table, new drainage-related fields included: drain type ("surface", "subsurface (with inlets)", or "subsurface (no inlet specified)"), drain spacing, and drain depth. In the case of surface drainage, drain spacing referred to spacing between ditches and drain depth was the ditch depth. The largest deviation from the existing MANAGE format was that in the new Drain Load table, each record represented an individual site-year, whereas previously, each record represented a site with data pooled. The site-year approach was taken here as it was easier to quantify trends when each record was weighted equally across time.

After development of the MANAGE Drain Load table, data were analyzed with counts (e.g., histograms), box plots, and regression analyses. Due to the large dataset and high nutrient load variability among many variables (e.g., crop type, tillage practices, precipitation), it was necessary to "bin" similar groups within certain categorical fields. For example, for hydrologic analysis, "wet" and "dry" years were separated. The approximate mean (846±219 mm) and median (828 mm) across all precipitation values in the Drain Load table (n= 889) were used as separation points, with precipitation values less than 820 mm or greater than 850 mm considered "dry" or "wet" site-years, respectively. Site-years with precipitation values falling between these separation points (n=28, or 3% of the precipitation-reporting site-years) were excluded to provide a distinct division. The majority of the data were non-normally distributed, thus were analyzed using Kruskal-Wallis one-way Analysis of Variance tests based on rank which uses median values. Mann-Whitney sum t-tests based on rank and median values were used when comparing only two treatments (Sigma Plot 12.5).

Results

Site-years over time and space

A total of 91 individual journal publications and 1279 site-years were included in the MANAGE Drain Load table. While this was based on a comprehensive search, MANAGE is intended to be dynamic with periodic additions (e.g., Harmel et al., 2006; Harmel et al., 2008).

Geography

The majority of nutrient load site-years were from Midwestern states which was unsurprising considering the prevalence of drainage, primarily subsurface drainage, in this region (Table 1; Figure 2; Pavelis, 1987; Sugg, 2007; Zucker and Brown, 1998). The states of Iowa and Illinois alone accounted for 50% of the site-years. Canadian drainage studies were also clearly important with Ontario the second most predominant state/province (255 site-years; Table 1). There were fewer site-years from the eastern Midwest (Indiana, Michigan, and Ohio), despite the strong presence of drainage in this area. There were also few nutrient load studies from the southeast and Mid-Atlantic states, despite the long history and prevalence of drainage in this region (Table 1; Madramootoo et al., 2007; Thomas et al., 1995). Notwithstanding efforts to ensure the comprehensiveness of this review, South Carolina and Delaware had no studies represented, in spite of a high prevalence of drained cropland in each state (both approximately 25%; Pavelis, 1987). Nevertheless, any spatial estimate of drainage is necessarily viewed with some uncertainty as there is no comprehensive assemblage of drainage records kept by any central authority (Blann et al., 2009).

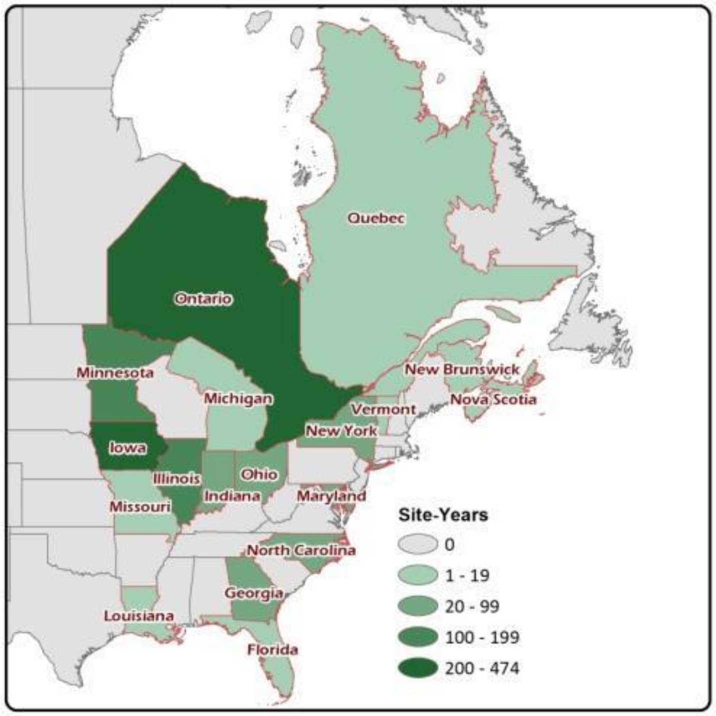

Figure 2: Map of data coverage from MANAGE Drain Load site-years (n=1279)

Table 1: Count by state/province of the MANAGE Drain Load studies and site-years showed with estimates of the percentage of cropland drained

	Percent of cropland drained (%)		Studies Included	Site-Years Included
	Pavelis (1987)	Sugg (2007)	------ Count (Percent of total, %) ------	
Iowa	25	32.4	24 (26%)	474 (37%)
Ontario	--	--	16 (18%)	255 (20%)
Illinois	35	47.8	9 (10%)	168 (13%)
Minnesota	20	14.4	7 (8%)	106 (8%)
North Carolina	25	--	3 (3%)	64 (5%)
Indiana	50	42.2	5 (5%)	54 (4%)
Ohio	50	48.3	2 (2%)	31 (2%)
Georgia	8	--	5 (5%)	24 (2%)
Maryland	30	--	3 (3%)	22 (2%)
New York	15	--	3 (3%)	20 (2%)
Quebec	--	--	4 (4%)	18 (1%)
Nova Scotia	--	--	2 (2%)	11 (1%)
New Brunswick	--	--	1 (1%)	10 (1%)
Louisiana	60	--	2 (2%)	7 (1%)
Florida	45	--	2 (2%)	6 (0.5%)
Michigan	30	28.7	1 (1%)	4 (0.3%)
Missouri	25	3.4	1 (1%)	3 (0.2%)
Vermont	--	--	1 (1%)	2 (0.2%)
		TOTAL	91 (100%)	1279 (100%)

Drainage type

Much more nutrient load information was available for subsurface drainage compared to surface drainage (1177 vs. 56 site-years, respectively; Figure 3). This may be complicated by the fact that, in canal or ditch drainage, it is sometimes difficult to differentiate between surface and subsurface discharge as flows are generally combined groundwater and surface runoff (Evans et al., 1995). Many surface drainage ditch studies were not included in the Drain Load table as they represented more than a single land use or did not report annual nutrient loads (e.g., presented storm event data). Nevertheless, the practice of surface drainage is widespread practice across North America (Strock et al., 2007). Because ditch systems are unique in terms of geomorphology and nutrient cycling (Needelman et al., 2007), future ditch nutrient studies may an important contribution to the MANAGE Drain Load table.

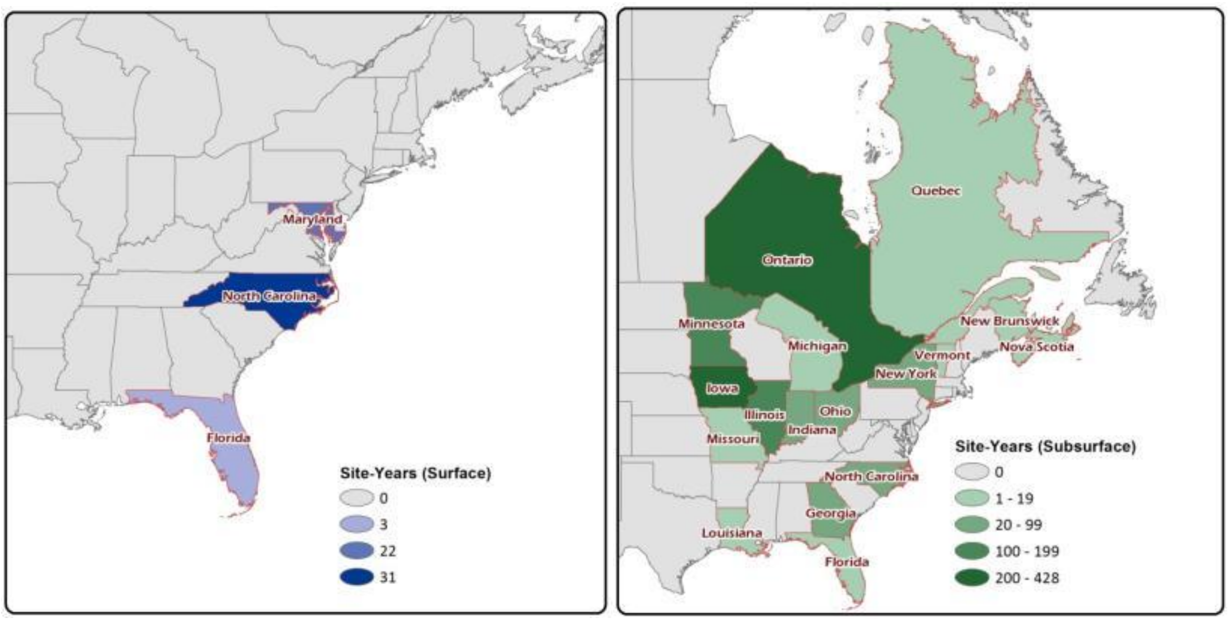

Figure 3: MANAGE Drain Load surface (left) vs. subsurface drainage site-years (right)

Surface intakes or inlets are an important component of many subsurface drainage systems, but mentions of these were curiously lacking across the literature review. Of the 1177 subsurface drainage site-years, only 22 (2%) were from studies that specified occurrence of surface intakes. Ball Coelho et al. (2010) established terminology of an “open” versus “closed” system referring to subsurface drainage systems with or without surface intakes, respectively. While several studies reported that surface inlets had a fairly small contribution to flow and nutrient loading (Ball Coelho et al., 2012a; Ball Coelho et al., 2012b; Ginting et al., 2000), pollution reduction approaches for these systems is an area of active research. Oolman and Wilson (2003) recommended use of slotted standpipes to control sediment and Smith and Livingston (2013) recommended blind inlets compared to tile risers.

Timeline

Although drainage nutrient studies date back to at least the late 19th century (Lawes et al., 1882), site-years in the Drain Load database ranged from 1961 to 2012 (Figure 4). There was a notable increase in the number of site-years in the 1990’s, potentially in response to nutrient concerns in the Mississippi River basin and Chesapeake Bay watershed (Hagy et al., 2004; Turner and Rabalais, 2003). More emphasis has been placed upon the study of drainage dissolved N loads as opposed to annual loads of other nutrients (Figure 4; note y-axes scales). The majority of total N and total P site-years stemmed from the 1960's. The more current interest in dissolved rather than total nutrient forms may be due to the increased understanding of the importance of reactive dissolved forms of nutrients in the environment.

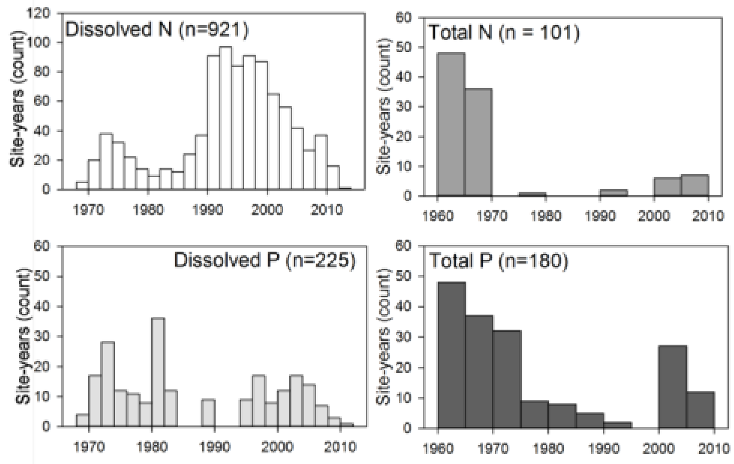

Figure 4: Histograms of Drain Load database dissolved nitrogen, total nitrogen, dissolved phosphorus, and total phosphorus site-year timing

Publication Source

The majority of site-years were sourced from the *Journal of Environmental Quality* and *Transactions of the ASABE* (33 and 27% of site-years, respectively; 43 and 18%, respectively, of total publications; Figure 5). The *Journal of Irrigation and Drainage Engineering* and *Agricultural Water Management* had curiously low counts considering the emphasis these outlets place on drainage studies. *Agriculture, Ecosystems and Environment*, the *Canadian Journal of Soil Science*, and *Water, Air, and Soil Pollution* all showed high site-years counts relative to their number of individual publications indicating their contribution of a few, high site-year studies. Nine other journals contributed only 5% of site-years that were generally from only one study per journal ("Others": *Air, Soil and Water Research*; *Canadian Water Resources Journal*; *Journal of Water Resource and Protection*; *Nutrient Cycling in Agroecosystems*; *Soil & Tillage Research*; *Soil Science Society of America Journal*; *Soil Use and Management*; *Water Science and Technology*; *Water Resources* which contributed two studies).

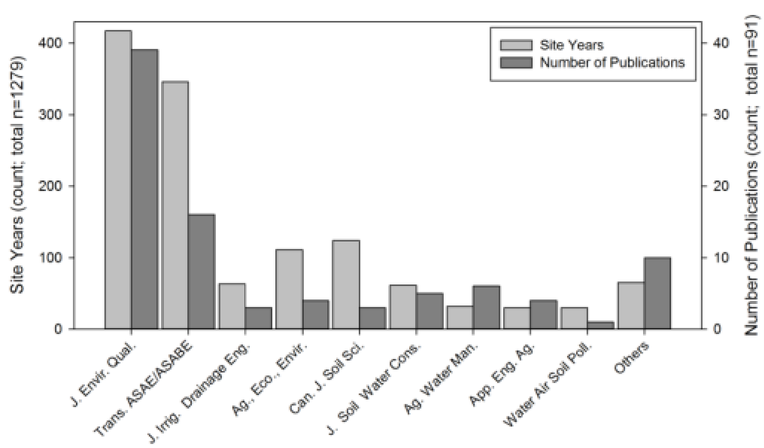

Figure 5: Peer-reviewed journals with studies/site-years included in the MANAGE Drain Load Table

Uncontrollable factors: Variable climate and hydrology

Precipitation and drainage discharge are strongly correlated with wetter years generating more drainage (Randall and Iragavarapu, 1995; Randall and Mulla, 2001). While total annual precipitation volume affects drainage discharge (Figure 6a), it is now also thought that in some locations, seasonal rainfall may be more relevant than the annual total (e.g., March – June in the upper Midwest; Bakhsh et al., 2007; Nguyen et al., 2013; Sands et al., 2008). Studies reporting the percent of annual precipitation occurring as subsurface drainage tend to give values ranging from approximately 15 to 40% (Figure 7). Where this could be calculated for the Drain Load table, the mean and median were 25 and 20%, respectively (n=827; Figure 7). Some variability can be explained by separating wet and dry years, with wet years resulting in a significantly higher percentage of precipitation occurring as drainage (Figure 6b).

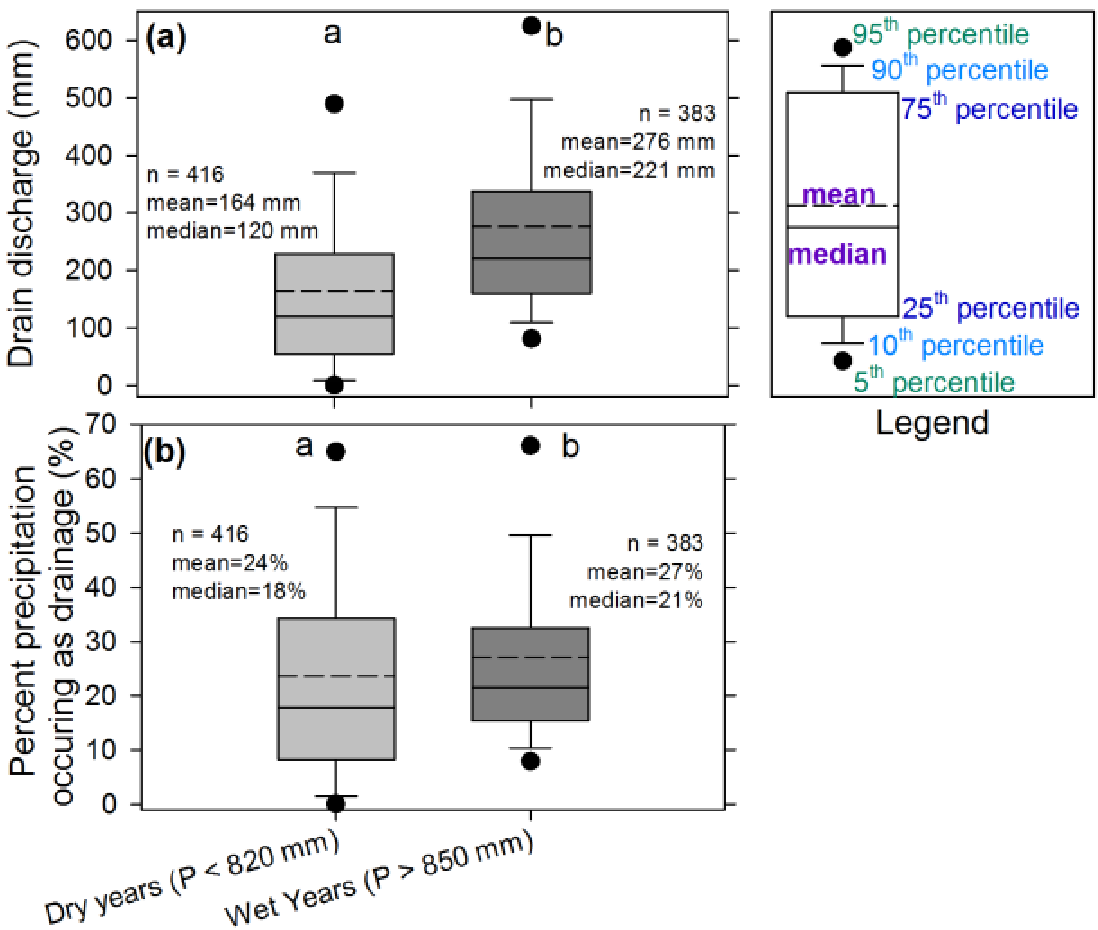

Figure 6: Drainage discharge (a) and percentage of reported precipitation occurring as drainage (b) for the MANAGE Drain Load table; medians with the same letters are not statistically significantly different based on a Mann-Whitney Rank Sum T-Test

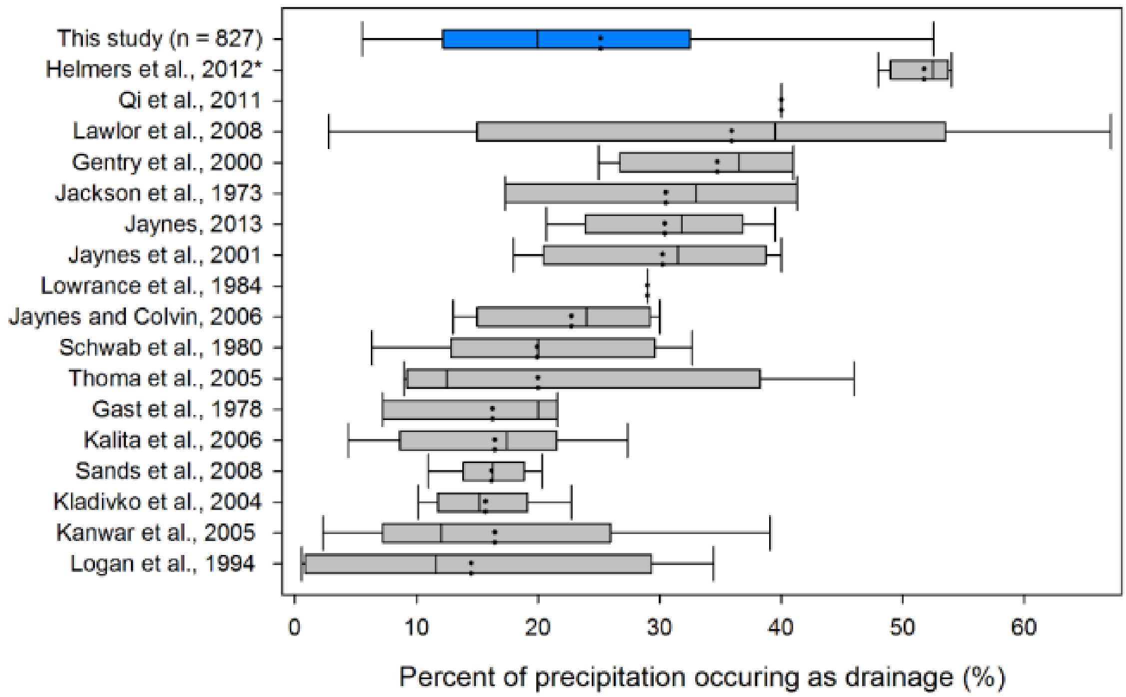

Figure 7: Percentage of precipitation occurring as subsurface drainage as reported by 17 studies (199 site-years) and over the Drain Load table; *Helmers et al. (2012b) considered the drainage season March-November only

Because drainage nutrient loads are highly dependent upon drainage volume (Bakhsh et al., 2002; Bolton et al., 1970; Nguyen et al., 2013), which is clearly related to precipitation trends, it follows that wetter years will have greater nutrient loads (Figures 8 and 9). The difference between wet and dry years was highly significant for dissolved N, dissolved P and total P loads at p<0.001, p<0.001, and p=0.002, respectively. The difference between years for total N loads was also significant but at a lower level potentially due to the low site-year count (p=0.072).

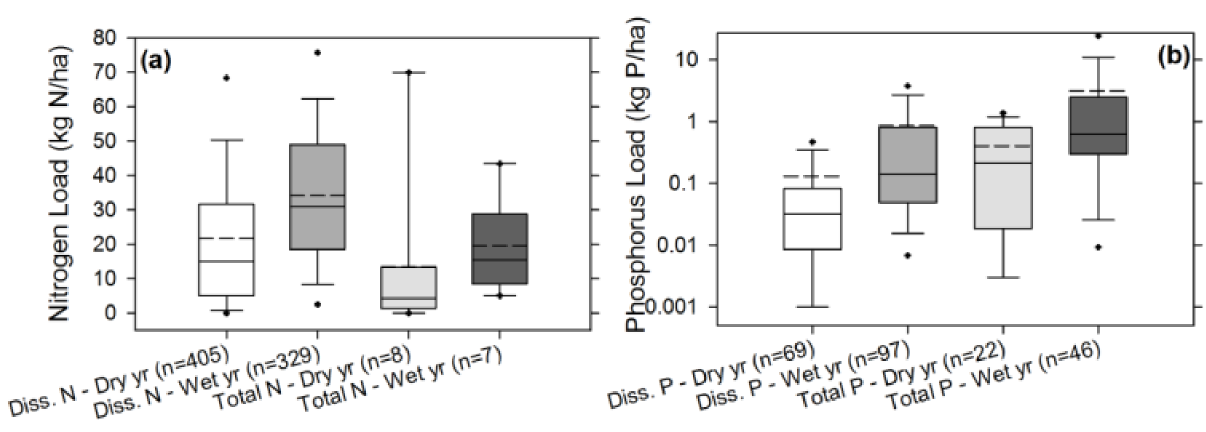

Figure 8. Dissolved and total nitrogen (a) and phosphorus (b; log axis) loads shown by wet or dry year from the MANAGE Drain Load database;

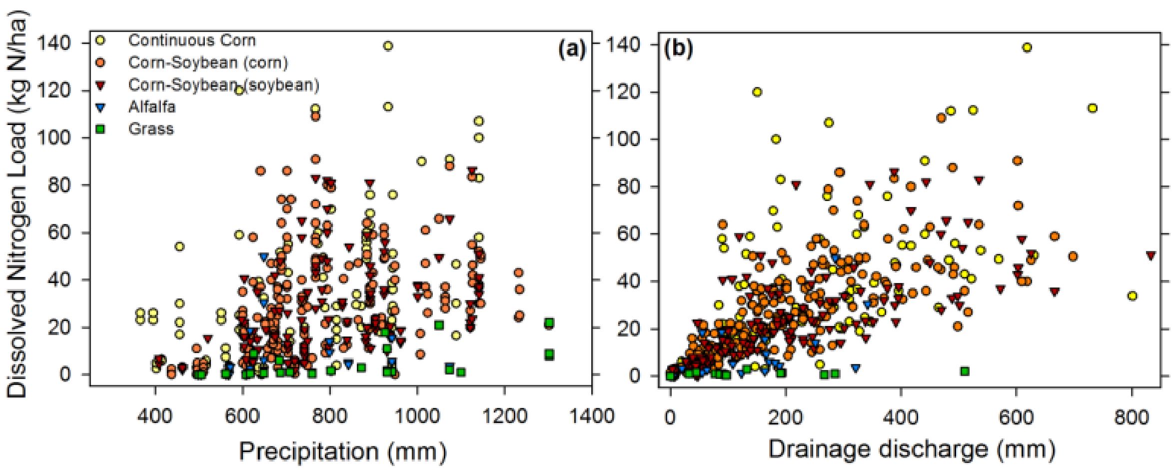

Figure 9: Dissolved nitrogen loads from five cropping types shown versus annual precipitation (a) and drain discharge (b); two high discharge outliers removed from (b)

There is now significant interest in understanding the impacts of increasingly variable climates on agriculture. In this context, “wet” and “dry” years may occur with higher frequency which has special implications for drainage discharge and nutrient loadings. Events such as hail storms or droughts that reduce crop growth may result in additional residual N in the soil available for drainage losses in subsequent years (Bakhsh et al., 2002; Bakhsh et al., 2005; Gentry et al., 2000; Gentry et al., 1998). Dry or drought years are especially associated with low nitrate losses, but high losses and/or concentrations the following year, particularly if the following year is wetter than average (Drury et al., 1993; Kladivko et al., 2004; Logan et al., 1994; Mitsch et al., 2001; Randall and Iragavarapu, 1995; Randall and Mulla, 2001; Sands et al., 2008; Tan et al., 2002b). For example, Bjorneberg et al. (1996) reported nitrate losses and annual flow weighted nitrate concentrations greater than 100 kg N/ha and 65 mg N/L during the wet year in a dry/wet cycle. To put this in context of production, Randall and Iragavarapu (1995) estimated the N loss during their observed dry year was less than 3% of the applied N, but in the wet year, this increased to 25 to 70%.

Aggregated data from studies where a reportedly dry year was followed by a wet year showed statistically significant differences between years in terms of precipitation, drainage discharge, and dissolved N loads, but not crop yields (Table 2: “selected” site-years). Compared against the entire dataset (i.e., Figures 6 and 8a but with the “selected” site-years removed), the drought years from the selected studies were indeed significantly drier than the pooled dry years. Drainage from the selected drought years was a significantly lower percentage of the precipitation compared to the other years (42 mm discharge; 8.3% of drought year precipitation). It is plausible that a year following a drought would experience a relatively lower percentage of the precipitation eluted as drainage due to a lingering soil moisture deficit. This was observed in terms of median values (18 versus 22% for selected wet and pooled wet years, respectively), although this result was not significant. While literature indicates these wet-following-drought years pose an elevated concern for extreme N loss, statistically, these selected wet years did not result in a greater dissolved N load than the pooled wet years (medians: 28.3 vs 30.2 kg N/ha; Table 2). Nevertheless, the mean dissolved N load from the selected wet years was slightly higher than the mean from the pooled wet years (35.8 and 33.9 kg N/ha, respectively; data not shown) indicating the impacts of climate variability on drainage nutrient loads is a potential topic meriting further investigation.

Table 2: Median (count) for precipitation, drainage discharge, dissolved nitrogen load and crop yields from eight studies that reported a drought followed by a wet year (Gentry et al., 2000; Gentry et al., 1998; Kladivko et al., 2004; Logan et al., 1994; Randall et al., 1997; Randall and Iragavarapu, 1995; Sands et al., 2008) compared against the remainder of the Drain Load dataset grouped by above- and below-average precipitation; medians with the same letters are not statistically significantly different based on a Kruskal-Wallis One Way Analysis of Variance on Ranks

	Precipitation	**Drainage discharge**	**Precipitation lost as drainage**	**Dissolved Nitrogen Load**	**Yield**	
					Corn	**Soybean**
	---------- mm ----------		%	**kg N/ha**	--------- Mg/ha ---------	
Selected dry/drought years	562(46)d	42(46)c	8.3(46)b	3.1(46)c	5.9(21)c	2.4(7)bc
Selected wet-following-drought years	933(40)b	180(40)b	18(40)a	28.3(40)ab	6.6(17)bc	2.6(6)ab
Remaining Drain Load table pooled dry years**	689(381)c	132(357)b	20(357)a	18.1(346)b	8.5(173)b	3.3(59)ab
Remaining Drain Load table pooled wet years**	1001(394)a	230(356)a	22(356)a	30.2(302)a	9.2(125)a	3.4(47)a

** Selected drought years and wet-following-drought years removed from the pooled data

These data lend evidence to the premise that surplus residual soil N following a poor crop yielding-year may be available for leaching in the future. Corn and soybean yields were lower in the selected drought years and wet-following-drought years compared to their respective pooled datasets; these differences in corn yields were significant (5.9 versus 8.5 Mg/ha and 6.6 versus 9.2 Mg/ha for dry and wet years, respectively).

Controllable factors

Crop selection and rotation

The most prevalent cropping systems within the MANAGE Drain Load database were a corn and soybean rotation and continuous corn (43 and 23% of site-years, respectively; Figure 10). Corn additionally featured prominently as part of other rotations (e.g., Corn-Oat-Alfalfa, Corn-Soybean-Oats, Corn-Wheat-Soybean). Thirty-five individual crops were represented, with over half the site-years planted to corn (including seed, silage, and white corn; Figure 10). Soybeans were also a major contributor at 27% of site-years, followed by alfalfa (6%), grasses ("grass", prairie, miscanthus, switchgrass; 5%), and oats (2%). "Other" crops included barley, cabbage, carrots, citrus, cotton, onions, peas, peanuts, potato, rye, snap beans, sugarcane, and wheat (5%).

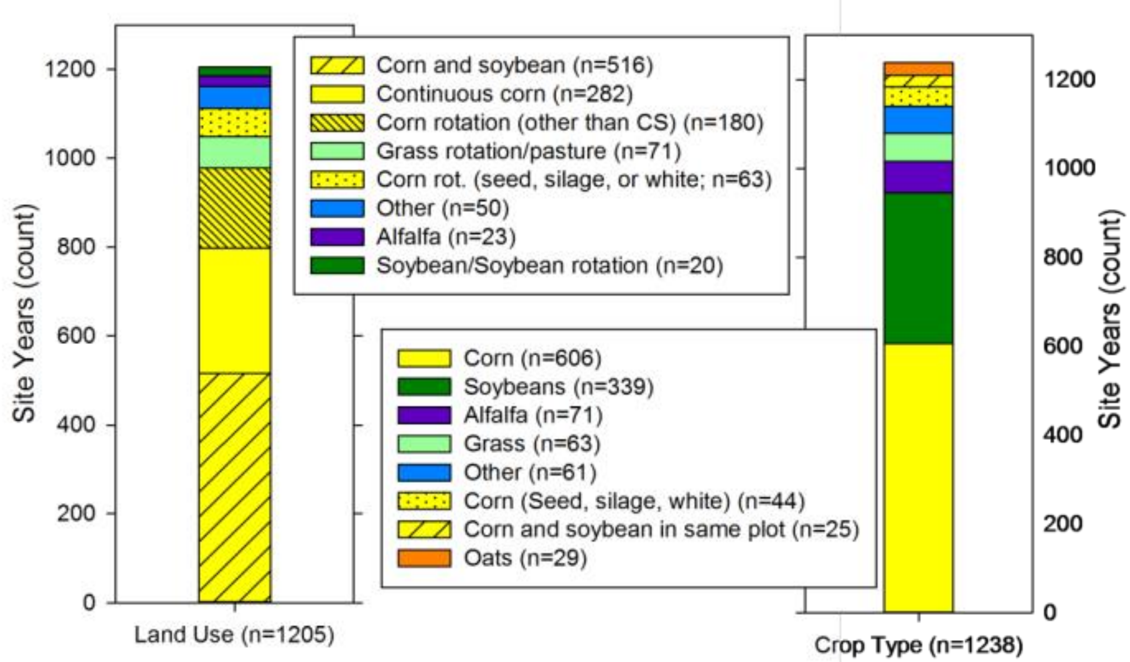

Figure 10: Land use (left) and crop type (right) across the MANAGE Drain Load database

There was no significant difference in drainage discharge or dissolved N load between the two most common cropping rotations, continuous corn and a corn-soybean rotation, in either wet or dry years (Figure 11 a-d). Higher flow weighted nitrate concentrations have been observed from continuous corn systems compared corn-soybean rotations (Kanwar et al., 1997; Randall et al., 1997), with several studies indicating continuous corn will also result in greater nitrate loads (Kanwar et al., 1997; Owens et al., 1995; Weed and Kanwar, 1996). However, Klocke et al. (1999) noted higher N leachate loads from a corn-soybean rotation. Helmers et al. (2012b) and Kanwar et al. (1997) both reported higher corn yields from corn rotated with soybeans versus continuously grown corn (both at recommended N application rates); this difference was shown to be significant here in both wet and dry years (Figure 11 e and f).The lack of significant difference in discharge and N load between the two systems is consistent with the variability in literature (Bakhsh et al., 2005). It is worth noting that when only the corn rotations were included in statistical analyses, continuous corn resulted in significantly greater discharge than the corn phase of the corn-soybean rotation ($p=0.020$) and significantly greater dissolved N loads than the soybean phase ($p=0.031$) but both only in wet years. Dry years still showed no difference between the three treatments in discharge or N load ($p=0.333$ and $p= 0.272$; i.e., consistent with the broader analysis in Figure 11).

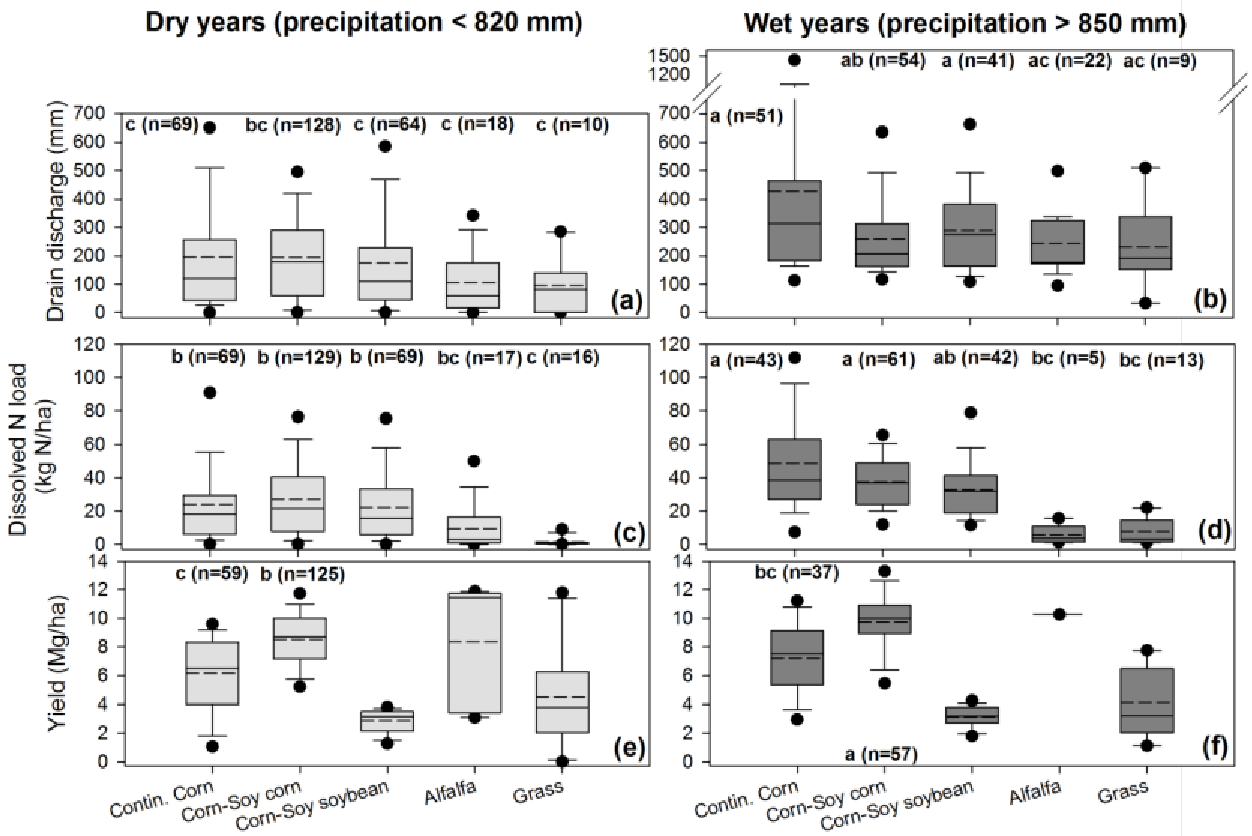

Figure 11: Drainage discharge (a, b), dissolved nitrogen load (c, d), and crop yield (e, f) between crops grouped by above- and below-average precipitation; medians with the same letters are not statistically significantly different based on a Kruskal-Wallis One Way Analysis of Variance on Ranks

There was no drainage difference between the two crop phases of a corn-soybean rotation (Figure 11 a-d), which was consistent with some findings (Lawlor et al., 2008; Logan et al., 1994), but not with other reports that soybeans produce greater drainage discharge than corn (Bakhsh et al., 2007; Drury et al., 2014). Investigating further into only studies where corn and soybeans were both grown in the same year on separate plots or fields, resulted in no statistically significant difference between the drain discharge from the two crop phases (Figure 12a; n=31 from 6 studies; Bakhsh et al., 2009; Bakhsh et al., 2002; Bakhsh et al., 2005; Bakhsh et al., 2007; Bjorneberg et al., 1996; Randall et al., 1997). The slope indicated that every 1.0 mm increase in discharge from corn would result in a 1.0 mm increase in the discharge from soybeans; however this relationship is offset by 20 mm (i.e., the y-intercept) indicating that soybeans result in a relatively minimal 20 mm greater discharge than corn within the rotation.

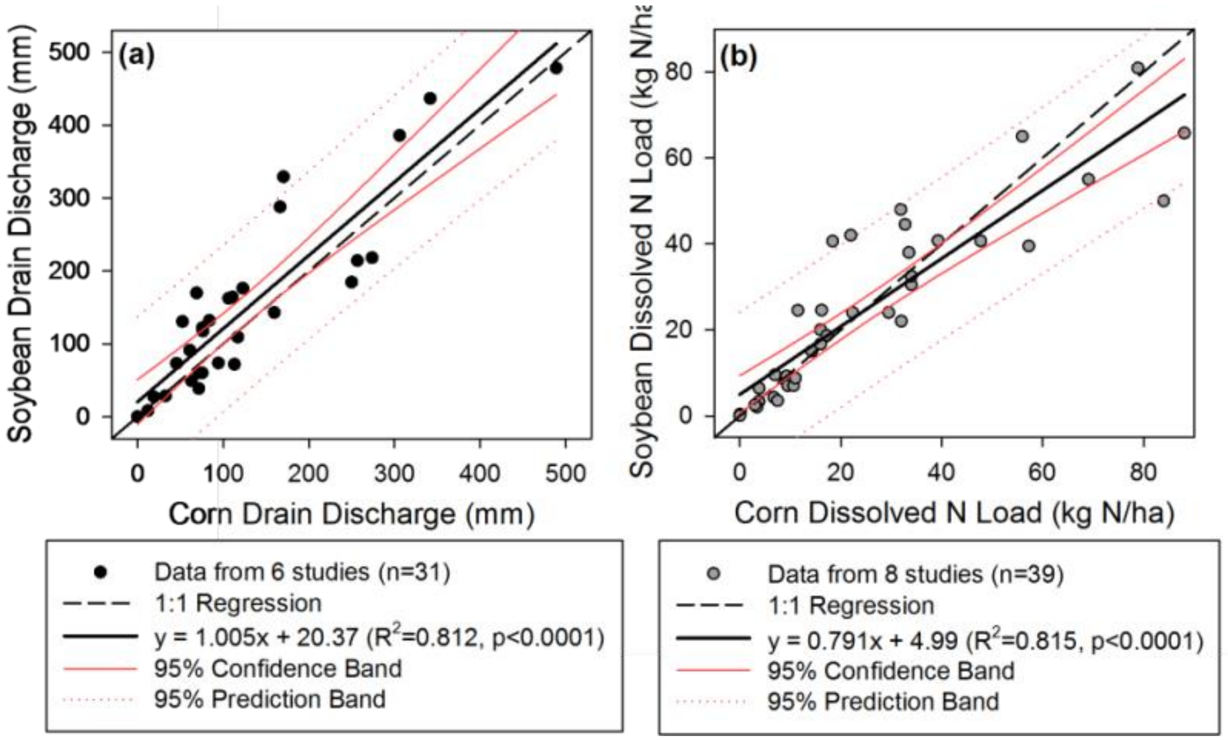

Figure 12: Corn and soybean drainage discharge (a) and dissolved N load (b) from selected studies where corn and soybeans were both grown in the same year on separate plots or fields

Similar to drainage discharge, there is variability in corn versus soybean phase N loss in literature. Jaynes and Colvin (2006) and Qi et al. (2011) both observed the soybean year had lower annual flow weighted nitrate concentrations than the corn year (not significant), although Logan et al. (1994) reported the soybean phase nitrate concentrations were as high or higher than the maize phase. Strock et al. (2004) reported soybeans had higher N loss and lower residual soil N compared to corn. Zhu and Fox (2003) found that soybeans may leach more N than corn at low corn N application rates, but the two phases were not different at an application rate of 200 kg N/ha to corn. Randall and Vetsch (2005) documented that 54 versus 46% of the N load occurred in the corn versus soybean phases (i.e., very similar), and Bakhsh et al. (2005) saw no significant N load difference between a corn-soybean and soybean-corn rotation over a six year study. Here, based from the selected corn-soybean studies, the slope of the dissolved N load regression between the two phases showed that for every additional kilogram of N per ha lost in drainage from the corn phase, only 0.8 kg N/ha would be lost from the soybean phase (Figure 12b; n =39 from 8 studies; studies above plus Lawlor et al., 2008; Logan et al., 1994). This lends support to findings of greater N loads from the corn phase. Nevertheless, the 95% confidence bands of this regression overlapped the 1:1 line for the most commonly reported N load range, likely indicating no practical difference in dissolved N load exists between the two phases. From a field-scale perspective, this lends credence to the notion that corn-soybean rotations will result in a net negative N balance (Gentry et al., 2009; Jaynes and Karlen, 2008), as both crop phases leach similar N loads, soybeans typically do not receive N fertilization, and they symbiotically fix less N than is exported in grain harvest. This finding also supports the investigation of corn-soybean rotations as a combined system (i.e., a given research plot planted to both corn and soybeans in a given year; e.g., Lawlor et al., 2008; Lawlor et al., 2011; Nguyen et al., 2013).

In the upper Midwest and Canada, the majority of the annual drainage volume and nitrate loss occurs during the spring when conventional row crops are not growing or during the very early growing season (Bakhsh et al., 2007; Ball Coelho et al., 2012a; Bjorneberg et al., 1996; Bryant et al., 1987; Gangbazo et al., 1997; Gentry et al., 2000; Kladivko et al., 2004; Lawlor et al., 2008; Randall and Vetsch, 2005; Randall et al., 2003). To address this crop-soil-water imbalance and to reduce this "asynchronous production and uptake of nitrate in the soil" (Cambardella et al., 1999; Sands et al., 2008), the use of perennials and diverse cropping rotations is an important area of research interest (Dinnes et al., 2002). Perennial crops (e.g., alfalfa, miscanthus, switchgrass, perennial forage, prairie) are widely thought to reduce both subsurface drainage discharge and nitrate loads compared to annual crops (Benoit, 1973; Burwell et al., 1976; Qi et al., 2011; Smith et al., 2013). Randall et al. (1997) reported that drainage nitrate losses were more than thirty-fold higher for row crops than perennials due to more prolonged uptake of water and N of the latter. Here, there was no significant drainage-reduction benefit of perennials compared to conventional crops in either dry or wet years, although alfalfa and grasses consistently had lower median/mean discharge volumes (Figure 11a and b). This lack of significance for perennial-induced reduction in drain flow may have been due to the relatively low site-year population size for alfalfa and grass crops. In terms of dissolved N, the alfalfa and grass median loads were always less than 4 kg N/ha whereas the other treatments were greater than 15 and 30 kg N/ha in dry and wet years, respectively.

Wetter years produced more drainage discharge regardless of the cropping option, though not always significantly (Figure 11a and b). It is plausible perennial crops provide buffering of drainage volumes and N loads in wet years, as both alfalfa and grass showed no significant increase in discharge or load between wet and dry years, whereas continuous corn and the soybean phase of the corn-soybean rotation produced significantly more drainage in wet years, and continuous corn and the corn phase produce significantly greater N loads (Figure 11c and d). Some caution is issued, however, when considering a perennial legume such as alfalfa, as this crop can result in drainage N losses especially after plow-down or overwintering (Fleming, 1990; McCracken et al., 1994). Additionally, it is worth noting that a drainage water quality benefit of perennials is not exclusively observed across all years in all studies where multiple cropping strategies were employed (Kanwar et al., 2005; Tan et al., 2002b). While choice of cropping system is one of the most "controllable" factors for having a major impact on drainage N loads (Randall and Goss, 2008), adequate economic returns are often challenging for the production of perennials (Randall and Mulla, 2001) and such a major paradigm shift in farming practice may be difficult to overcome (Qi et al., 2011).

Tillage

Conventional tillage (moldboard plow, "conventional tillage"), conservation tillage (ridge till, chisel plow, "conservation tillage"), no till, and pasture each accounted for 430, 202, 170, and 34 site-years, respectively, of the total 836 site-years where a tillage practice was reported. Drain discharge differences between tillage types were most apparent during the dry years with the practice of conventional tillage yielding significantly greater discharge than conservation and pasture tillage practices (Table 3). Across literature, drainage discharge is reported to be greater with no-tillage compared to conventional tillage (or other forms of conservation tillage) due, in part, to increased macropore flow under no-till (Bakhsh et al., 2007; Bjorneberg et al., 1996; Blann et al., 2009; Patni et al., 1996; Randall and Iragavarapu, 1995; Tan et al., 2002a). This drainage volume difference between no-till and conventional tillage may be as high as 2 to 3 times (Endale et al., 2010). However, the increased presence of macropores under no-tillage systems may cause infiltrating water to have decreased soil contact and relatively lower drainage nitrate concentrations compared with intensive tillage (Angle et al., 1993; Bjorneberg et al., 1996; Kanwar et al., 1988; Kanwar et al., 1997). Thus, this potential for greater flow volume from no-till systems is confounded by this practice's reduced drainage nitrate concentrations; this combination may mask any significant differences in N loading due to no-till. No-till sometimes results in greater N loads than more conventional tillage practices (Bakhsh et al., 2002; Kanwar et al., 1997; Patni et al., 1996), but not always (Francesconi et al., 2014; Randall and Iragavarapu, 1995). Here, the practice of no-till had lower dissolved N load means and medians compared to the conventional and conservation treatments in both wet and dry years, though this difference was not statistically significant.

Table 3: Median (count) for drain discharge, percentage of precipitation resulting as drainage, and dissolved nitrogen loads in dry and wet years for four tillage practices; medians with the same letters for a given parameter (including dry v. wet year comparisons) are not statistically significantly different based on a Kruskal-Wallis One Way Analysis of Variance on Ranks

	Drainage discharge		Percent of precipitation occurring as drainage		Dissolved N Load	
	------------ mm ------------		--------------- % --------------		------------ kg N/ha ------------	
	Dry*	Wet*	Dry	Wet	Dry	Wet
Conventional	165 (221) b	259 (137) a	23 (222) ab	25 (138) a	23.0 (217) b	37.0 (141) ab
Conservation	92 (74) cd	221 (49) ab	13 (74) bc	20 (49) ab	13.6 (72) b	37.3 (47) a
No till	120 (66) bc	215 (52) ab	18 (67) b	20 (52) ab	9.1 (63) bc	22.0 (51) ab
Pasture	0 (11) d	200 (8) ab	0 (11) c	22 (7) ab	0.0 (11) c	3.0 (3) bc

* Dry years: precipitation < 820 mm; Wet years: precipitation > 850 mm

Comparing beyond no-tillage systems, Weed and Kanwar (1996) reported higher drainage N losses for chisel plow compared to ridge plow, moldboard plow, and no-till. However, Karlen et al. (1998) reported the two lowest 15-yr N loads were from chisel and ridge tillage systems (467, 369, 352, and 466 kg N/ha for moldboard plow, chisel plow, ridge tillage, and no tillage treatments, respectively). Drury et al. (1993) similarly reported lower N losses and drainage discharge from a conservation tillage treatment compared to conventional tillage, which was consistent with this analysis, but not significantly (Table 3). There is variability in any conclusion on this subject, however, as some have documented no tillage-induced drainage flow effect (Logan et al., 1994; Tan et al., 1993). In the end, tillage management may play a fairly minor role in determining drainage N loads (Kanwar et al., 1997; Randall and Goss, 2008; Randall and Mulla, 2001).

Not only did the pasture treatment experience increased drainage volumes in wet versus dry years, but this was also the only treatment where the percentage of precipitation experienced as drainage statistically increased in the wet year (Table 3). This may mean during a wet year, increases in drainage flow may result even with a “conservation-oriented” land cover and tillage approach. Nevertheless, importantly for water quality goals, the pasture treatment consistently leached the lowest dissolved N loads (Table 3; note the low site-year population).

Drainage system design

Tile drainage spacing in the Drain Load database ranged from 2.5 to 43 m (one 100 m spacing outlier was removed from analysis; Gilliam and Skaggs, 1986), with 27 and 24% of studies using 7.6 and 28.5 m spacing, respectively (Figure 13a). Drain depths ranged from 0.5 to 1.5 m with a third of studies using a 1.2 m depth which is consistent with conventional tile drain depth in the US Midwest (Figure 13b). Due to the low number of site-years and lack of information reported for surface drainage studies, this drainage design section focused on subsurface design trends.

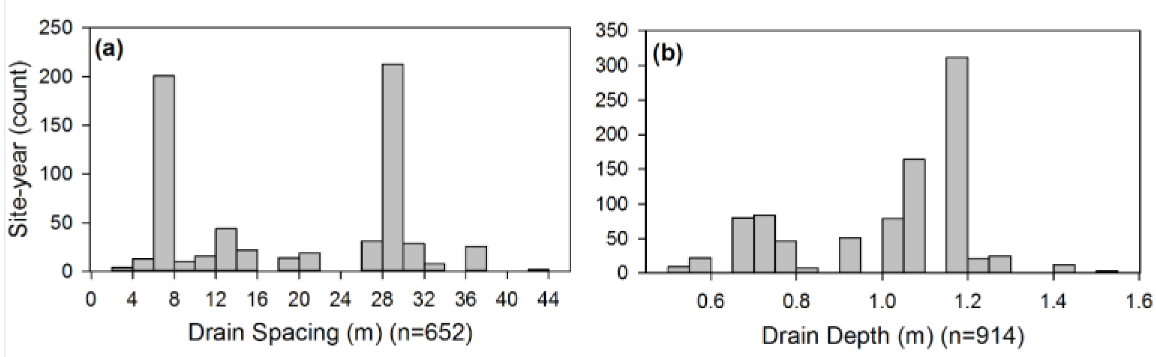

Figure 13: Prevalence of Drain Load database site-years by either drain spacing or drain installation depth

Wider drain spacing reduces drainage discharge and N loading (Davis et al., 2000; Hoover and Schwab, 1969; Kladivko et al., 2004; Sands et al., 2008) but may decrease crop yield if trafficability is reduced (Bolton et al., 1980; Skaggs et al., 2005). The increase in nitrate loading with narrow drain spacing is a factor of increased flow rather than differences in nitrate concentrations (Kladivko et al., 2004). Binning the Drain Load spacing data into discrete groups confirmed narrow drain spacing tended to elute greater dissolved N loads, although differences in drainage discharge between spacings were less clear (Figure 14a and b). Here, a smaller bubble size indicated a larger data population and thus reduced uncertainty surrounding the median of a group of binned drainage spacings.

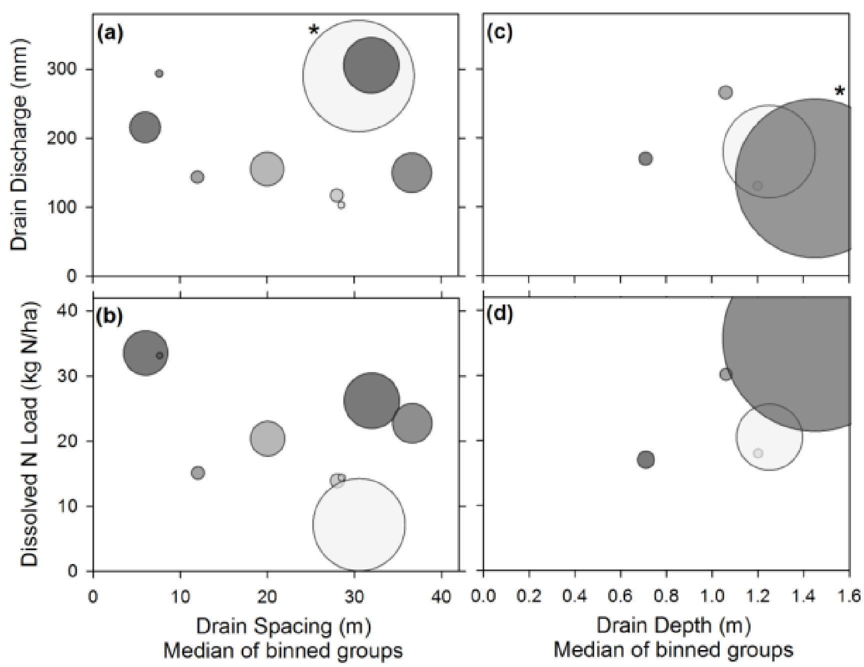

Figure 14: Median drain spacing (a and b) and drain depth (c and d) of binned spacing- or depth-groups graphed against the median drain discharge (a, c) and median dissolved nitrogen load (b, d); bubble size determined by the scaled inverse of the population for each binned group (spacing bubble size = 10/n; depth bubble size = 25/n), thus smaller bubbles indicate larger populations and greater certainty; the starred spacing and depth bubbles were scaled by 1 and 10, respectively due to small populations (n=1 and 7, respectively)

Shallow placement of drainage pipes decreases drainage discharge and associated nitrate loads compared to conventional depths, the latter of which was corroborated here (Figure 14c and d; (Burchell et al., 2005; Davis et al., 2000; Gordon et al., 2000; Helmers et al., 2012a; Sands et al., 2008; Schwab et al., 1980; Smith and Kellman, 2011). Any such shallow placement must be couched within proper depth of installation to avoid structural failures (ASABE, 2014; Gordon et al., 2000) and take into consideration potential yield impacts (Gordon et al., 2000; Kalita and Kanwar, 1993; Smith and Kellman, 2011), though these may be minimal (Helmers et al., 2012a).

Similar to wider drainage spacing, the reduction in dissolved N load due to shallow drain placement is largely due to reduction in drainage volume rather than changes in nitrate concentration (Sands et al., 2008). The impact of shallow drainage on nitrate concentrations is variable; Sands et al. (2008) found no significant difference between shallow and deep drainage flow weighted nitrate concentration (though the shallow treatment concentrations tended to be lower), and Helmers et al. (2012a) reported shallow drainage had higher concentrations than conventional drainage. Graphing drainage discharge by dissolved N load for different binned drain depth groups revealed a distinct trend in the binned groups' regressions (Figure 15). As the drain depth increased, the regression slope increased and y-intercept decreased. The slopes indicated at a given drainage discharge, a deeper drain depth would result in a disproportionately higher dissolved N load. For example, application of these slopes yields that for 100 mm additional drainage discharge between two hypothetical years (e.g., a normal vs a wet year), a drain placed within 1.0 m of the soil surface would generate 8.4 kg N/ha of additional N load in the wet year, whereas a drain placed 1.2 m below the soil surface would generate an additional 11 kg N/ha. The difference in y-intercepts is also interesting, as these seem to indicate that in dry years (i.e., when drain discharge is around 0 mm at the y-intercept), deeper drains have lower N loads. This may be rooted in the concept that deeper placement of drains provides greater volume of soil storage for precipitation especially notable in dry years. In such years, this increased storage capacity and the theoretical potential for increased denitrification over this volume may act to reduce N loads. The y-intercepts tended to be less statistically significant than the slopes, however. While there is wide scatter between these data, the impact of drainage spacing, particularly in wet versus dry years, is an important topic meriting further investigation.

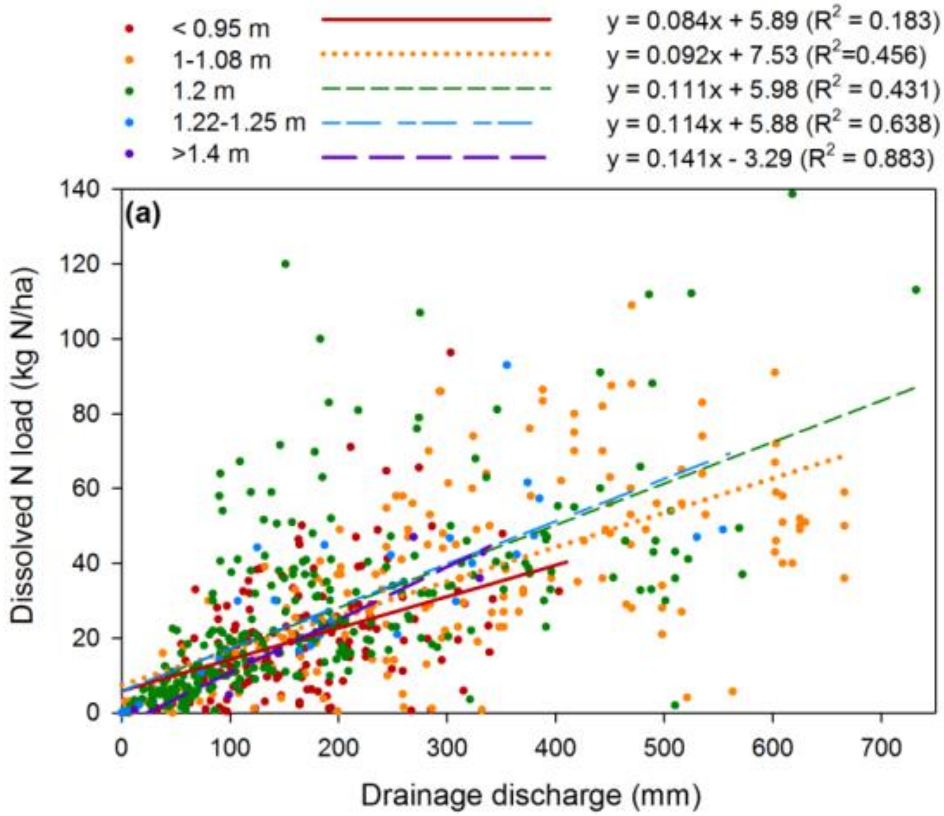

Figure 15: Drainage discharge and dissolved nitrogen load relationships for five groups of binned drain depths

Gaps in knowledge

An important outcome of any review-oriented study is the identification of potential gaps in scientific understanding. The Drain Load database development underscored several areas for future drainage research including more intensive year-round monitoring (in some locations) and improved monitoring in newly drained areas, ditch drained areas, and surface intakes. Perhaps most importantly, more long-term studies with coordinated controls across multiple sites and years would improve knowledge of drainage-associated nutrient loads. Randall and Vetsch (2005) noted the common lack of statistically significant differences in N loss between treatments is not surprising considering this metric compounds variability associated with both drainage flow and nitrate concentration. For more robust statistical comparisons, long-term studies are necessary to increase statistical power to overcome both the strong effect and high variability of precipitation (Randall et al., 2003).

In the northern Midwest, winter drainage usually ceases due to low precipitation or frozen soil (Kalita et al., 2006). Challenging field conditions, restricted site access, and limitations of monitoring equipment mean that drainage potentially occurring over this period, particularly snow melt drainage, will not be captured (Ball Coelho et al., 2010; Ball Coelho et al., 2012a; Fleming, 1990; Milburn et al., 1990). Drainage hydrology during the late winter/early spring is complicated by the breaking of the 'freeze line' which may result in a very rapid transition from no drainage to pipe-full flow, by diurnal freeze/thaw cycles, and by rain falling on snow (Ball Coelho et al., 2012b; Bottcher et al., 1981). Studies that have been able to avoid retiring monitoring equipment over the winter have noted the importance of monitoring year-round and suggest that better and more intensive monitoring strategies are needed to capture these critical missed periods (Ball Coelho et al., 2012a; Milburn and MacLeod, 1991). Snowmelt drainage can contribute significantly to total annual drainage volume and nutrient/sediment loading (Ball Coelho et al., 2012a; Ball Coelho et al., 2012b; Gangbazo et al., 1997; Jamieson et al., 2003; Klatt et al., 2003; Milburn et al., 1990). In a study specifically intended to investigate snowmelt contributions, Ginting et al. (2000) found that snowmelt drainage mobilized dissolved pollutants, whereas storm event drainage may be responsible for more of the particulate and sediment-bound nutrient loads.

By noting the Drain Load studies that retired monitoring equipment over the winter, it was possible to separate potential effects between studies that monitored all year versus studies that represented early spring through late fall as the annual period (Table 4). The site-years covering a full year had significantly higher precipitation and drainage discharge compared to the site-years where the monitoring equipment was retired for the winter. Confoundingly, the studies where it was necessary to winterize monitoring equipment yielded significantly higher dissolved N loads than full year-studies, which may be an indication the former were not underestimating loads. There is likely a spatial/geographic aspect for this discussion, however, as the majority of the “not full year” studies were from Iowa. It may be that certain locations will not significantly underestimate nutrient losses by excluding winter months (e.g., Iowa), whereas full years of monitoring will be critical for other locations (e.g., Ontario, New York, Indiana).

Table 4: Median (count) for precipitation, drainage discharge, and dissolved nitrogen loads from studies reporting a full year versus those reporting early spring though late fall; medians with the same letters are not statistically significantly different based on a Mann-Whitney Rank Sum T-Test

	Precipitation	Drainage Discharge	Dissolved N Load
	-------------------- mm	----------------------	kg N/ha
Not a full year	790 (331) b	170 (393) b	24.0 (401) a
Full year	889 (489) a	191 (565) a	20.0 (517) b

There are clearly gaps in peer-reviewed records of annual drainage nutrient loads from certain geographic regions and from certain types of drainage systems (Figures 2 and 3). The lack of site-years from areas widely practicing ditch drainage (e.g., Maryland, Delaware) complicates efforts to develop drainage best management practice guidance specific for these locations (Shirmohammadi et al., 1995). Fortunately, recent efforts are aiming to address this (Bryant et al., 2012; Penn et al., 2007)**.** Improved knowledge about the water quality impacts of surface intakes could also contribute to water quality improvement efforts (Ginting et al., 2000; Schilling and Helmers, 2008). There were only four studies included in the Drain Load database that specifically mentioned surface inlets (Ball Coelho et al., 2012a; Ball Coelho et al., 2012b; Bottcher et al., 1981; Hanway and Laflen, 1974), despite their widespread implementation.

It is critical for the Dakotas and other newly drained areas (e.g., Jia et al., 2012; Nash et al., 2014); Table 1) to continue to build their library of region-specific drainage water quality data especially considering new drainage systems, and their associated soil disturbance, pose a special concern for water quality (Fausey, 1983; Ritter et al., 1995; Roberts et al., 1986). The long history of drainage in many locations across North America means that many aging drainage systems are now starting to be replaced and upgraded. This is a pivotal opportunity to upgrade systems not only for improved crop growth and yield but also to advance water quality goals (Strock et al., 2010). The redesign of existing drainage systems means understanding drainage design impacts becomes increasingly important.

Conclusions

Artificial drainage will remain a vital component of many agricultural systems across North America, and improved understanding of management impacts, especially under variable climate conditions, can help point the way to a more sustainable future. This compilation of nearly 1300 drainage nutrient load site-years in the new MANAGE Drain Load table facilitated quantitative analyses of the history of drainage water quality research in North America.

Across site-years, the mean and median percent of precipitation occurring as drainage were 25 and 20%, respectively, with wet years resulting in significantly greater drainage discharge, percentages of precipitation eluted as drainage, and nutrient loads. In terms of controllable factors, no significant difference was observed in drainage discharge or N loads between continuous corn and corn-soybean cropping systems, although corn in rotation showed significantly greater yields. The evidence also supported investigation of corn-soybean rotations as a single system as there was no significance difference in discharge or N load between the two phases. Alfalfa and grasses provided notable N loading benefits compared to row crops and small grains, but these comparisons were limited by low site-year counts. Consistent with literature, tillage management resulted in no clear best practice to reduce drainage discharge or N loads. Wider drain spacing and shallower drain placement tended to decrease N loss in subsurface drainage, but the aggregated effects on drainage discharge were less apparent.

As drainage water quality research continues into the 21st century, MANAGE's Drain Load table would benefit from annual nutrient loading data in newly drained areas, ditch drained areas, and areas where surface intakes are specified. Most importantly, more long-term drainage nutrient loss studies with coordinated controls across multiple sites and years would increase statistical power for more robust comparisons in the future.

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4Rs water quality impacts: A review and synthesis of forty years of drainage nitrogen losses

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Abstract

The intersection of agricultural drainage and nutrient mobility in the environment has led to multi-scale water quality concerns. This work reviewed and quantitatively analyzed nearly 1300 site-years of drainage nitrogen load data to develop a more comprehensive understanding of the impacts of 4Rs practices within drained landscapes across North America. Using the new Drain Load table in the existing "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database, relationships were developed across N application rates for dissolved N drainage loads and corn yields. Corn-soybean rotations and use of organic N sources were more favorable for N loss and corn yield than continuous corn and inorganic N sources, respectively. The lack of significant differences between N application timing or application method treatments indicated neither should receive primary focus as dissolved N load reduction strategies (p = 0.934 and 0.916, respectively). Broad-scale analyses such as this can help identify major trends for water quality, but accurate implementation of the 4Rs approach will require site-specific knowledge to balance agronomic and environmental goals.

Introduction

Changing global diets, intensified climate variability, and increasingly degraded land and water resources have created an urgent need to revisit agriculture's approach towards sustainability. This mounting complexity now requires agricultural producers to undertake new and redefined roles as integrated landscape managers, directors of natural capital, and ecosystem service suppliers, beyond their more overt responsibilities as providers of food, fiber, and fuel. With society demanding that farmers now balance multiple environmental and agronomic objectives, it is critical to provide comprehensive and useful information to producers and industry stakeholders on the impacts of recommended on-farm management practices.

In most agronomic systems, nutrient additions are used to enhance the productivity of soil resources. The 4R Nutrient Stewardship approach to nutrient management is an integrated strategy developed to foster achievement of agricultural production goals while minimizing associated negative environmental, economic, and social effects. Simply put, this approach advises the application of the right source of nutrients, at the right rate and time, and in the right place (4R Nutrient Stewardship, 2015). Despite the simplicity of and increasing momentum behind this concept, accurate implementation of the 4Rs requires site-specific knowledge of any given field's biophysical and social constraints in tandem with associated economic and production goals.

In addition to the 4Rs framework, maintained and improved artificial agricultural drainage networks are used in many areas of North America to meet production goals (Blann et al., 2009; Skaggs et al., 1994; Skaggs and van Schilfgaarde, 1999). Unfortunately, the intersection of agricultural drainage and nutrient mobility in the environment has led to multi-scale water quality concerns (David et al., 2010; Shirmohammadi et al., 1995; Thomas et al., 1995). Nitrogen (N) management within drained landscapes is particularly vexing due to the mobility of nitrate within the soil profile. Variation in soil type, weather, climate, drainage system, and management practices, among other factors, affect drainage N losses (Randall and Goss, 2008; Skaggs and van Schilfgaarde, 1999). This means individual N management practices will have differing effectiveness and differing compatibility with farm management and profitability goals in any given location and year.

While studies of the agronomic and environmental impacts of improved N management practices have been conducted, there is a need to assemble and further analyze these previous works to develop a more comprehensive understanding within drained landscapes. In particular, an applied synthesis of water quality and crop yield effects of the 4Rs in artificially drained agronomic systems would enrich knowledge across industry, academia, and agencies. To this end, nearly 1300 site-years of drainage nutrient studies have recently been compiled in a free, publically available database. This new Drain Load table in the existing "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database provides comprehensive N and phosphorus (P) load data from peer-reviewed studies across North America. The addition of drainage studies complemented the greater than 1800 existing agricultural and forest runoff watershed-years in this database hosted by the United States Department of Agriculture, Agricultural Research Service, Grassland, Soil, and Water Research Laboratory in Temple, Texas (www.ars.usda.gov/spa/manage-nutrient;Harmel et al., 2006; Harmel et al., 2008).

This work aimed to use the drainage data in the new MANAGE Drain Load database to better identify and define the consequences of the 4Rs N management strategies. Specifically, this analysis asked the questions "How do the 4R practices affect N losses in waters from artificially drained agricultural fields?", and (2) "How do the 4R N practices affect crop yield and on-farm profitability in drained agronomic systems?"

Methods

The MANAGE Drain Load table was developed based on a comprehensive literature search between April and October 2014. Over 400 publications were reviewed, of which 91 contained suitable drainage nutrient load data for inclusion (dissolved, particulate, and total, N and P loads). Studies suitable for MANAGE must be: peer-reviewed, from study areas of at least 0.009 ha with a single land-use in North America, not be a rainfall simulation or lysimeter study, and include data from at least one year. When necessary, data were extracted from graphs using Data Thief® software. The literature review methods and the development of MANAGE's Drain Load table were fully described by Christianson and Harmel (2015) and MANAGE has previously been described by Harmel et al. (2006) and Harmel et al. (2008).

In the Drain Load table, N application source, timing and placement/method were recorded separately for up to two individual fertilizer formulations for a given site-year. Application timings were grouped into one of four options: "At Planting, within 1 week of plant", "Out of Season, > 2 months before plant", "Pre-Plant, 2 months-1 week before plant", or "Side/Top Dress, > 1 week after plant". Application placements were also grouped into four categories. "Surface applied" included a publication's reference to surface applications and applications specified as broadcast (no incorporation noted); "incorporated" included applications specified as such as well as those that were broadcast incorporated; "injected" included both knifed and injected specifications; "banded" applications were denoted as such. One N application rate (i.e., the summed total of each reported formulation's application rate), was reported for a given site-year.

Nitrogen loads in the Drain Load database were analyzed using graphical methods including box plots and regression analyses. To aid in comparisons particularly with drainage discharge and N application rate, the large dataset was binned into "wet" and "dry" years. The approximate mean (846±219 mm) and median (828 mm) across all precipitation values in the Drain Load table (n= 889) were used as separation points, with precipitation values less than 820 mm or greater than 850 mm considered "dry" or “wet”, respectively. The majority of the data were non-normally distributed, thus were analyzed using Kruskal-Wallis one-way Analysis of Variance tests based on rank which uses median values (Sigma Plot 12.5).

Meta-analysis methods

In a few specific cases, it was possible to analyze 4Rs impacts using meta-analytical methods. By definition, meta-analyses allow quantification of the impact of an experimental treatment relative to a control that is consistently defined across all studies evaluated (Arnqvist and Wooster, 1995; Hedges et al., 1999). The result of each individual study is expressed as a treatment effect size estimator, also called a response ratio, *r;* this is the ratio of the experimental treatment mean ($\overline{X_e}$) to the control treatment mean ($\overline{X_c}$) (Hedges et al., 1999; Johnson and Curtis, 2001):

$$r = \frac{\overline{X_e}}{\overline{X_c}} \qquad \text{Equation 1}$$

These individual study response ratios can be combined across all the studies under investigation to generate an overall mean effect indicating the magnitude of the summarized treatment effect.

Meta-analyses were performed on N application timing and source, the 4Rs strategies for which the most studies were available. The treatment effect estimators were the ratios of the dissolved N load and/or yield of a conventional N application timing or source (i.e., the “control”) to a more recommended timing or alternative source. Specifically, the comparisons made were: (1) at-plant application versus sidedressed and versus pre-plant applications, (2) out-of-season application versus at-plant and versus pre-plant applications, and (3) inorganic fertilizer versus organic sources. The procedural necessity of identifying a control that is defined consistently across all potential studies was a major complicating factor. For example, selection of “out-of-season” N application timing as the control meant that only studies where out-of-season and at least one additional application timing were reported could be used to develop a response ratio (Eq. 1). Despite the nearly 1300 site-years (91 studies) compiled in the Drain Load database, this significantly reduced the number of suitable meta-analysis studies. Continuing with the same example, while there were 85 site-years (13 studies) where N fertilizer was applied “Out of Season, > 2 months before plant”, only four and five studies reported using both out-of-season and either at-plant or pre-plant timing, respectively, in sufficient quantity to develop a mean and standard deviation for each treatment.

Because the sample variances were not reported across all studies suitable for inclusion in the meta-analyses, resampling procedures to develop un-weighted meta-analysis statistics were the best approach (Adams et al., 1997; Gurevitch and Hedges, 1999). A bootstrapping procedure (999 iterations) allowing development of 95% confidence intervals within a random effects model was performed using MetaWin software. Following Johnson and Curtis (2001) and Tonitto et al. (2006), treatment means were considered significantly different from zero if the 95% confidence interval did not overlap zero. Philibert et al. (2012) recommended eight criteria for robust meta-analyses. Five of the eight were observed here (repeatable procedure, references list available, and available dataset, see Christianson and Harmel (2015); use of random effects model; available meta-analysis program). A sensitivity analysis, assessment of publication bias, and data weighting were not performed due to the small number of studies per analysis and data availability.

Results

The MANAGE Drain Load table included 987 and 162 dissolved N and total N site-years from 73 and 7 studies, respectively, from the total 1279 site-years (91 studies). The greatest dissolved N load reported was 245 kg N/ha, with 19 site-years reporting nitrate loads greater than 90 kg N/ha (Baker et al., 1975; Bjorneberg et al., 1996; Gast et al., 1978; Kladivko et al., 2004; Lawlor et al., 2008; Miller, 1979; Nash et al., 2014; Randall et al., 1997; Randall and Iragavarapu, 1995). Site-years reporting dissolved N and total N loads spanned 1969 to 2012 and 1961 to 2005, respectively (Christianson and Harmel, 2015). The majority of dissolved N load site-years were from the US Midwest and Ontario, Canada (Figure 1). Iowa and Illinois contributed 60% of the dissolved N site-years (472 and 123, respectively), though contributions were made from 17 states/provinces. Only five states/provinces presented total N load data with Ontario and North Carolina dominating (90 and 62 site-years, respectively; Figure 1b).

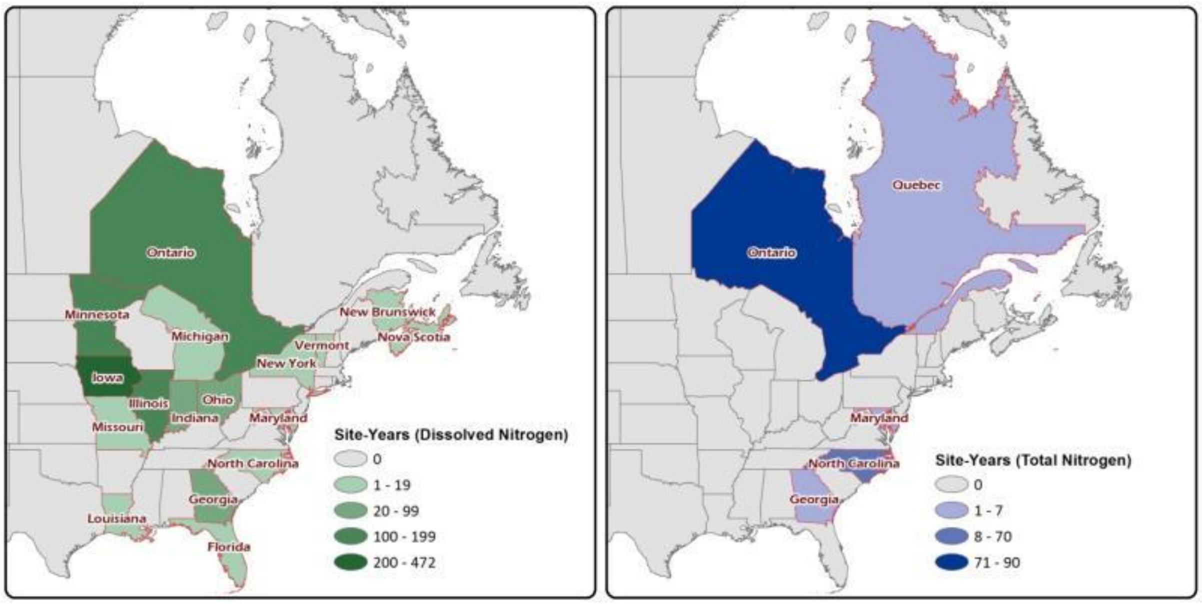

Figure 1: Locations of dissolved (left) and total (right) N load site-years in the Drain Load database

N Application Rate

The prevalence of corn cropping systems in North American agriculture has resulted in corn being one of the largest single users of N fertilizer (Ribaudo et al., 2012). The exact application rate required to maximize profitability at a given site in a given year, however, is highly variable and continues to pose challenges to farmers and scientists (Kladivko et al., 2004; Sawyer et al., 2006). Traditionally recommended application rates are known to impair drainage water quality (Jaynes et al., 2001; Tan et al., 2002). Even when no N is applied (e.g., a “0” control plot or during a soybean year), drainage may contain elevated nitrate concentrations and/or loads (Gupta et al., 2004; Lawes et al., 1882; Lawlor et al., 2008). Fine-tuning N application rates would help reduce “insurance” N applications which, for example, have ranged from 22 to 67 kg N/ha in Minnesota (Legg et al., 1989; Mitsch et al., 2001). While application of the optimal N rate is thought to be the most important and most accessible of the 4Rs N management strategies (Lawlor et al., 2011), optimizing application rates will likely not be sufficient alone to meet water quality goals (Kaspar et al., 2007).

Increasing N fertilization rates correspond with increased drainage nitrate concentrations (Angle et al., 1993; Baker and Johnson, 1981; Bergström and Brink, 1986; Chichester, 1977; Drury et al., 2009; Gast et al., 1978; Hallberg et al., 1986; Helmers et al., 2012; Jaynes and Colvin, 2006; Jaynes et al., 2001; Lawlor et al., 2011; Miller, 1979) and losses (Andraski et al., 2000; Baker and Johnson, 1981; Bergström and Brink, 1986; Bolton et al., 1970; Gast et al., 1978; Guillard et al., 1999; Hallberg et al., 1986; Jaynes et al., 2001; Lawes et al., 1882). Likewise, decreased application rates (in combination with other practices) can reduce nitrate in drainage waters over time (Kladivko et al., 2004). This strong rate effect on drainage N concentrations has been observed regardless of N source (e.g., manure vs. inorganic fertilizer; (Bakhsh et al., 2009; Evans et al., 1984). This rate effect was evident across the Drain Load corn production site-years for both continuous corn and corn-soybean rotations under wet and dry conditions (Figure 2 a and b). At a given N application rate, the wet years produced higher dissolved N loads than the dry for both rotations, with the relationships significant for continuous corn (Table 1; α=0.10). The effects of N fertilization rate may be confounded by precipitation (e.g., dry conditions, rainfall timing relative to application) and drainage flow trends (Guillard et al., 1999; Jaynes, 2013; Zwerman et al., 1972). This variability may explain why some authors have not observed a relationship between N rate and drainage N concentrations or losses (Cambardella et al., 1999; Hanway and Laflen, 1974; Schwab et al., 1980).

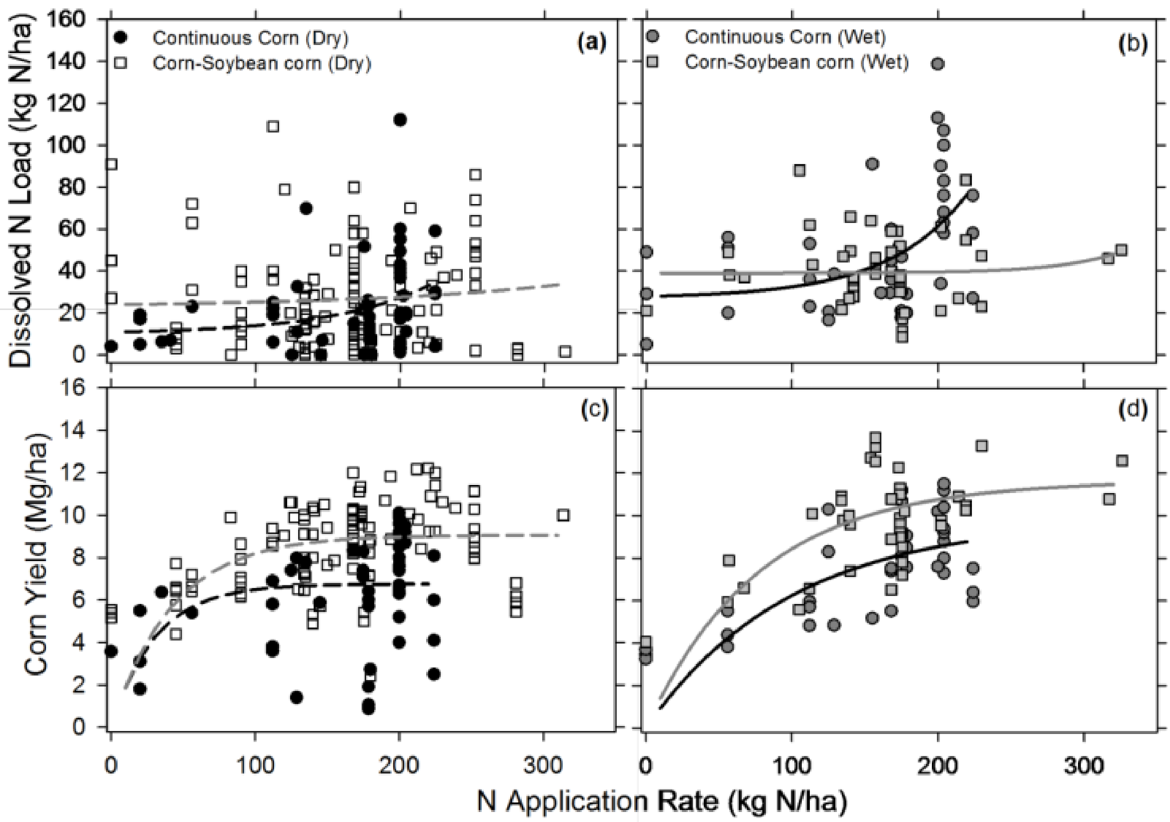

Figure 2: Dissolved N load (a and b) and corn yield (c and d) versus N application rate for continuous corn (black line) or corn in a corn-soybean rotation (gray line) for dry (dashed line; a and c) and wet years (solid line; b and d); three outliers at a 448 kg N/ha application rate were removed; dry: precipitation <820 mm, wet: precipitation >850 mm

Table 1: Dissolved N load regressed against N application rate for four cropping/precipitation combinations from the Drain Load database

		N Load Regressions from Figures 2a and b	R^2	*p*	n	Yield Regressions from Figures 2c and d	R^2	*p*	n
Dry	**Continuous corn**	$y = 10.0 + 0.94e^{0.015x}$	0.076	0.089	69	$y = 6.76 * (1-e^{-0.03x})$	0.075	0.045	59
	Corn-soybean rotation corn	$y = 23.0 + 1.04e^{0.007x}$	0.006	0.691	129	$y = 9.06 * (1-e^{-0.02x})$	0.012	0.224	125
Wet	**Continuous corn**	$y = 27.1 + 0.84e^{0.019x}$	0.245	0.004	43	$y = 9.89 * (1-e^{-0.01x})$	0.336	0.0002	37
	Corn-soybean rotation corn	$y = 39.0 + 0.004e^{0.024x}$	0.011	0.759	61	$y = 11.66 * (1-e^{-0.01x})$	0.331	<0.0001	57

In both wet and dry years, the exponential increasing trend for the N load regressions was much stronger for continuous corn than for the corn-soybean rotation. The relatively more consistent N loads across application rate from corn following soybean resulted in greater N loads than continuous corn at low N application rates and lower loads at higher rates. This change point between the two cropping systems occurred at a lower N application rate in the wet years (approximately 140 versus 200 kg N/ha for wet and dry years, respectively), indicating a special sensitivity to cropping choice and application rate in wetter years. This inflection point for continuous corn in wet years (i.e., approximately 140 kg N/ha applied) roughly corresponded with model simulations from Davis et al. (2000) showing a change point between application rates of 125 and 150 kg N/ha resulted in significantly higher drainage dissolved N loads at the higher application rate (0.9 vs 7.1 kg N/ha, respectively). Nevertheless, comparing the two cropping systems across a given application rate may lead to spurious conclusions, as university recommended application rates are lower for corn following soybeans compared to continuous corn (Sawyer et al., 2006). Following university recommendations, N application rates of 213 and 154 kg N/ha (Corn Nitrogen Rate Calculator at a 0.1 price ratio; Sawyer et al., 2006)) to continuous corn and corn following soybeans, respectively, would result in 31.1 and 26.3 kg N/ha and 70.5 and 39.2 kg N/ha drainage loads in dry and wet years, respectively, using the relationships developed here.

The flatter dissolved N load regressions from the corn-soybean rotation observed in both dry and wet years (albeit, non-significant) indicated this cropping system may be associated with a relatively consistent, quasi-"baseline" load. It is possible that regardless of application rate, in dry and wet years, the corn phase will lose approximately 23 to 28 and 39 kg N/ha, respectively, at application rates below 200 kg N/ha. These corn phase loads were slightly higher than average loads resulting from the soybean phase across the Drain Load dataset (dry years soybean mean: 22 kg N/ha, median: 16 kg N/ha, n=69; wet years soybean mean: 33 kg N/ha, median: 32 kg N/ha; n=42; Christianson and Harmel, 2015). Nevertheless, Christianson and Harmel (2015) showed no significant difference in drainage discharge or dissolved N load between the two phases.

The difference in severity of the regression curves between the two cropping systems may be an indication dissolved N loads from the two should be modeled differently, and may explain some of the difference in previous regression modeling approaches. Hallberg et al. (1986), Baker and Laflen (1983), and the Minnesota Pollution Control Agency (MNPCA, 2013; leaching to groundwater) reported the N load/N application rate relationship was linear, whereas Bergström and Brink (1986) and Guillard et al. (1999) showed clearly exponential correlations. Lawlor et al. (2008) developed an exponential relationship between drainage nitrate concentrations and applied rate but observed no such trend for nitrate losses due to variability in drainage discharge. When the Drain Load regressions (Table 1) were put in context of literature, the high variability in dissolved N loads across site-years was evident. For example at applications of 150 kg N/ha, the regressions predicted a range from 18 kg N/ha loading (continuous corn, dry years, this study) to greater than 50 kg N/ha (Hallberg et al., 1986), although several models predicted very similar loads of 25-27 kg N/ha (corn-soybean rotation, dry years, this study; Bergström and Brink, 1986; Guillard et al., 1999).

While a relationship exists between N application rate and drainage nitrate concentrations/loads, the positive fertilization impact on the yield of corn and other crops is equally well established (Andraski et al., 2000; Chichester, 1977; Helmers et al., 2012; Jaynes et al., 2001; Lawlor et al., 2008; Stevens et al., 2005; Vetsch et al., 1999). This important effect was shown here for both continuous corn and corn-soybean cropping systems in both wet and dry years (Figure 2 c and d). Corn following soybeans had greater yields than continuous corn systems over both wet and dry years at any given N rate. At application rates between 170 and 220 kg N/ha (i.e., the maximum extent of the continuous corn regression), the yield difference between the two cropping systems was consistently very close to 2.25 Mg/ha regardless of wet or dry conditions. While, again, comparing the two systems across the same N application rate may lead to spurious conclusions, even when the lower recommended application rate for corn following soybeans was considered, the rotation corn's yield was still higher than yields from a continuous corn system at its recommended application rate. For example, at university recommended application rates of 154 and 213 kg N/ha for the rotation and continuous corn, respectively (Corn Nitrogen Rate Calculator at a 0.1 price ratio; Sawyer et al., 2006)), the regressed corn grain yields were 8.8 and 6.8 Mg/ha, respectively, in dry years and 10.0 and 8.7, respectively, in wet years. Helmers et al. (2012) and Kanwar et al. (1997) both observed this yield benefit, and recommended a corn-soybean rotation may be the better cropping system of the two for the Midwest.

Percentage of applied N lost in drainage

Across literature, drainage N losses in context of the amount of N applied generally range from just under 10% to roughly 40%, although this can extend much higher (e.g., Gentry et al., 1998; Kanwar et al., 1988; Figure 3). An average of 20% of N applied to corn was lost in drainage in the Drain Load dataset (median: 15%; n=495). Separating these values based precipitation revealed a similar trend between wet and dry years (Figure 4a). The percentage lost in drainage decreased at increasing N application rates for both, with convergence of the trends at high application rates. At lower application rates, a relatively greater percentage of the application would be lost in drainage in a wet versus a dry year, but at higher application rates, the impact of annual precipitation may be overshadowed by excessive applications. Regressing these loss percentages against precipitation seemed to indicate a precipitation change point existed, below which increasing annual precipitation increased the percentage of N lost in drainage, but above which no notable increase occurred (Figure 4b). This may mean that increased annual precipitation poses an increased risk of drainage N loss but only to a point at which a relatively consistent percentage of a given year's application will be lost. Jaynes et al. (1999) issued a caveat about making interpretations too deeply on a percent-of-N-applied basis as: "…we do not wish to imply that fertilizer is the only source of nitrate, nor that the current year's chemical applications contributed to losses during that year."

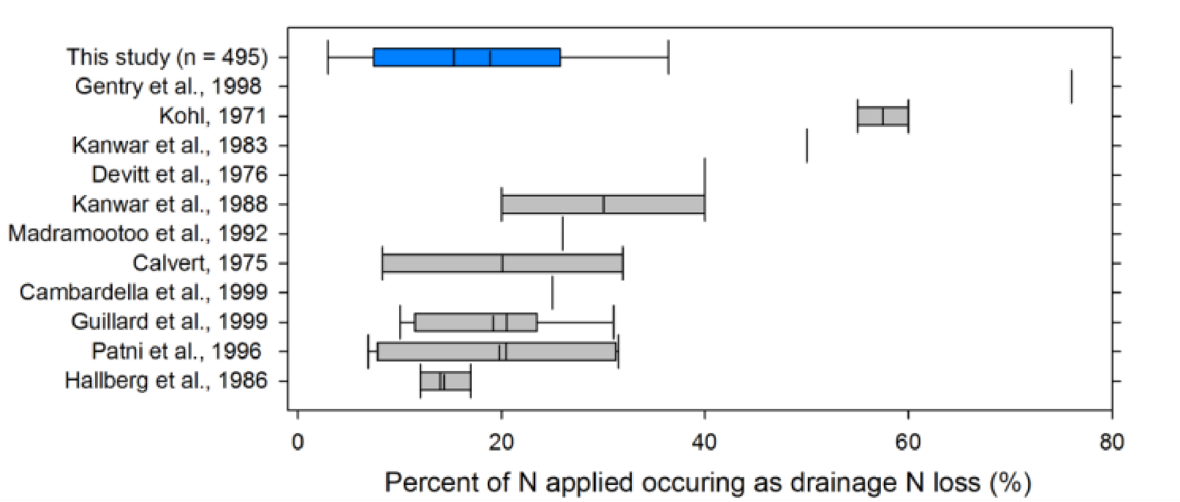

Figure 3: Percent of N applied lost in drainage as reported in literature and from corn site-years in the Drain Load database (n=495)

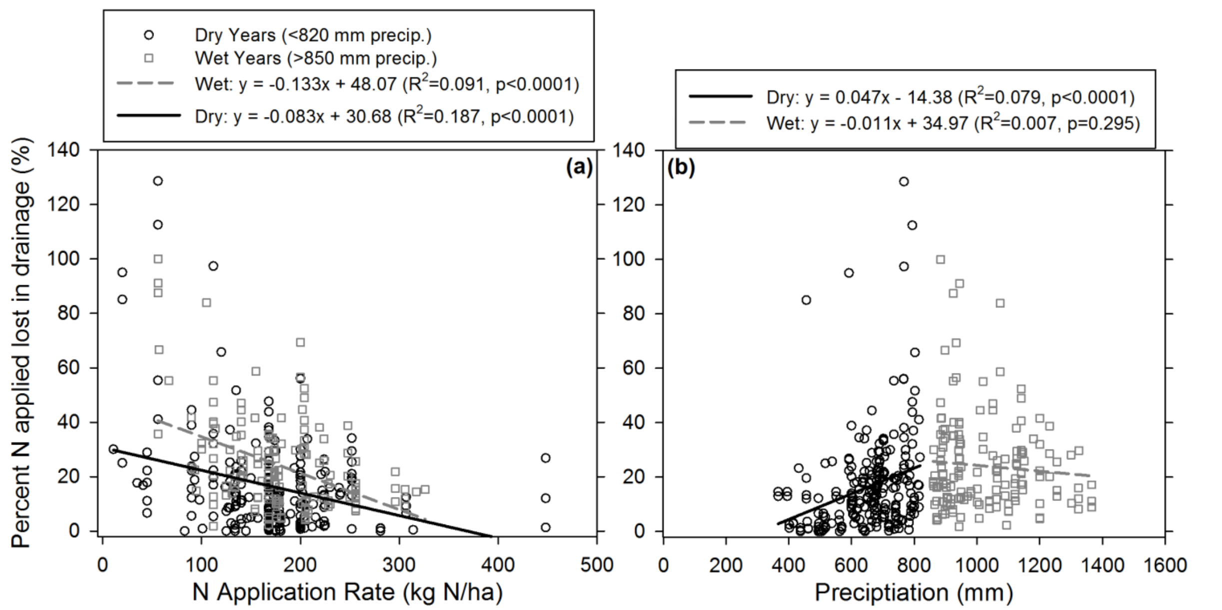

Figure 4: Percentage of a given site-year's annual N application lost in drainage across corn site-years in the Drain Load database shown against N application rate (a) and annual precipitation (b)

N Application Timing

Because multiple fertilizer formulations are often applied at/in a given site-year, the MANAGE framework allowed recording of up to two fertilizer formulations/methods/timings for each site-year (i.e., Fertilizer #1 and #2). The total annual N application rate was entered as one summed value for each record. If a N application rate was reported for a site-year, the timing of Fertilizer #1 was generally at planting or pre-plant (31 and 36% of Fertilizer #1's site-years, respectively; n=530; Figure 5a), whereas if a second fertilizer was reported, it was, not surprisingly, often a side-dressed application (50% of Fertilizer #2; n=210; Figure 5a). The most common source/timing combinations for corn were UAN applied at-planting, pre-plant anhydrous ammonia, pre-plant ammonium nitrate, and out-of-season liquid swine manure (Figure 5b).

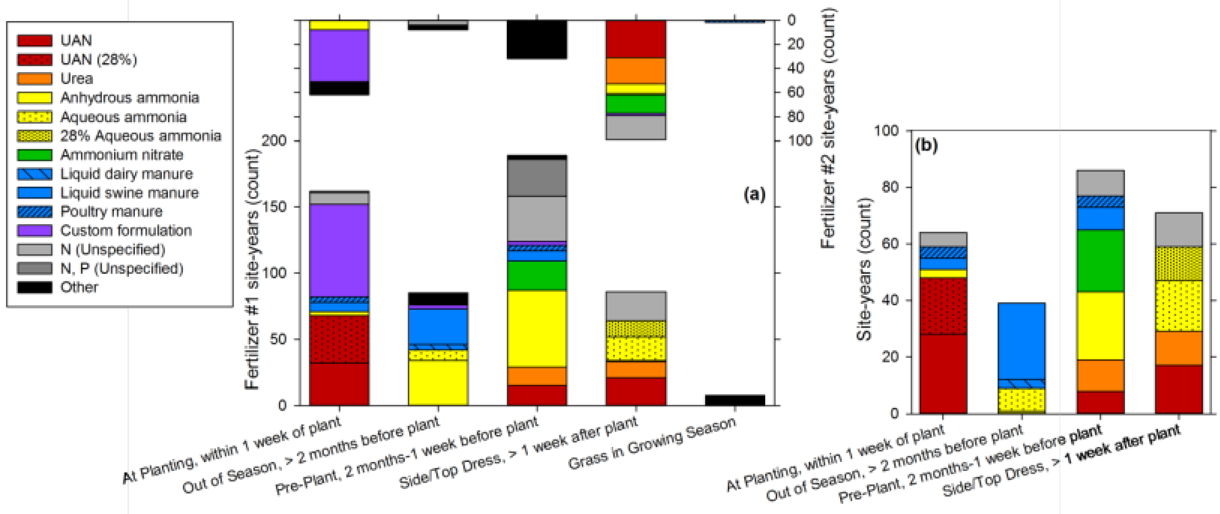

Figure 5: Nitrogen application timing by product for Drain Load Fertilizer #1 site-years (a, bottom), Fertilizer #2 site-years (a, top), and when only one application was reported in a corn site-year (b)

Across many watersheds, fall application is more common than spring pre-plant (e.g., 51% vs 35%, respectively, in the Lake Bloomington watershed; Smiciklas and Moore, 1999), and increased farm size, a widespread trend, may result in the increasing prevalence of fall application (Gentry et al., 2000). Nevertheless, spring application of N is generally recommended to reduce N leaching losses and to improve profitability (Malone et al., 2007; Randall et al., 2003b; Vetsch and Randall, 2004). Studies from Minnesota showed a 13% - 14% reduction in drainage nitrate loss for spring pre-plant anhydrous ammonia application compared to fall and found that yields and corn N uptake were higher from spring application (Randall and Vetsch, 2005a; Randall and Vetsch, 2005b; Randall et al., 2003a; Randall et al., 2003b). Nevertheless, it is possible drain N load impacts of changing applications from the fall to the spring will be minimal compared to the effect of annual precipitation, as some studies have observed no difference in nitrate concentration or grain yield when the same N rate is applied in the fall and spring (Lawlor et al., 2011).

Analysis of fertilizer timings (only corn site-years where only one fertilizer was reported to avoid confounding effects of Fertilizer #1 vs #2) showed out-of-season fall N applications were not statistically different from pre-preplant applications in terms of application rate, corn yield, or dissolved N load (Table 2). The lack of significant difference in drainage dissolved N load was also echoed in the meta-analysis (Figure 6). The meta-analysis was confounded by the lack of suitable studies (i.e., a given study had to include at least two application timings), and these small population sizes may have contributed to all the confidence intervals overlapping each other and overlapping zero. While there was no significant difference in dissolved N load between the four application timings (Table 2), the out-of-season and pre-plant approaches had the highest median loads and the at-planting and side/top dress medians were the lowest. This is generally consistent with conventional water quality assumptions. However across the database, the latter two treatments also received significantly lower N application rates.

Table 2: Median (count) N application rate, corn yield, and dissolved nitrogen load by N application timing for corn site-years where only one application was reported in the Drain Load database; medians with the same letters are not statistically significantly different

	N Application Rate kg N/ha	Corn Yield Mg/ha	Dissolved Nitrogen Load* kg N/ha
Out of Season, > 2 months before plant	168 (39) ab	8.7 (34) ab	29.5 (39)
Pre-Plant, 2 months-1 week before plant	200 (86) a	9.4 (69) a	30.5 (86)
At Planting, within 1 week of plant	152 (64) b	7.5 (56) b	27.0 (63)
Side/Top Dress, > 1 week after plant	160 (71) b	8.4 (62) b	27.9 (71)

* No significant difference between treatments (*p*=0.934)

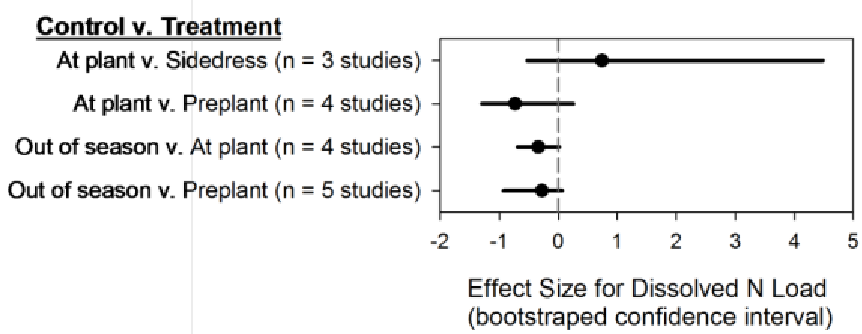

Figure 6: Meta-analyses dissolved N load effect sizes and confidence intervals by nitrogen application timing treatment; negative effect size indicates the control resulted in higher dissolved N loads (i.e. positive effect size shows a positive or increased contribution to dissolved N loads by the treatment)

Split N applications are thought to reduce N leaching and increase plant uptake and/or yield by synchronizing application with plant growth (Kanwar et al., 1988; Waddell et al., 2000). Bakhsh et al. (2002) reported a 25% reduction in nitrate leaching and a 13% corn grain yield increase with a sidedressed N application versus a single application. Economically, Randall et al. (2003a) determined the highest profitability in their study resulted from a split N application and the lowest from a fall application without the use of a nitrification inhibitor ($239.40/ha-yr versus $166.70/ha-yr, respectively). The water quality benefits of side-dressing may be variable as Tan et al. (2002) observed a spike in tile drainage nitrate concentrations coincident with the timing of such an application (at the 6-leaf stage). Several studies have noted a yield benefit of split N application but no significant difference in drainage nitrate loads (Bjorneberg et al., 1998; Jaynes and Colvin, 2006). Jaynes and Colvin (2006) went as far as to note the “reactive strategy” (Jaynes, 2013) of split N applications should not be considered a water quality improvement practice. Several investigations showed no consistent, significant positive yield impact of a split approach (Guillard et al., 1999; Jaynes, 2013) or increased yield risk in years where it is wet after the sidedress application (Karlen et al., 2005). Analysis of the Drain Load database revealed no significant water quality or corn yield benefit of a side-dressed application (Table 2).

N Application Method

The most predominant N application method across Drain Load site-years was injection (256 of 394 Fertilizer #1 site-years; 18 of 86 Fertilizer #2 site-years; Figure 7a). In records where more than one fertilizer was applied, surface application was the predominant method for the second application (50 of 86 site-years for Fertilizer #2; Figure 7a) likely due to its occurrence later in the season (i.e., most second applications were side-dressed; Figure 5a). Nitrogen application placement is not a widely studied topic in the drainage water quality literature. For example, across the Drain Load database, there were approximately twenty studies where nutrient application timing or source could be compared within the study (i.e., more than one timing or source were used). Contrastingly, only six studies specifically reported more than one application method.

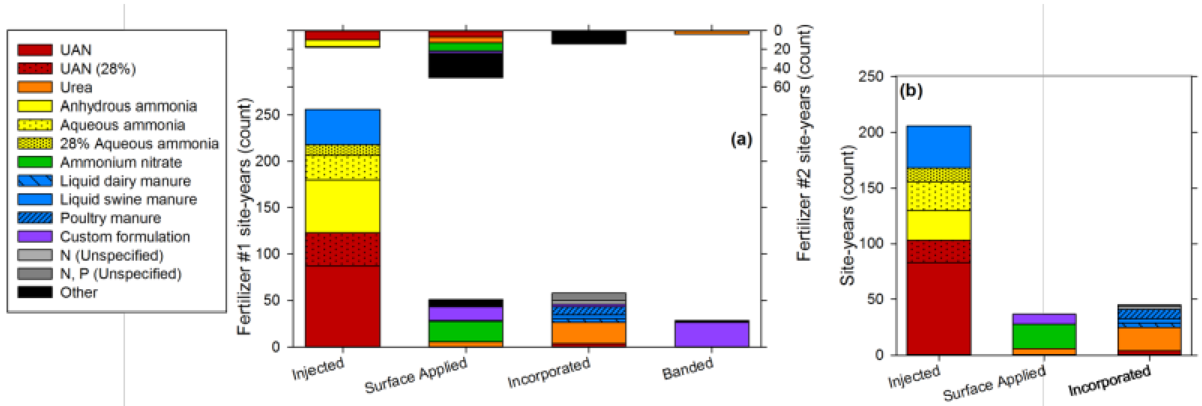

Figure 7: Nitrogen application method by product source for Drain Load Fertilizer #1 site-years (a, bottom); Fertilizer #2 site-years (a, top), and when one application was reported in a corn site-year (b)

When sorting similar to the N fertilizer timing analysis was done (i.e., data were sorted to exclude site-years using more than one application method to ensure the entire application was applied using one method), injected UAN was the most prevalent N source/method combination for corn-site-years followed by injected ammonia products and injected liquid swine manure (Figure 7b;). From this selected dataset, application rates for surface-based N applications were significantly greater than the rate when injected (Table 3). However, these data were skewed by one study with a relatively high site-year count (Randall and Iragavarapu, 1995); when this study was removed, there was no significant difference in application rates between methods, although the population became relatively small (surface applied median: 156 kg N/ha applied, n=15; *p* = 0.070). Injection and incorporation resulted in the highest yields. Unfortunately, the predominance of injection (e.g., Figure 7) does not bode well for drainage water quality, as this method had the highest dissolved N loads (not significant). Physically, it is plausible that injecting N sources may result in higher subsurface drainage N loads as surface applied or surface incorporated methods place the product further from the tile.

Table 3: Median (count) N application rate, corn yield, and dissolved nitrogen load by N application method for corn site-years where only one application was reported in the Drain Load database; medians with the same letters are not statistically significantly different based on a Kruskal-Wallis One Way Analysis of Variance on Ranks

	N Application Rate	**Corn Yield**	**Dissolved Nitrogen Load***
	kg N/ha	**Mg/ha**	**kg N/ha**
Incorporated	168 (45) ab	9.4 (36) a	19.6 (45)
Injected	167 (206) b	8.4 (198) a	28.0 (206)
Surface Applied	200 (37) a	6.7 (31) b	20.0 (37)

* No significant difference between treatments (*p*=0.916)

N Source

There was no predominant N source when all Drain Load site-years were evaluated (Figure 8). Nitrogen application rate was reported in 784 site-years, within which a Fertilizer #1 was reported 746 times with a site-year having a second application in 242 records. Across both Fertilizer #1 and #2, the most prevalent sources were unspecified N sources, custom formulations, UAN, and anhydrous ammonia (Figure 8).

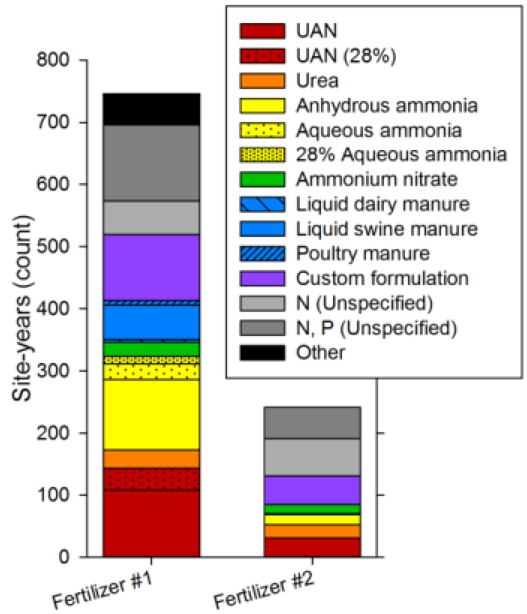

Figure 8: Nitrogen application sources for Drain Load database Fertilizer #1 and Fertilizer #2 site-years

Several studies report a yield boost due to organic fertilizers compared to inorganic N (Lawlor et al., 2011; Malone et al., 2007; Thoma et al., 2005), but this is not exclusively the case (Randall et al., 2000). There was a significant positive corn yield impact due to organic (manure or litter) versus inorganic applications in the Drain Load database (Table 4). Some studies suggest organic N sources may provide a drainage water quality benefit (Bakhsh et al., 2007; Kimble et al., 1972; Malone et al., 2007; Nguyen et al., 2013; Thoma et al., 2005), but here, there was no significant difference in dissolved N loads between the sources. Nevertheless, the mean and median dissolved N loads were lower for organic compared to inorganic sources (Table 4; mean/median of 24.2/18.4 and 28.4/23.6 kg N/ha, respectively). Organic applications resulted in significantly higher total N loads in drainage (Table 4), but there were only seven site-years upon which this comparison was based.

Table 4: Median (count) of Drain Load database precipitation, drainage discharge, corn yield and dissolved and total N loads by nutrient source

	Precipitation	Drain discharge	Corn Yield	Dissolved N Load	Total N Load
	---------------- mm	----------------	Mg/ha	-------------- kg N/ha	--------------
Inorganic fertilizer	797 (n=505)	165 (n=542)	8.1 (n=322)	23.6 (n=571)	14.0 (n=30)
Organic N Sources	885 (n=86)*	185 (n=93)	10.4 (n=55)*	18.4 (n=95)	15.5 (n=7)**

* indicates significant difference between sources at α=0.05

** indicates significant difference between sources at α=0.10

The meta-analysis echoed these results by showing no difference between dissolved N loads from organic versus inorganic sources (i.e., confidence interval overlapped zero; Figure 9). However, the meta-analysis, which was based on a smaller dataset than the general comparisons in Table 4, indicated there was no difference in crop yield between the two grouped sources. There were no drainage total N loads reported in studies suitable for the meta-analysis.

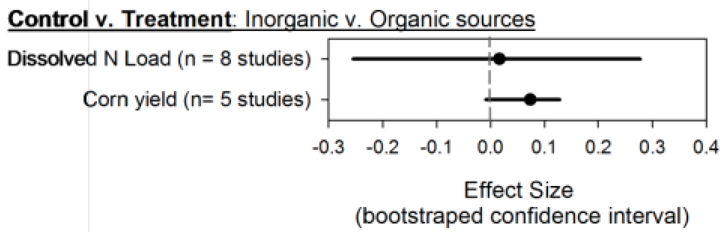

Figure 9: Meta-analyses effect sizes and confidence intervals for inorganic versus organic nutrient applications; positive effect size meant the treatment (organic sources) showed an increase in the parameter over the control (inorganic sources)

Despite potential benefits of application of organic forms of nutrients (including contribution to soil carbon pools), there are several caveats. Firstly, it is difficultly to exactly compare application rates between organic and inorganic N sources because the “available” N in manure or litter is highly variable and requires intensive sampling, analysis, and application procedures for uniformity in application. Moreover, some comparisons do not compensate for the macro- and micronutrients provided with the manure. Secondly, injection of liquid manures can pose a direct water quality threat as there are several reports of visible contamination of drainage waters immediately following liquid manure injection (Ball Coelho et al., 2012; Ball Coelho et al., 2007; Burchell et al., 2005). Lastly, use of organic N sources requires improved management strategies just as inorganic sources do. Mitsch et al. (2001) reported that improved manure management could save 500 x 10^3 MT total N per year transported through the Mississippi River basin.

Advanced N products

Nitrification inhibitors are generally thought to increase N use efficiency and crop yield and reduce greenhouse gas emissions from fertilizer (Decock, 2014; Ladha et al., 2005). On the water quality-front, Randall et al. (2003b) and Randall and Vetsch (2005b) reported a 10 to 18% reduction in drainage N losses due to use of Nitrapyrin with fall anhydrous ammonia application compared to a fall application without the inhibitor. Smiciklas and Moore (1999) similarly observed a 9% reduction in drainage nitrate concentration due to use of a fall applied N inhibitor. Ladha et al. (2005) summarized that N inhibitors have a low benefit to cost ratio and low profitability. Across the Drain Load database, N inhibitors accounted for 34 site-years from only one study (i.e., Kladivko et al., 2004). Polymer coated urea, or slow-release products, show potential to reduce N leaching in leachate and drainage (Mikkelsen et al., 1994; Waddell et al., 2000; Wang and Alva, 1996; Zvomuya et al., 2003), although they may not provide significant agronomic N use efficiency or yield benefits under all climatic types or years (Nelson et al., 2009). There was only one polymer coated urea study suitable for inclusion in the MANAGE Drain Load database, thus no advanced comparisons could be made (Nash et al., 2014).

Conclusions

Nitrogen, an element essential for life, is particularly vexing as its ubiquity and mutable nature confounds attempts to manage it agronomically. The statistical significance of some of the 4Rs practices for reduction of drainage dissolved N loads was stronger than for others. Optimizing N application rates will continue to receive primary research and regulatory focus. Across site-years, wetter conditions resulted in greater dissolved N losses from corn-based systems, although at very high N application rates, the impact of annual precipitation may be overshadowed by excessive applications (e.g., Figure 6). Corn-soybean rotations resulted in lower N loads when compared with continuous corn across university recommended application rates, and greater yields compared across all rates. "Fine-tuning" N rates is clearly important, but it would be short-sighted and unrealistic to focus solely on this practice.

Use of organic N sources could boost corn yields with potentially no increase in dissolved N loads compared to inorganic N fertilizer. However, this approach may pose a risk for increased loss of total N, and adherence to 4Rs strategies is vital regardless of the nutrient source (i.e., organic v. inorganic). The lack of significant differences between N application timing or application method treatments indicated neither should receive primary focus as dissolved N load reduction strategies. The application timing analysis was complicated by differences in application rates between treatments; the highest application rates resulted in the greatest N losses and yields. Nevertheless, the typically recommended practices such as applying at-planting or side-dressing had lowest median N losses (not significant). Similarly, there was no significant difference between application methods. Injection of N sources, the most widely used method across studies, posed a N loss risk over incorporation and surface application based on median values (not significant). Broad-scale analyses such as this can help identify major trends for water quality, but accurate implementation of the 4Rs approach will require site-specific knowledge to balance agronomic and environmental goals.

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A quantitative review and synthesis of fifty years of drainage phosphorus losses

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Abstract

The prevalence of drainage systems in intensively cropped areas across North America overlain with the importance of freshwater resources in these regions has created a critical intersection where understanding drainage phosphorus transport is vital. This work reviewed and quantitatively analyzed nearly 1300 site-years of drainage nutrient load data to develop a more comprehensive understanding of the phosphorus loading and crop yield impacts of agronomic management practices within drained landscapes. Using the new Drain Load table in the existing "Measured Annual Nutrient loads from AGricultural Environments" (MANAGE) database, factors such as soil test P, tillage, and nutrient management were analyzed. Across site-years, less than 2% of applied P was lost in drainage which corroborates order of magnitude difference between agronomic P application rates and P loadings that can cause ecological damage. The practice of no-till significantly increased drainage dissolved P loads but also improved corn yields compared to conventional tillage (0.12 v. 0.04 kg P/ha; 8.7 v. 8.1 Mg/ha, non-significant). The timing of P application and the method of application were both thought to be important for drainage P losses, but these conclusions could not be verified due to low site-year counts. This work serves as a call to increase the number of field-scale studies documenting not only drainage P losses, but also important cropping management, nutrient application, soil property, and drainage design impacts on such losses.

Introduction

Phosphorus (P), as an integral component in nucleic acids and for cellular energy production, is an element essential for life (Correll, 1998; Sharpley and Menzel, 1987). While a low level of P is desirable for primary production in lakes and rivers (Correll, 1998), such low levels required for natural cycling result in waters that can easily become impaired by additional P inputs. In terms of agricultural impacts on water quality, literature historically indicates surface runoff poses a greater risk of both sediment and P transport than subsurface agricultural drainage due to additional soil/water contact (Algoazany et al., 2007; Eastman et al., 2010; Schwab et al., 1973; Sharpley and Syers, 1979; Sharpley and Withers, 1994; Sims et al., 1998; Skaggs et al., 1994). Artificial subsurface drainage has even been proposed as a management practice to reduce P transport from agricultural fields (Ball Coelho et al., 2012; Bengtson et al., 1995; Blann et al., 2009; Bottcher et al., 1981; Eastman et al., 2010). However, as major P-related water impairments continue to generate headlines and stir regulatory interest (Moore et al., 2011; Scavia et al., 2014), there is an increasing need to better quantify the potential contribution of P loads transported via agricultural drainage and its potential to be a critical pathway (Ball Coelho et al., 2012; Gächter et al., 1998; McDowell et al., 2001; Ruark et al., 2012).

Orthophosphate (often referred to interchangeably as dissolved P or soluble reactive P) and organic P are the most prevalent forms of P in the environment (Ryden et al., 1974). Orthophosphate is the predominant form that can be uptaken by autotrophs such as algae and cyanobacteria, the growth, death, and decomposition of which exacerbates a feedback loop in aquatic systems where dissolved oxygen is consumed and additional P may be released from aquatic sediments (Correll, 1998; Sharpley and Menzel, 1987). Particulate- or sediment-bound P, generally defined as being larger than 45 µm (Ryden et al., 1974; Simard et al., 2000), refers to P that has been strongly sorbed by soil clays or organic particles (Sharpley and Menzel, 1987). Through generally not considered to be "reactive P", a certain percentage of particulate P is bioavailable (approximately 20%, ranging widely from 10-90%; Hansen et al., 2002; Sharpley and Menzel, 1987). Variability in availability and P release dynamics mean that total P, as opposed to soluble or reactive P, is the more ecologically relevant parameter (Correll, 1998; Vidon and Cuadra, 2011). In-stream and in-lake total P concentrations of less than 25 to 30 µg TP/L are recommended to avoid algal growth (Dodds et al., 1997; Fleming, 1990), although some suggest as low at 10 to 20 µg TP/L (Correll, 1998; Daniel et al., 1998). The United States Environmental Protection Agency has also issued a recommended value of 0.10 mg TP/L, though this is not a regulatory criterion (USEPA, 1986).

The form and extent of tile drainage P losses depends upon the interaction of factors including climate, soil, hydrology, land management, P application strategies, and drainage design (Culley et al., 1983; Dils and Heathwaite, 1999; King et al., 2014; Sims et al., 1998; Skaggs et al., 1994). Of primary concern for subsurface P leaching are sites: prone to preferential flow (e.g., deep sandy soils or soils with macropores), with high organic matter soils, and with historically high P applications and soil P concentrations (Blann et al., 2009; Hansen et al., 2002; Miller, 1979; Sims et al., 1998). Soils containing iron ores or aluminum with high P fixing potential may have minimal P leaching losses (Hansen et al., 2002; Sims et al., 1998).The importance of soil type and properties means the site-specificity of the potential for P transport in drainage is extremely important (Sims et al., 1998).

Dissolved and total P concentrations in drainage studies have peaked at as high as 85.7 and 9.7 mg P/L, respectively (Duxbury and Peverly, 1978; Fleming, 1990; Miller, 1979; Owens and Shipitalo, 2006; Sallade and Sims, 1997). Some of these values were strongly influenced by spatial or temporal effects (i.e., 9.7 mg TP/L near a milk house, Fleming (1990); 6.14 mg TP/L under high-flow conditions in spring, Sallade and Sims (1997); event-based, following a fertilizer application: 85.7 mg TDRP /L, Owens and Shipitalo (2006)). However, even many reported average P concentrations are above the critical limits known to impair freshwaters (Ahiablame et al., 2011; Sallade and Sims, 1997; Xue et al., 1998). Reported high total P loads from drainage systems include 250 g/ha/dy (Smith et al., 1995), 26 kg P/ha (Kleinman et al., 2007), and 36.8 kg P/ha (Miller, 1979).

Human and environmental health concerns associated with P-related algal blooms in freshwater resources necessitate increasing attention must be paid to mitigating drainage P loads (e.g., the Lake Erie Toxic bloom in summer 2014). There is a need to assemble and further analyze field-scale drainage P studies conducted over the past fifty years to enhance understanding across drained landscapes. Recently, information from nearly 1300 drainage nutrient study site-years was compiled in a free, publically available database. This new Drain Load table in the existing “Measured Annual Nutrient loads from AGricultural Environments” (MANAGE) database provides comprehensive P load data from peer-reviewed studies across North America. MANAGE is hosted by the United States Department of Agriculture, Agricultural Research Service, Grassland, Soil, and Water Research Laboratory in Temple, Texas (www.ars.usda.gov/spa/manage-nutrient; Harmel et al., 2006; Harmel et al., 2008). This update to MANAGE complemented the more than 1800 existing agricultural and forest runoff watershed-years in this database. Here, the drainage data pooled in the new MANAGE Drain Load table was used to better identify and define the water quality and yield consequences of P management strategies. Special emphasis was placed upon better understanding the impacts of the 4Rs approach to nutrient management (i.e., applying the right nutrient source at the right rate, right time, and right place).

Methods

A literature review encompassing over 400 publications focused on artificial agricultural drainage nutrient loads was conducted between April and October 2014. Ninety-one peer-reviewed publications were deemed suitable for inclusion in the MANAGE Drain Load table. Suitable studies must be: peer-reviewed, from study areas of at least 0.009 ha with a single land-use in North America, not be a rainfall simulation or lysimeter study, and include data from at least one year. In general, information on each site-year's location, drainage and cropping system, nutrient application, yield, precipitation, and citation were sourced. Data Thief® software was used to extract information from figures and graphs when necessary. Development of MANAGE's Drain Load table has previously been described (Christianson and Harmel, 2015), as have the nitrogen load data (Christianson and Harmel, 2015).

Within the database, nutrient application source, timing and method were recorded separately for up to two individual formulations for a given site-year. Application timings were grouped into one of four options: "At Planting, within 1 week of plant", "Out of Season, > 2 months before plant", "Pre-Plant, 2 months-1 week before plant", or "Side/Top Dress, > 1 week after plant". Application placements were also grouped: "Surface applied" (including broadcast); "incorporated" (including broadcast incorporated); "injected" (including knifed and injected); and "banded". One P application rate (i.e., the summed total of each reported formulation's application rates), was reported for a given site-year. Application rates were all normalized to kg P/ha (i.e., not P_2O_5). Standard statistical approaches such as box plots and regression analyses were used. The majority of the data were non-normally distributed, thus were analyzed using Kruskal-Wallis one-way Analysis of Variance tests based on rank which uses median values (Sigma Plot 12.5).

Results

The MANAGE Drain Load table included 225, 50, and 242 dissolved, particulate, and total P site-years from 26, 4, and 19 studies, respectively, from the complete set of 1279 site-years (91 studies). Because there were relatively few P loads reported from surface drainage studies, this analysis focused on subsurface drainage. Nevertheless, a general comparison showed surface drainage site-years reported significantly greater P loads than subsurface (Figure 1).

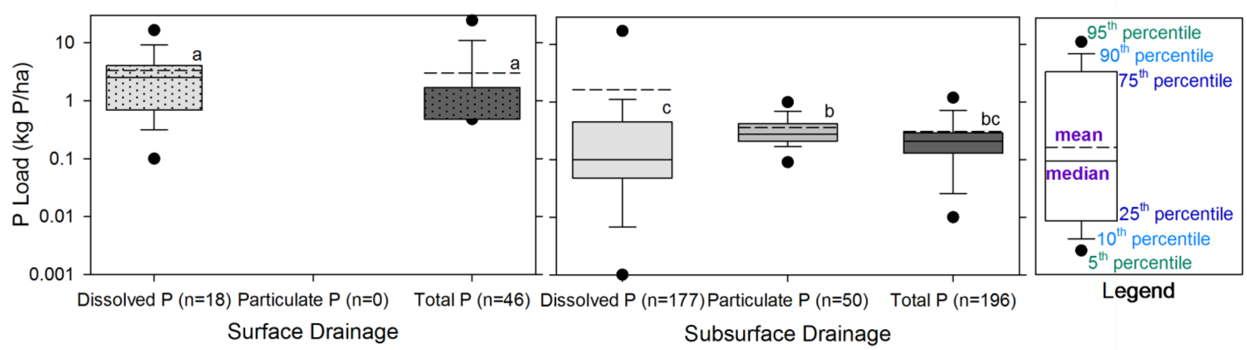

Figure 1: Range of surface and subsurface drainage P loads with legend defining box plot symbols

The greatest load reported across the database was 36.8 kg dissolved P/ha (Miller, 1979), with 15 site-years reporting loads greater than 10 kg P/ha (Miller, 1979- subsurface; Kleinman et al., 2007 - surface; Duxbury and Peverly, 1978 - subsurface). Site-years reporting dissolved, particulate, and total P loads spanned 1968 to 2010, 1976 to 2004, and 1961 to 2009, respectively. The majority of P load site-years were from the Ontario, which contributed 33 and 41% of dissolved and total P site-years, respectively (Figure 2). Sims et al. (1998) identified states with high P loss risk due to high soil test P levels included: Wisconsin, Delaware, New Jersey, South Carolina, Idaho, Washington, Oregon, California, Illinois, Indiana, Michigan, Maryland, North Carolina, Florida, and Louisiana, the latter seven of which emerged here. Regardless, agricultural P loadings present problems at several scales, because within a P-impaired watershed, there will be areas of both P limitation and surplus (Daniel et al., 1998).

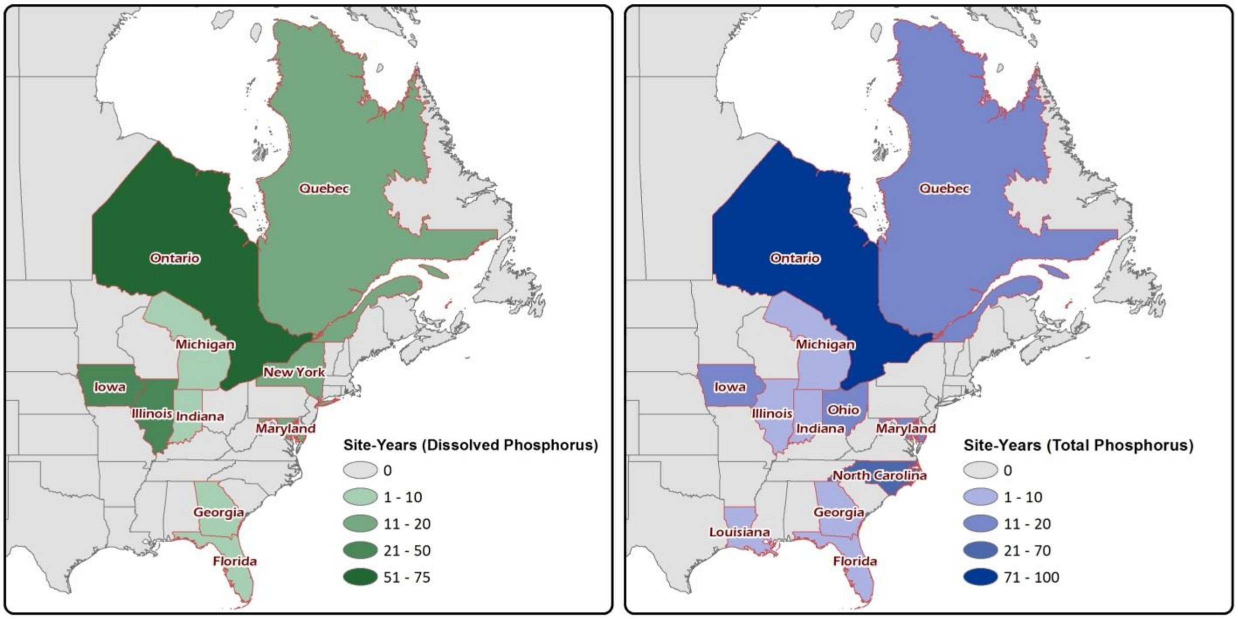

Figure 2: Locations of dissolved (left) and total (right) P load site-years in the Drain Load database

P Fractionation

The fractionation between P forms in drainage can be site specific with factors such as flow, source of drainage (i.e., snowmelt, rainfall, irrigation-based), temporal effects, soil type, and management all playing a role. Generally, the P form is largely a function of the contact time or travel distance between infiltrating water and the soil (Addiscott et al., 2000; Sharpley and Syers, 1979), and the fractions of P can change during transport through the highly complex soil ecosystem (McDowell et al., 2001). Historically, more emphasis has been placed upon documenting soluble P loads in drainage compared to particulate, although they are both clearly related to the total load (Figure 3; dissolved n = 68, particulate n = 15). Based on strongly correlated regressions, 86% of the total P load could be contributed to sediment-bound P when both values were reported in a given site-year, whereas 40% of the total load was due to soluble forms when both soluble and total P loads were reported. When five dissolved P outliers were removed from analysis (where total P was greater than 5 kg P/ha), the soluble P regression slope increased to indicate 66% of the total load was in the soluble form ($y = 0.66x - 0.14$, R^2=0.81).

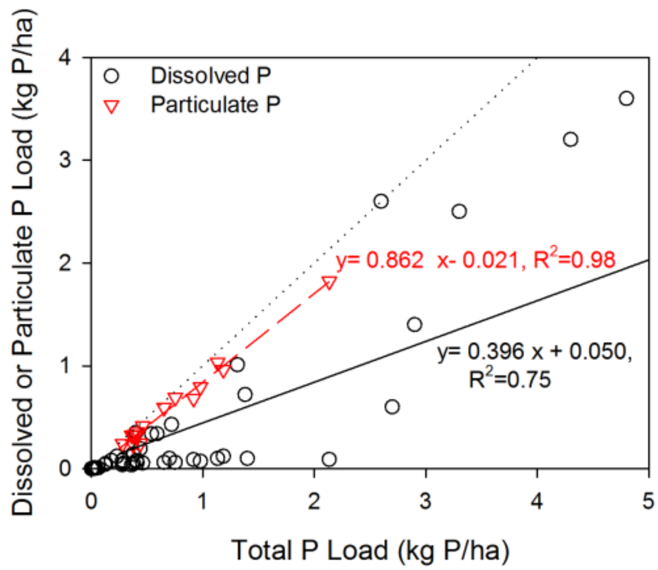

Figure 3: Relationship between soluble or particulate P and total P loads in the Drain Load database; 1:1 relationship shown with dotted line; five outliers on the dissolved P regression with total P loads exceeding 5 kg P/ha not shown

While many have observed particulate P to be the dominant form in drainage, often in context of soils with the potential for large macropore-based transport (Beauchemin et al., 1998; Bottcher et al., 1981; Carefoot and Whalen, 2003; de Jonge et al., 2004; Eastman et al., 2010; Goulet et al., 2006; Grant et al., 1996; Heathwaite and Dils, 2000; Schwab et al., 1980; Schwab et al., 1973; Uusitalo et al., 2001), there has nevertheless been nearly equal weight placed upon the predominance of dissolved forms of P in drainage (Djodjic et al., 2002; Gaynor and Findlay, 1995; Gold and Loudon, 1989; Haygarth et al., 1998; Heckrath et al., 1995; Hergert et al., 1981b; Kleinman et al., 2007; Sanchez Valero et al., 2007; Sharpley and Syers, 1979)). Preferential flow paths will be important P conduits to many subsurface drainage systems, but matrix flow is not to be ignored. McDowell and Sharpley (2001a) attributed increased drainage dissolved P concentrations a year after manure applications to slow transport via matrix flow paths, though Simard et al. (2000) reported matrix flow may originate near drain lines. Most generally, it is thought that movement of total P loads will predominantly occur via preferential flow paths, while dissolved P loads will be a combination of matrix and preferential flow (Vidon and Cuadra, 2011).

Hydrology: Annual, seasonal, and storm-based trends

Neither annual precipitation nor drainage discharge values collected in MANAGE's Drain Load database were strongly correlated with dissolved, particulate, or total P loads indicating that drainage P loads were less dependent upon yearly cumulative hydrology factors than N loads (Figure 4). Within a given year, there is a strong seasonality component to drainage P losses with transport correlated with periods of high drainage flow (e.g., the very early growing season or the non-growing season; Ball Coelho et al., 2012; Duxbury and Peverly, 1978; Longabucco and Rafferty, 1989; Macrae et al., 2007; Owens and Shipitalo, 2006). High flows (e.g., peak flow conditions; storm flows) have been correlated with high P concentrations in drainage waters (Dils and Heathwaite, 1999; Duxbury and Peverly, 1978). These high flow rate P-transporting events are primarily associated with macropore flow (Macrae et al., 2007; Vidon and Cuadra, 2011), and such, preferential flow paths have been identified as a very important potential conduit for P (Scott et al., 1998; Simard et al., 2000; Stamm et al., 1998; Vidon and Cuadra, 2011; Wesström and Messing, 2007). The effect of antecedent soil moisture is important

(Macrae et al., 2007). Dissolved P concentrations may increase during drier summer months (Watson et al., 2007) and these forms may dominate over particulate under base flow conditions, where low water velocities may lack the energy to discharge particulate P. Conversely, the particulate phase may dominate in the spring's high flow events (Dils and Heathwaite, 1999; Gentry et al., 2007). Under high-flow conditions, P concentrations and sediment follow a hydrograph trend, with increasing concentrations during the rising stage, peak concentrations at or before peak flows, and lower concentrations during the falling stage or as drainage cedes (Grant et al., 1996; Hergert et al., 1981a; Kinley et al., 2007; Schelde et al., 2006; Sharpley et al., 1976; Stamm et al., 1998). While storm discharge flow rates have been correlated positively with both particulate and dissolved P concentrations (Djodjic et al., 2000; Gächter et al., 1998; Gentry et al., 2007), the extent of soluble P contribution to total P loads may vary with the size of the precipitation event and extent of macropore flow (Vidon and Cuadra, 2011). Because MANAGE is aggregated at the annual time step, the values compiled in the Drain Load database may not be the most useful to evaluate temporal effects on drainage P transport which appears to be event driven (although the source data from individual studies could be useful for this purpose).

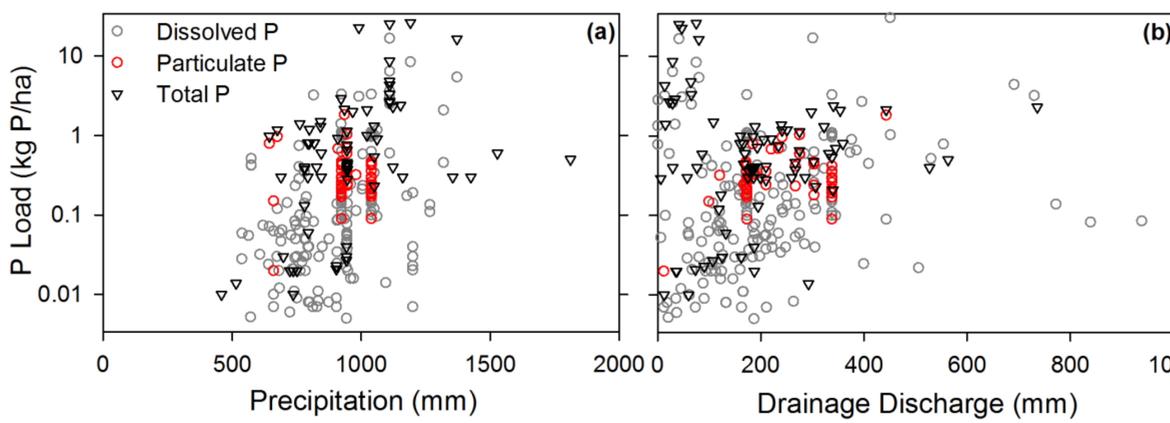

Figure 4: Dissolved, particulate, and total P loads versus annual precipitation (a) and drain discharge (b); four high discharge outliers removed from (b)

In addition to flow velocity and soil moisture impacts, the source of drain flow can influence P fractionation. For example, snowmelt drainage may play a significant role in the transport of dissolved P (Jamieson et al., 2003; Klatt et al., 2003; Macrae et al., 2007). Ginting et al. (2000) reported the majority of P in snowmelt drainage was soluble, whereas rainfall-associated water entering surface inlets contained relatively less P in the dissolved form (18-33% of total P). Ponding of snowmelt drainage near surface intakes will allow P to come into solution from settled sediment-associated P (Ginting et al., 2000). While dilution of nutrient concentrations due to snowmelt is possible (Hawkins and Scholefield, 1996), and several report lower P concentrations in winter drainage compared to the rest of the year, the increased discharge volume compared to rainfall-induced drainage may result in relatively greater snowmelt-associated drainage P loads (Ball Coelho et al., 2012; Jamieson et al., 2003). To further complicate this type of drainage, rain falling on snow may be particularly important for drainage P transport (Ball Coelho et al., 2012).

Soil – type and STP

Soil type, texture, and properties are important determinants of P movement to drainage systems (Eastman et al., 2010). In general, sandier soils have had higher total P drainage losses than clayey soils

(Feyereisen et al., 2010; Kleinman et al., 2009), and organic soils may cause a concern compared to mineral soils (Cogger and Duxbury, 1984; Duxbury and Peverly, 1978; Miller, 1979). Consideration of soil type and P adsorption characteristics is extremely important before application of manure, in particular, is initiated (Kuo and Baker, 1982; Miller, 1979).

Soil test phosphorus (STP) and available P levels, particularly in subsoils, are known to be significant factors for P loads in drainage (Hanway and Laflen, 1974; Klatt et al., 2003; Sharpley et al., 1977; Smith et al., 1998; Watson et al., 2007; Xue et al., 1998). Dissolved P in drainage is particularly thought to be influenced by STP, and there has been much interest in attempts to identify a STP "change point", above which P is relatively more easily leached (McDowell and Sharpley, 2001a; McDowell and Sharpley, 2001b). Heckrath et al. (1995) identified a breakpoint of approximately 60 mg Olsen P/kg soil, but Ball Coelho et al. (2010) suggested it may be lower. Hesketh and Brookes (2000) reported these thresholds could vary widely from 10 to 119 mg Olsen P /kg soils. Mehlich-3 P soil test levels have also been used, with Carefoot and Whalen (2003) reporting drainage dissolved P concentrations were positively correlated with this parameter. There is variability here, however, as Haq et al. (2011) reported that STP and P losses in drainage were not always correlated, even considering plots with STP levels several times the optimum recommended level. Watson et al. (2007) noted it was difficult to identify any clear Olsen STP threshold value for drainage P leaching due to variability in year-to-year hydrologic conditions. McDowell et al. (2001) perhaps put it best: "*By establishing that P losses in drainage water are related to soil test P, an indicator for the potential loss of P may be developed. However the frequency, initiation of P loss and to a certain extent the fractionation of P forms … is determined by hydrology and the underlying hydrological pathways.*" Maguire and Sims (2002) added: "*Soil testing alone cannot be expected to answer all questions about the potential for subsurface P losses as every agricultural field will have variable chemical … and hydrologic properties*…" It is likely that any such STP "change point" value will be site and soil-type specific (King et al., 2014).

Of the 148 site-years reporting STP values, very few simultaneously reported a corresponding dissolved, particulate, or total P load (n = 31, 15, and 26, respectively). Additionally, comparisons between studies were complicated by use of differing STP methods (Bray 1 n = 105 but no corresponding P loads; Mehlich 3 n = 25; Olsen/Bicarbonate P n = 14). The four studies reporting Mehlich 3 values and corresponding P loads resulted in a correlation between STP and dissolved and total P loads although the mixing of surface and subsurface drainage studies may be misleading (Figure 5; Bryant et al., 2012; Eastman et al., 2010; Goulet et al., 2006; Kleinman et al., 2007). In Figure 5, the STP values greater than 300 kg P/ha were all from surface drainage studies by Kleinman et al. (2007) and Bryant et al. (2012) at sites with historically high levels of poultry litter application.

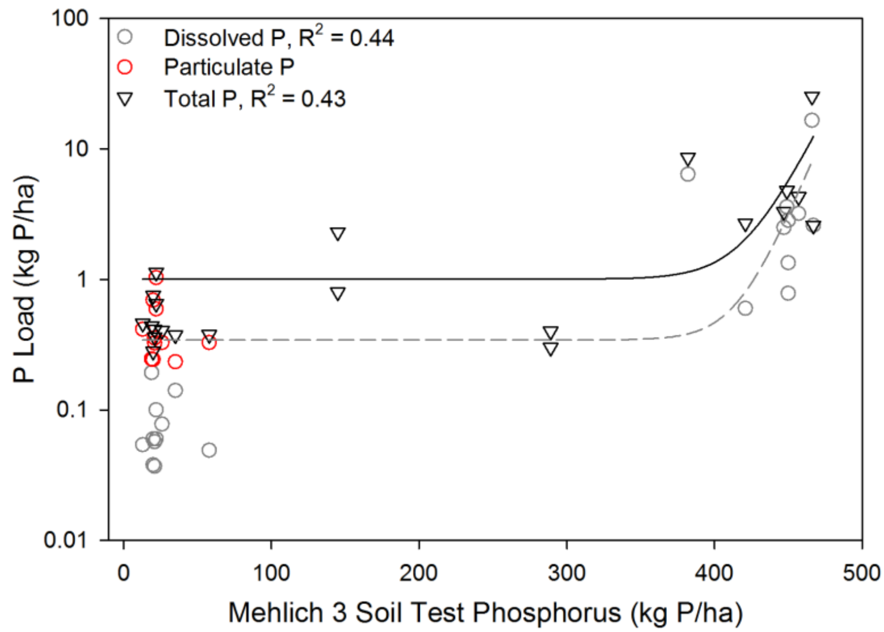

Figure 5: Dissolved, particulate and total P loads versus Mehlich 3 soil test phosphorus levels from four studies in the Drain Load database

Tillage

Tillage systems that incorporate surface residues are more susceptible to loss of sediment-associated pollutants, whereas lower disturbance tillage systems will be more prone to transport of dissolved pollutants (Zhao et al., 2001). This trend was observed in the Drain Load database with no-tillage site-years showing statistically greater dissolved P loads than conventional tillage (Table 1). There were very few particulate and total P site-years to verify the tillage impact on sediment-associated pollutants (Table 1). Brye et al. (2002) reported greater dissolved P concentrations in drainage from no-till system, though their chisel plow treatment always resulted in greater discharge. Djodjic et al. (2000) observed greater dissolved P losses from a no-till versus a tilled plot, but tillage treatments may not always result in an observable water quality impact (Djodjic et al., 2002). Conservation tillage may reduce drainage discharge (Gaynor and Findlay, 1995) but increase overall P loads compared to conventional tillage potentially due to altered crop residue mineralization and less retention of mineralized P in the soil (Gold and Loudon, 1989). This was observed here with significantly lower drainage volumes from conservation compared to conventional tillage and a greater median total P load from the conservation tillage treatment, although a more robust statistical comparison was precluded by small sample sizes (Table 1). Conservation tillage is widely known to reduce surface runoff and erosion, thus providing significant surface water quality benefits (Gaynor and Findlay, 1995; Gold and Loudon, 1989). Conservation tillage may also provide an economic benefit as this treatment here returned the greatest corn yields (Table 1). There will likely be water quality and economic tradeoffs surrounding the practice of tillage as reduced macropore connectivity can reduce soluble P transport to drainage tiles, but may increase sediment-bound P loads in surface runoff (Kleinman et al., 2009).

Table 1: Median (count) of precipitation, drainage discharge, corn yield, and phosphorus loads by tillage type; medians with the same letters are not statistically significantly different

	Precipitation*	Drainage discharge	Corn yield	Dissolved P load	Particulate P load**	Total P load**
	------------ mm ------------		Mg/ha	------------------ kg P/ha ------------------		
Conservation	790 (n=132)	140 (n=156) b	9.9 (n=53) a	0.04 (n=29) ab	0.81 (n=2)	0.42 (n=4)
Conventional	789 (n=406)	200 (n=378) a	8.1 (n=226) b	0.04 (n=52) b	0.37 (n=4)	0.36 (n=35)
No Till	770 (n=129)	170 (n=151) ab	8.7 (n=68) b	0.12 (n=21) a	0.88 (n=4)	1.18 (n=5)
Pasture	756 (n=20)	78 (n=19) b	---	0.08 (n=5)**	0.33 (n=5)	0.41 (n=5)

* No significant difference between treatments ($p = 0.719$)
** excluded from statistical analysis due to small sample size

4Rs Phosphorus Application Strategies

P Source

Phosphorus nutrient sources were divided between nine general categories for all site-years (Figure 6). Custom blends and unspecified P sources (commonly where an application rate was reported, but the formulation was not) were the most predominant (140 and 112 site-years, respectively). At 121 combined site-years, organic sources of P such as manure and litter also proved popular. Liquid swine manure was the most commonly reported organic P source in the Drain Load database (67 site-years).

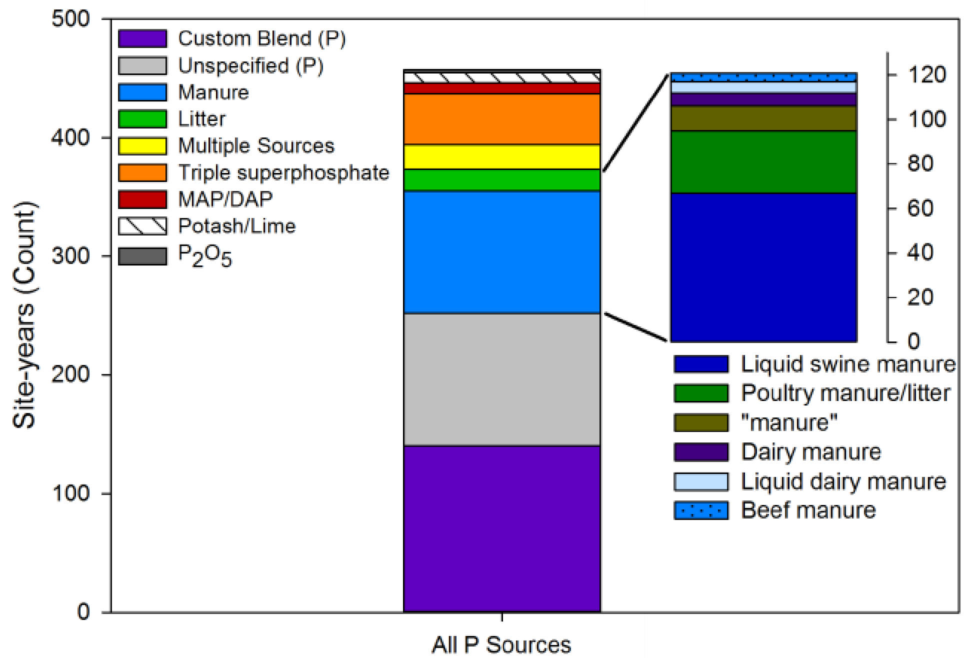

Figure 6: Phosphorus application sources reported in the Drain Load database

Across P application site-years, organic versus inorganic applications did not result in significantly different dissolved or total P drainage losses (Figure 7c and d), though this conclusion for total P losses may be limited by a small sample size (organic total P load $n = 7$). Studies comparing the drainage P load impact of organic versus inorganic P applications have generally shown higher P loads may occur from organic applications (Delgado et al., 2006; Haq et al., 2011; Macrae et al., 2007; Phillips et al., 1981), although this may not always be the case (Gangbazo et al., 1997). Use of organic P sources increased corn yields over inorganic, although manure/litter were generally applied at significantly greater rates than inorganic P sources (Figure7a and b).

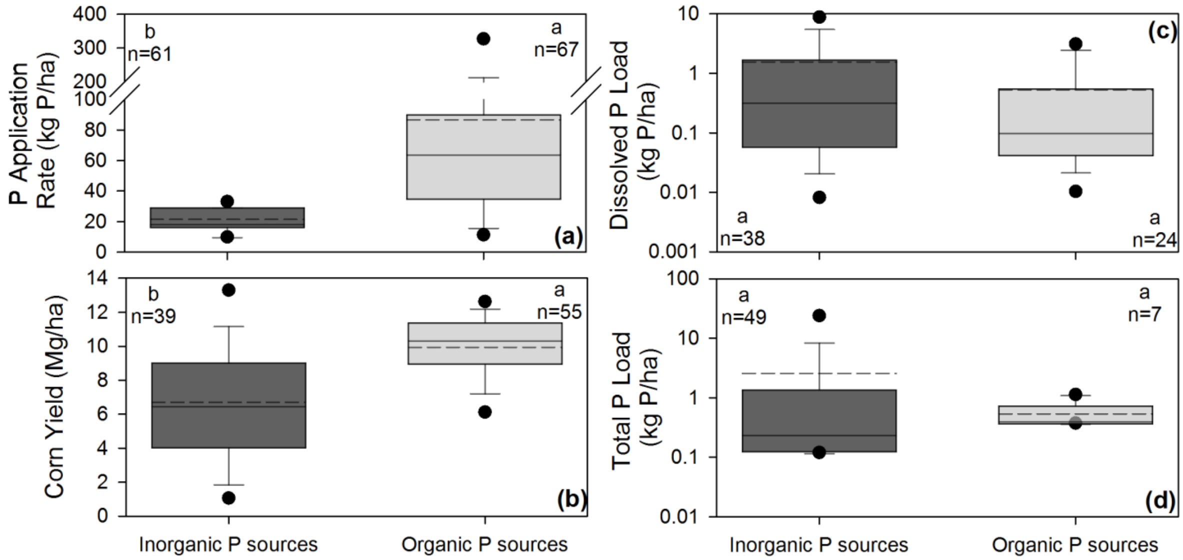

Figure 7: Drain Load database P application rate (a), corn yield (b), and dissolved and total P loads (c, d) by nutrient source category; Source impacts on dissolved and total P loads were not statistically different at p=0.219 and p=0.095, respectively

Application of different organic nutrient sources may result in varying magnitudes of drainage P losses, but there were not sufficient site-years to make such comparisons here. Any such difference may be due to the different nutrient content and P availability in each type of manure or litter (Hodgkinson et al., 2002; Kleinman et al., 2005). Withers et al. (2003) reported water extractable P was greatest for cow and pig manure slurries followed by broiler litter and farmyard cattle manure (approximately 60, 55, 25, and 15% of total P, respectively). Kleinman et al. (2005) offered a variation on this order with their ranking: swine, turkey, layer chickens, dairy cattle, broiler, and beef cattle manure.

P Application Rate

There is an obvious distinction between the level of P required to contribute significantly to agronomic production and the level of P loading that begins to impair water quality (Heathwaite and Dils, 2000; Schelde et al., 2006). Compared to nitrogen losses, P losses in drainage often do not occur at levels of economic relevance to farmers (Daniel et al., 1998; Owens and Shipitalo, 2006). Nitrogen losses in drainage are generally on the order of 15-20% of that applied in a given year (Christianson and Harmel, 2015). When P losses are expressed in similar terms, values tend to be less than 5% of that applied, though can be greater than 20% (Algoazany et al., 2007; Baker and Laflen, 1982; Culley et al., 1983; Sharpley and Withers, 1994; Withers et al., 2003). Across the literature here, this value was very close to 1% (total P median; Figure 8), with particulate P losses tending to be a relatively greater percentage of the applied P rate than soluble P losses (median: 1.0 v 0.2%). Nevertheless, conclusions shouldn't be drawn too deeply on this "annual % applied" basis, as P accumulated in soils can be transported years later (Kröger et al., 2008).

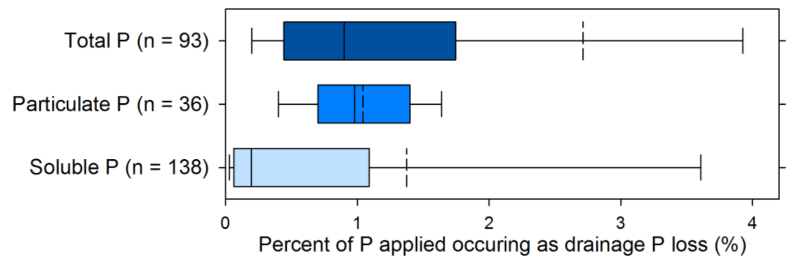

Figure 8: Percent of P applied lost in drainage as total, particulate, or dissolved P from site-years in the Drain Load database

A correlation between P application rate and P losses in drainage has been reported for both inorganic (Algoazany et al., 2007; Culley et al., 1983; Hawkins and Scholefield, 1996; Watson et al., 2007) and organic P sources (Hergert et al., 1981b; Hodgkinson et al., 2002). However, the rate effect for P in drainage may not be as strongly observed as the rate effect for N in drainage (Schwab et al., 1980), and may not become apparent until excessive P fertilization levels are reached (Izuno et al., 1991). This lack of significant correlation seemed to be the case here (Figure 9a). Dissolved, particulate, and total P loads showed an increasing trend at increasing application rates, but were generally much less than 1 kg P/ha across most reasonable P rates. Reduced rates or even cessation of P application will not be sufficient in the short term to completely mitigate drainage P loads, as a significant component of drainage P concentrations may be sourced from the soil (Izuno et al., 1991; Watson et al., 2007). The 37 and 50 inorganic and organic P application site-years, respectively, also showed P rate exerted an impact on corn yield (Figure 9b and c).

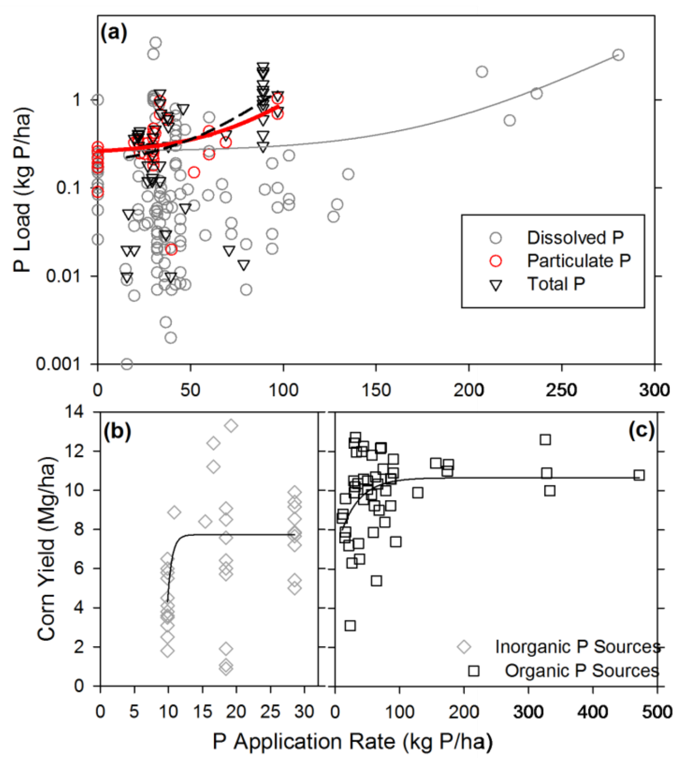

Figure 9: Dissolved (n =145, R^2=0.18), particulate (n =48, R^2=0.35), and total P (n =79, R^2=0.41) loads (a) and corn yield (b: inorganic P sources n = 37, R^2=0.25; c: organic P sources n = 50, R^2=0.14) versus P application rate

P Application Timing

Because multiple fertilizer formulations are often applied at/in a given site-year, up to two fertilizer formulations/methods/timings could be recorded in the Drain Load table for each record (i.e., Fertilizer #1 and #2). If a P application rate was given for a site-year, application of Fertilizer #1 most often occurred at planting, with out-of-season and pre-plant applications also common (41, 28, 24% of Fertilizer #1's site-years, respectively; total *n* =176; Figure 10a). If the P application occurred as the second fertilizer reported, it was most often a pre-plant application (46% of Fertilizer #2; *n* =74; Figure 7a). None of the site-years in the Drain Load database reported side-dressed P applications.

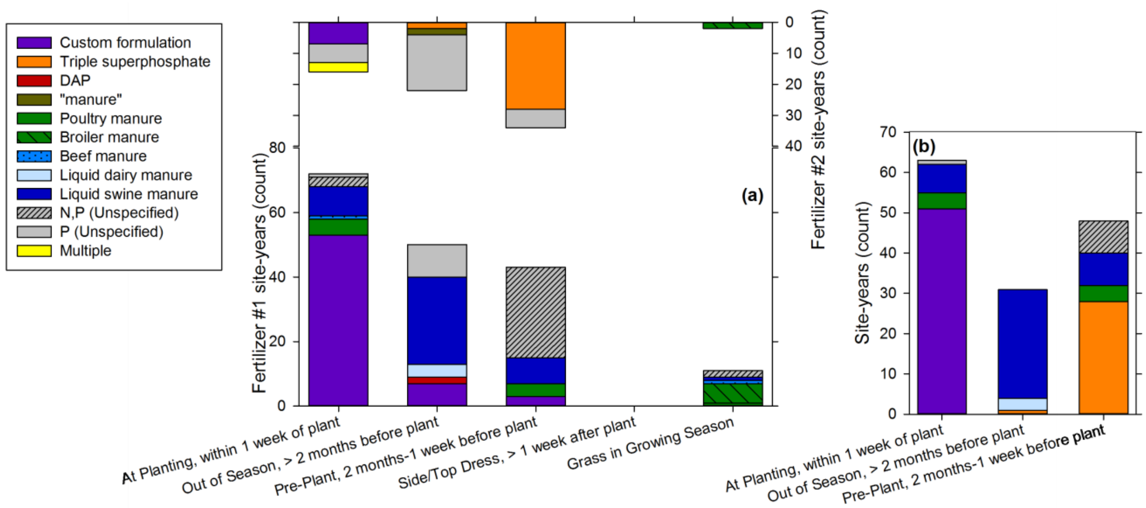

Figure 10: Phosphorus application timing by product for Drain Load Fertilizer #1 site-years (a, bottom), Fertilizer #2 site-years (a, top), and when only one P source was reported in a corn site-year (b)

Analysis of application timing from corn site-years where only one fertilizer was reported (to avoid confounding effects of Fertilizer #1 vs #2), showed the most common source/timing combinations for corn were custom P blends applied at-planting, pre-plant application of superphosphate, and out-of-season liquid swine manure application (Figure 10b). These data revealed the out-of-season and pre-plant applications resulted in greater corn yields, although they also received significantly greater P rates (Table 2). There was no significant difference between application timings for drainage P losses; comparisons were limited by very small sample sizes (Table 2). This lack of differences confounds any viable conclusion about fall versus spring P applications, and the unavailability of side-dressed site-years negated any possible conclusions about split applications. Nevertheless, temporal factors related to P application are known to be important. Timing of P applications, particularly manure, has seasonal relevance due to climate, soil moisture, and soil freezing conditions (Gentry et al., 2007; Geohring et al., 2001; Hodgkinson et al., 2002; Klausner et al., 1976). Phosphorus application relative to storm events is also known to be important as Owens and Shipitalo (2006) reported the ten largest events transported 39 to 58% of the total P load over 14 years, and Kleinman et al. (2009) noted a similar flashiness for total dissolved and total P losses in the first leaching event following manure application (Algoazany et al., 2007). To complicate matters further, "timing" impacts can also be influenced by the age of the subsurface drainage system as installation of new tiles has been shown to increase P loss (Heckrath et al., 1995; Hodgkinson et al., 2002).

Table 2: Median (count) P application rate, corn yield, drainage discharge, and phosphorus loads by P application timing for corn site-years where only one P source was reported in the Drain Load database; medians with the same letters are not statistically significantly different

	P application rate kg P/ha	Corn yield Mg/ha	Drain discharge* mm	Dissolved P load*	Particulate P load	Total P load*
				------------------ kg P/ha ------------------		
At Planting, within 1 week of plant	19 (63) b	6.0 (47) b	138 (53)	0.03 (9)	---	0.18 (3)
Out of Season, > 2 months before plant	47 (31) a	9.0 (27) a	151 (31)	---	---	---
Pre-Plant, 2 months-1 week before plant	32 (48) a	9.9 (18) a	131 (48)	0.03 (28)	0.82 (2)	1.05 (2)

* No significant difference between treatments (discharge p = 0.631; dissolved P p = 0.338; total P p = 0.105)

P Application Method

The method of P application was strongly related to the P source for several placement types. Banded and injected applications were nearly entirely comprised of custom formulations and liquid swine manure, respectively (Figure 11a). Injected liquid manure applications pose a particular concern for leaching of nutrients directly to tile lines. Shipitalo and Gibbs (2000) noted that while injection of liquid manures is recommended for odor management and nutrient usage, it may be problematic for drainage P transport especially when no-till is practiced. In the United Kingdom, guidance recommends liquid manure not be applied to lands that have been mole or pipe drained within the past year (DEFRA, 2009), but considering the prevalence of liquid manure injection in North American drainage studies (e.g., Figure 11), such recommendations may be met with large resistance in this region.

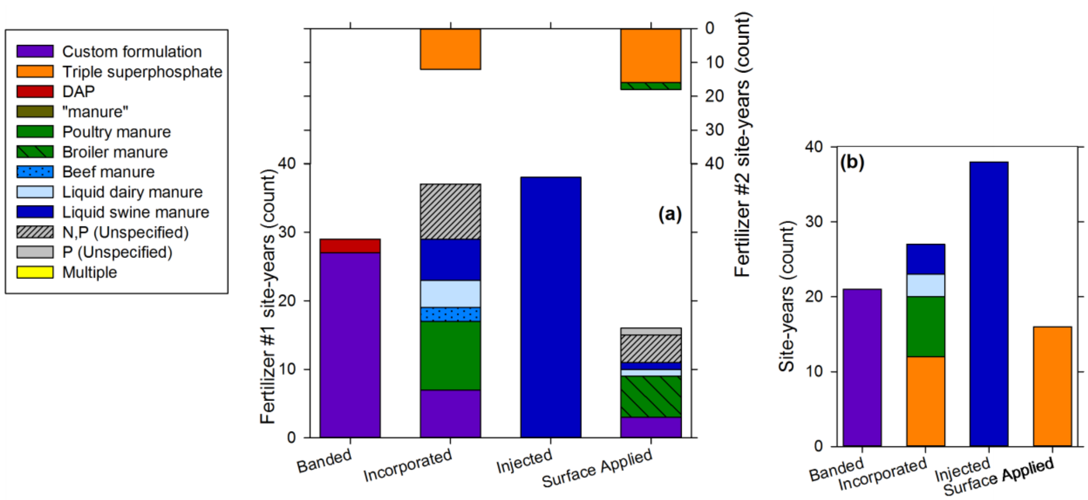

Figure 11: Phosphorus application method by product source for Drain Load Fertilizer #1 site-years (a, bottom); Fertilizer #2 site-years (a, top), and when only one P source was reported in a corn site-year (b)

The Drain Load dataset also showed a relatively similar mix of P sources was applied via incorporation and surface application, with the former proving more prevalent (Figure 11a). Plowing-in or significantly incorporating solid manures is a recommended practice to reduce P loss in drainage, as these methods disrupt the hydraulic conductivity of soil macropores (Geohring et al., 2001; Hodgkinson et al., 2002; Kleinman et al., 2009). However, Timmons et al. (1973) reported deep incorporation of N and P inorganic fertilizer resulted in drainage nutrient losses. Similarly, Feyereisen et al. (2010) reported their highest total P loss was from subsurface incorporated litter while the lowest was from a no litter control (0.48 and 0.19 kg P/ha, respectively); broadcast manure and broadcast incorporated manure P losses fell in between.

When sorting similar to the P application timing analysis was done (i.e., data were sorted to exclude site-years using more than one placement/method), injected liquid swine manure and banded custom formulations were the most prevalent combinations for corn-site-years, although superphosphate either surface applied or incorporated was also common (Figure 11b). From this selected dataset, there were no significant differences in corn yield, drainage discharge, or dissolved P load, although greater application rates were reported for incorporated and injected methods (Table 3). Though not significant, the three application methods fell in the same order for dissolved P loads regardless of mean or median (injected> surface applied>incorporated; means: 0.09, 0.028, and 0.025 kg P/ha, respectively). There were no particulate or total P loads reported in the dataset for this corn site-year only analysis. Similar to P application timing conclusions, P application placement/method conclusions were limited due to low site-year counts across the Drain Load database.

Table 3: Median (count) P application rate, corn yield, drainage discharge, and dissolved phosphorus load by P application method for corn site-years where only one P source was reported in the Drain Load database; medians with the same letters are not statistically significantly different based on a Kruskal-Wallis One Way Analysis of Variance on Ranks

	P Application Rate kg P/ha	Corn Yield* Mg/ha	Drainage Discharge* mm	Dissolved P Load* kg P/ha
Banded	18 (21) c	8.4 (15)	223 (18)	---
Incorporated	50 (27) a	10.6 (11)	124 (23)	0.02 (14)
Injected	57 (38) ab	9.6 (31)	151 (35)	0.04 (3)
Surface Applied	32 (16) b	---	155 (16)	0.03 (16)

* No significant difference between treatments (Yield p = 0.066; Discharge p = 0.076; Dissolved P p = 0.147)

Conclusions

The widespread prevalence of drainage systems in intensively cropped areas across North America overlain with the importance of freshwater resources in these regions has created a critical intersection where understanding drainage P transport is vital. Historically, dissolved nitrogen loads in subsurface drainage have received much greater attention than P loads, and this now presents a large gap in knowledge. The order of magnitude difference between agronomic P application rates and P loadings that can cause ecological damage presents a serious environmental challenge, especially compared to nitrogen. Across the literature, generally less than 2% of applied P was lost in drainage in a given site year. Reduced forms of tillage showed increased drainage dissolved P loads which was consistent with literature. However, this approach improved corn yields compared to conventional tillage indicating further evaluation of water quality/economic tradeoffs may be necessary. The timing and method of P application are both thought to be important for drainage P losses, but these conclusions could not be verified due to low site-year counts across the Drain Load database.

Mitigating P losses in drainage is complicated as (Daniel et al., 1998) noted: "*By the time these impacts are manifest, remedial strategies are often difficult and expensive to implement; they cross political and regional boundaries; and it can be several years or decades before an improvement in water quality* *occurs*." This work serves as a call to increase the number of field-scale studies documenting not only drainage P losses, but also important cropping management, nutrient application, soil property, and drainage design impacts on such losses. Moving forward, while more field-scale studies are needed, the Drain Load database may itself need to be further developed to better capture important P loss criteria (e.g., add peak flow rate information, etc.). Potential future studies should consider the high temporal frequency of sampling that is required for accurate P load determination particularly during storm-events (Culley and Bolton, 1983; Grant et al., 1996; Kinley et al., 2007; Sharpley et al., 1976). There is an additional need to consider all seasons, including snowmelt periods, for quantification of annual drainage P loads.

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