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

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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 spanning 1961–2012. 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 investigated. 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.



**Contents**



## 1. Introduction

A strong "development ethos" in North America first demonstrated through early settlement water development projects set

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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 in the US 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 $740–$1480 per ha but can boost yields by 5–25% (Blann et al., 2009). Pavelis (1987) reported the average US replacement value of drainage was $914 per ha, 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 the 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).

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 (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, and nutrient loss (Harmel et al., 2006). This well-established database was the ideal platform for this work aimed at integrating and compiling water quality and crop yield information from drained landscapes in 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.

## 2. Materials and 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). Information sources were identified through web and journal searches and by tracing citations in relevant papers and review articles. In total, 394 individual publications were reviewed. Data in MANAGE are based on 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, or (4) the paper 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 × 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).

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, yield, and 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"). 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.

Drain Load table data were analyzed with counts (e.g., histograms), box plots, and regression analyses. Due to the large dataset and high variability, 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 ($846 \pm 219$ mm) and median (828 mm) across all precipitation records ($n = 889$) 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%) were excluded to provide a distinct break point. 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).

## 3. Results and discussion

### 3.1. 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).

#### 3.1.1. 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 (Fig. 1, Pavelis, 1987; Sugg, 2007; Zucker and Brown, 1998). 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). There were relatively fewer site-years from the eastern Midwest (Indiana, Michigan, and Ohio). 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 (Madramootoo et al., 2007; Thomas et al., 1995). South Carolina and Delaware had no studies represented in spite of a high prevalence of drained cropland in each state (both approximately 25% in Pavelis, 1987). Any spatial estimate of drained lands is necessarily viewed with some uncertainty as there is no comprehensive assemblage of drainage records kept by any central authority (Blann et al., 2009).

Much more nutrient load information was available for subsurface drainage compared to surface drainage (1177 vs. 56 site-years, respectively, Fig. 1). 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 reviewed 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 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 transport studies may an important contribution to the MANAGE Drain Load table.

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 intakes 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.

#### 3.1.2. 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 (Fig. 2). There was a notable increase in the number of dissolved N site-years in the 1990's, potentially in response to nutrient concerns in the Mississippi River basin (Turner and Rabalais, 2003). More emphasis has been placed upon the study of these loads compared to total N/P and dissolved P (Fig. 2; note y-axes scales). The majority of total N and total P site-years stemmed from the 1960's.

### 3.2. 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 (Fig. 3a), it is now also thought that in some locations, seasonal rainfall totals may be more relevant than annual (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 15–40% (Fig. 4). Where this could be calculated for the Drain Load table, the mean and median were 25 and 20%, respectively, ($n = 827$; Fig. 4). Some variability can be explained by separating wet and dry years, with wet years resulting in a significantly higher percentage (Fig. 3b).

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 (Fig. 5). The difference between wet and dry years was highly significant for dissolved N ($p < 0.001$), dissolved P ($p < 0.001$) and total P ($p = 0.002$) loads; total N loads were also significantly different between binned wet and dry years but at a lower level potentially due to the low site-year count ($p = 0.072$).

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, 1998). Dry or drought years are especially associated with low nitrate losses, but high losses and/or concentrations the

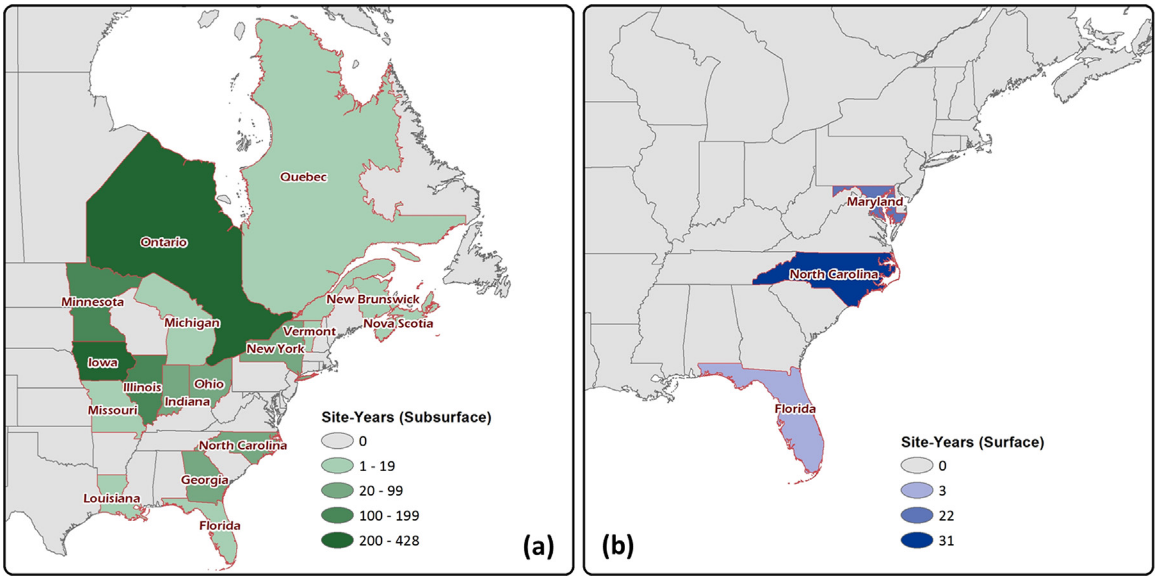


**Fig. 1.** MANAGE Drain Load subsurface (a) vs. surface drainage site-years (b).

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 N loss during a dry year was less than 3% of the applied N, but in the wet year, this increased to 25–70%.

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 yield (Table 1: "selected" site-years). Compared against the entire dataset (i.e., Figs. 3 and 5a 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 greater dissolved N loads than the pooled wet years (medians: 28.3 vs 30.2 kg N/ha; Table 1). 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.

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 com-

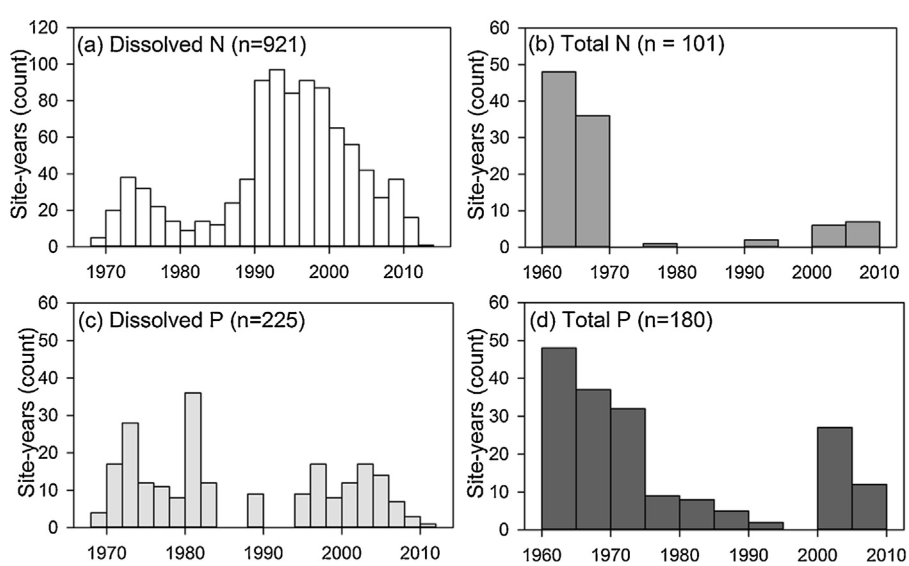


**Fig. 2.** Histograms of Drain Load database dissolved nitrogen (a), total nitrogen (b), dissolved phosphorus (c), and total phosphorus (d) site-year timing.

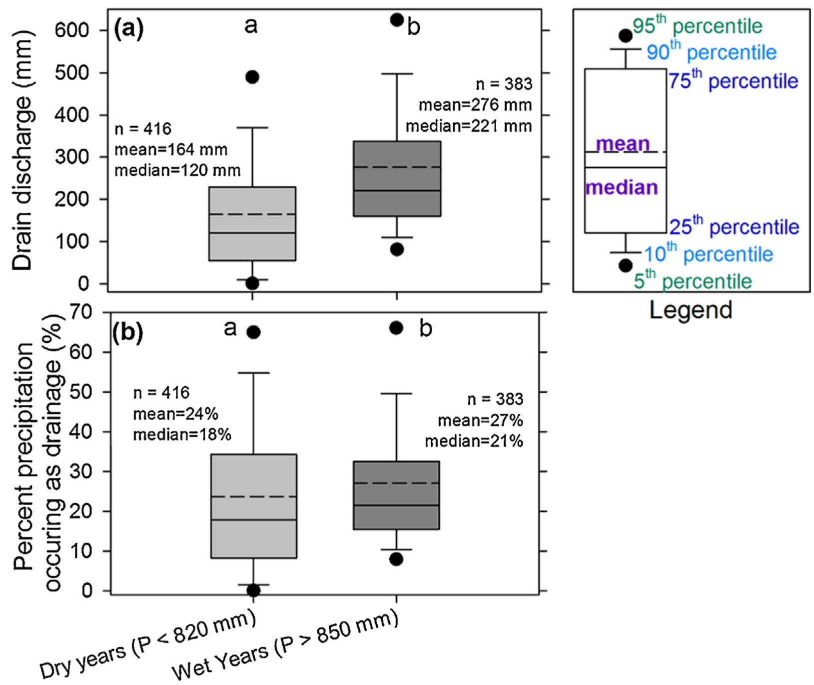


**Fig. 3.** Drainage discharge (a) and percentage of precipitation occurring as drainage (b) for the MANAGE Drain Load table; medians with the same letters are not statistically significantly different based on a Mann–Whitney Rank Sum *t*-test.

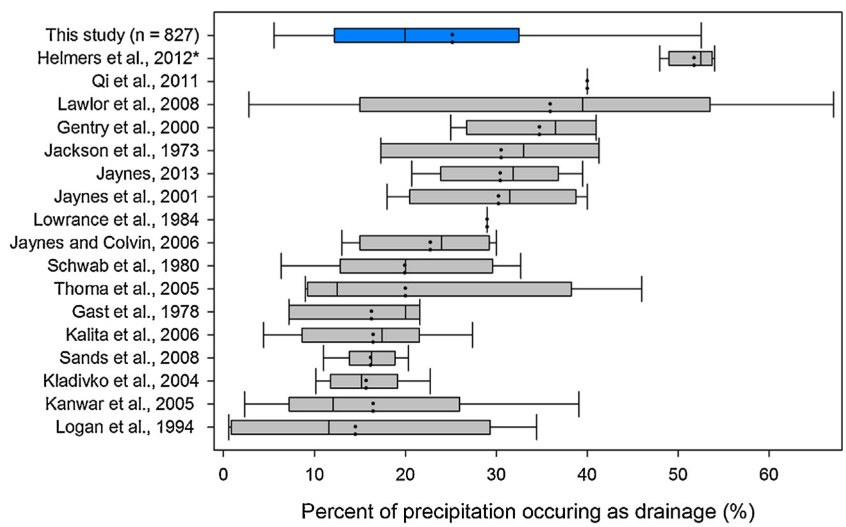


**Fig. 4.** Percentage of precipitation occurring as subsurface drainage as reported by 17 studies (199 site-years) and over the Drain Load table; *Helmers et al. (2012b) considered the drainage season March–November only.

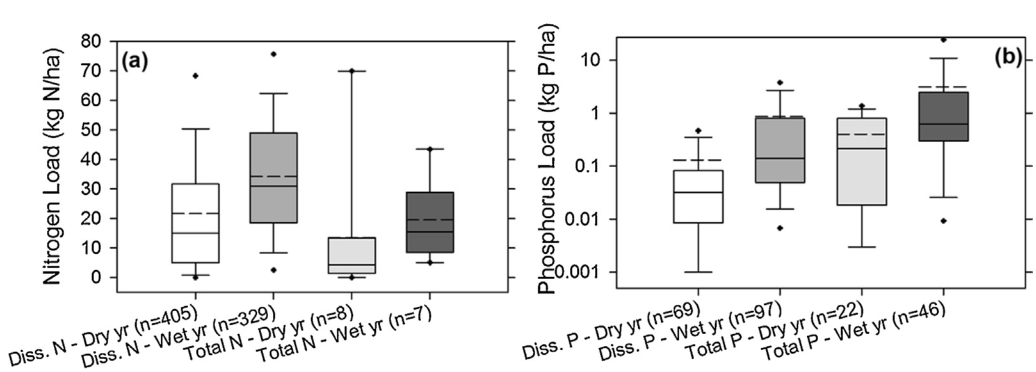


**Fig. 5.** Dissolved and total nitrogen (a) and phosphorus (b; log axis) load ranges shown by wet or dry year from the MANAGE Drain Load database.

**Table 1**
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 [a] | 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[a] | 1001(394)a | 230(356)a | 22(356)a | 30.2(302)a | 9.2(125)a | 3.4(47)a |

[a] Selected drought years and wet-following-drought years removed from the pooled data.

pared to their respective pooled datasets. These differences in corn yields were significant (Table 1: 5.9 versus 8.5 Mg/ha and 6.6 versus 9.2 Mg/ha for dry and wet years, respectively).

### 3.3. Controllable factors

#### 3.3.1. 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; Fig. 6). 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; Fig. 6). 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%).

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 (Fig. 7a–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 (Fig. 7e 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). When only the corn rotations were included in the statistical analyses (i.e., Alfalfa and Grass treatments not included), 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 Fig. 7).

There was no drainage difference between the two crop phases of a corn-soybean rotation (Fig. 7a–b), 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 (Fig. 8a; $n = 31$ from 6 studies; Bakhsh et al., 2009; Bakhsh et al., 2002, 2005, 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.

In terms of corn versus soybean phase N loss, Jaynes and Colvin (2006) and Qi et al. (2011) both observed the soybean year had lower annual flow weighted nitrate concentrations (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 on the selected corn-soybean studies, the slope of the dissolved N load regression between the two phases showed that for every additional kilogram N per ha lost in drainage from the corn phase, only 0.8 kg N/ha would be lost from the soybean phase (Fig. 8b; $n = 39$ from 8 studies; studies in Fig. 8a 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 loads 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 yet 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) are widely thought to reduce both subsurface drainage discharge

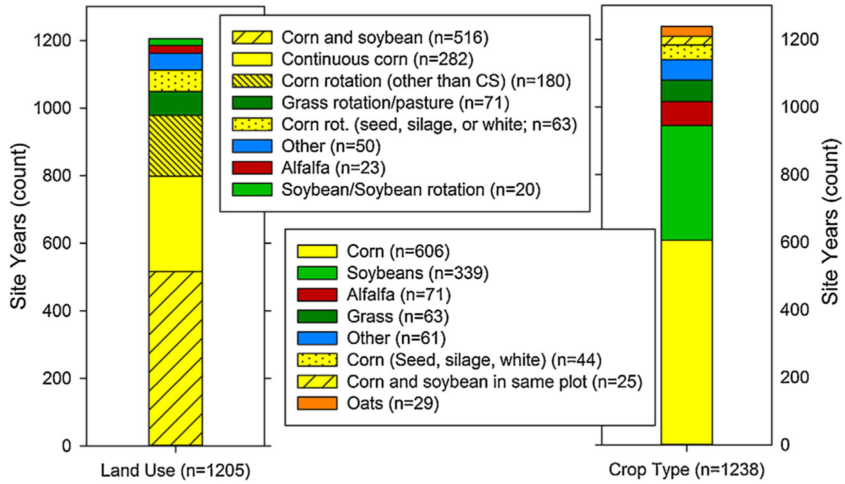


**Fig. 6.** Land use (left) and crop type (right) across the MANAGE Drain Load database.

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 (Fig. 7a and b). This lack of significance for perennial-induced reduction in drain flow may have been due to the relatively low site-year count 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 (Fig. 7a 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 (Fig. 7c 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 over-wintering (Fleming, 1990; McCracken et al., 1994). Additionally, it is worth noting that a drainage water quality benefit of perennials has not exclusively been observed across all years in all studies where

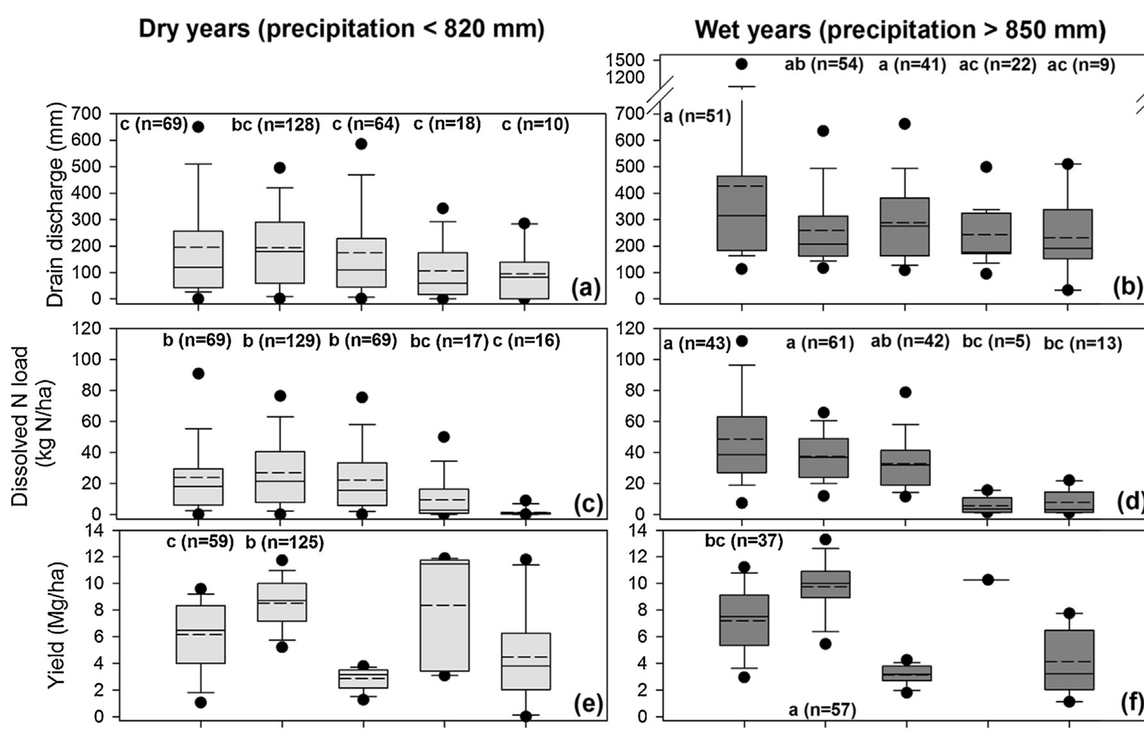


**Fig. 7.** Drainage discharge (a, b), dissolved nitrogen load (c, d), and crop yield (e, f) between crops grouped by wet and dry years; medians with the same letters are not statistically significantly different based on a Kruskal–Wallis one way analysis of variance on ranks.

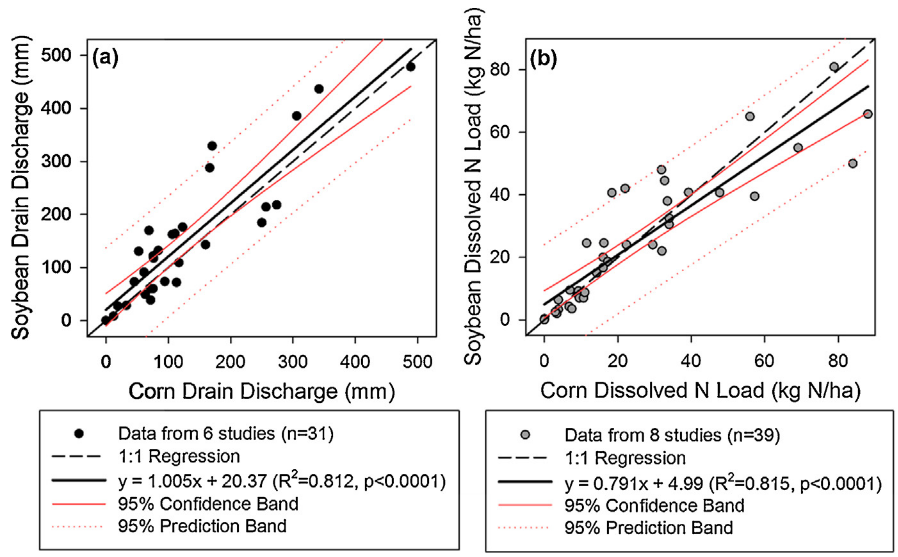


**Fig. 8.** Corn versus soybean drainage discharge (a) and dissolved N load (b) from selected studies where corn and soybeans were both grown in the same year on separate plots or fields.

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).

### 3.3.2. Tillage management

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 2). 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, 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.

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 their 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 generally consistent with this analysis, but not significantly (Table 2). In the end, tillage management may play a fairly minor role in predicting 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 2). 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 2; note the low site-year count).

### 3.3.3. Drainage system design

Tile drainage spacing in the Drain Load database ranged from 2.5 to 43 m (Fig. 9a; one 100 m spacing outlier was removed from analysis; 27 and 24% used 7.6 and 28.5 m spacing, respectively). Drain depths ranged from 0.5 to 1.5 m with a third of studies using 1.2 m which is consistent with conventional tile drain depth in the US Midwest (Fig. 9b). 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.

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 tile 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 (Fig. 10a 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.

Shallow placement of drainage pipes decreases drainage discharge and associated nitrate loads, the latter of which was

**Table 2**
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 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[a] | Wet[a] | 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 |

[a] Dry years: precipitation <820 mm; wet years: precipitation >850 mm.

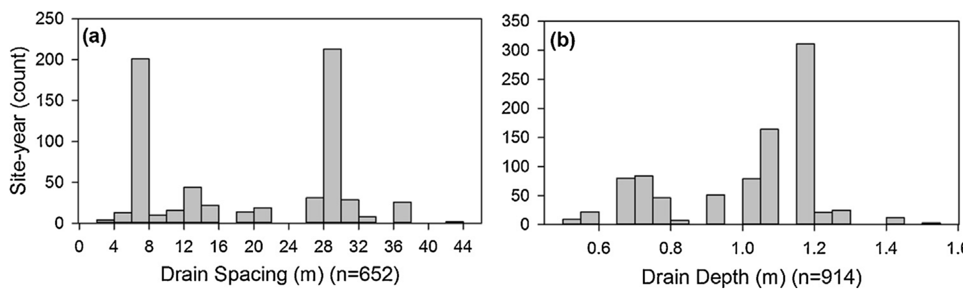


**Fig. 9.** Histograms of Drain Load database site-years by tile drain spacing (a) or tile drain installation depth (b).

corroborated here (Fig. 10c and d; Burchell et al., 2005; Davis et al., 2000; Gordon et al., 2000a; Sands et al., 2008; Schwab et al., 1980; Smith and Kellman, 2011). However, shallow drain placement must allow sufficient depth 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).

The reduction in dissolved N load due to shallow drain placement is largely thought to be 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. Regressing drainage discharge by dissolved N load for five binned groups of drain depths revealed an interesting trend in the binned groups' regressions (Fig. 11). As the drain installation depth increased, the regression

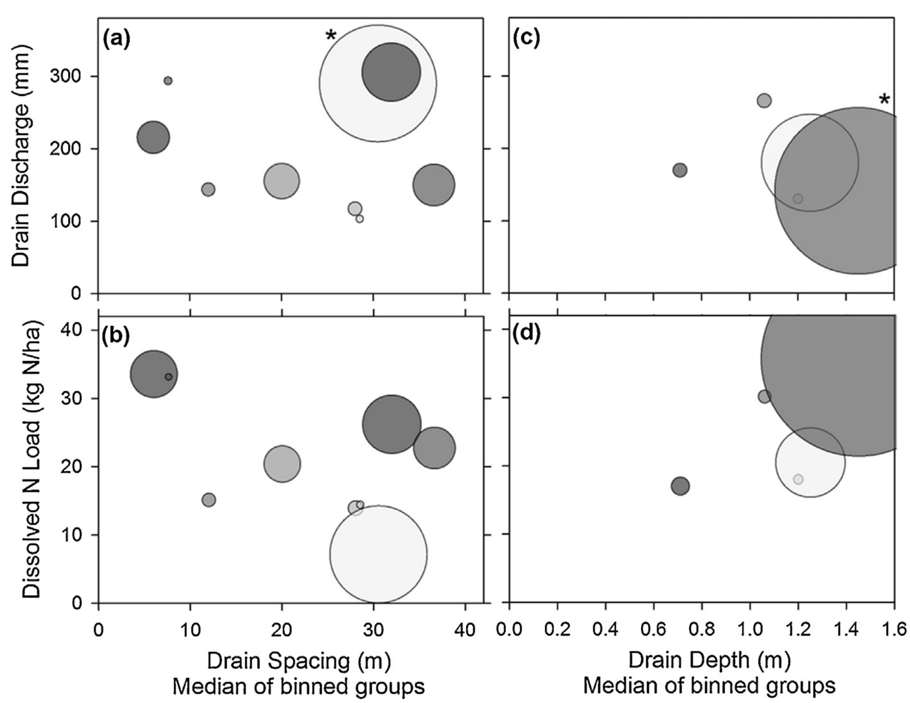


**Fig. 10.** Median drain spacing (a and b) and drain depth (c and d) of binned spacing- or depth-groups graphed against the median drain discharge (a, c) and median dissolved nitrogen load (b, d); bubble size determined by the scaled inverse of the population for each binned group (spacing bubble size = 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).

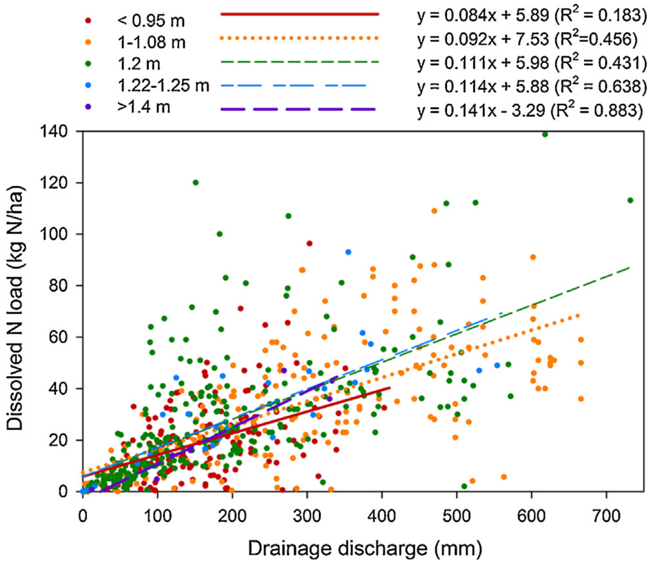


**Fig. 11.** Drainage discharge and dissolved nitrogen load relationships for five groups of binned drain tile installation depths.

slope increased and y-intercept generally 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. In dry years, this increased storage capacity and the theoretical potential for increased denitrification over this volume may act to reduce N loads. 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.

### *3.4. Knowledge gaps*

An important outcome of any review is the identification of potential gaps in scientific understanding. 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 of 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 snowmelt 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 diurnal freeze/thaw cycles, rain falling on snow, and the breaking of the 'freeze line' which may result in a very rapid transition from no drainage to pipe-full flow (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.

**Table 3**
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 mm | Drainage discharge | Dissolved N load 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 |

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 3). 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).

There are clearly gaps in peer-reviewed records of annual drainage nutrient loads from certain geographic regions and from certain types of drainage systems (Fig. 2). 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) 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 aging infrastructure is 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.

## 4. 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 practical 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 transport studies with coordinated controls across multiple sites and years would increase statistical power for more robust comparisons in the future.

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