Nitrogen Losses: A Meta-analysis of 4R Nutrient Management in U.S. Corn-Based Systems

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Interpretive Summary

Nitrogen (N) fertilizer has enabled modern agriculture to produce sufficient food for a growing population. However, environmental damages from the loss of nitrous oxide (N_2O) – a potent greenhouse gas – and nitrate (NO_3) – a water quality pollutant – increase the need for improved management that minimizes losses and improves efficiency. Beginning in May 2014, we collected data from published reports of N_2O emissions and NO_3 leaching losses from corn-based cropping systems throughout North America with the goal of exploring how these N losses are influenced by 4R fertilizer N management (right rate, source, timing, and placement). The specific aim of this project was to determine the impact of 4R nutrient management on the unintended losses of fertilizer N as either N_2O or NO_3.

A comprehensive literature search identified 4400 research papers that mention fertilizer, nitrogen, or nutrient management in agriculture, or fertilizer-associated N_2O or NO_3 losses. After a review of titles and abstracts, the majority of studies were discarded because they were not about cropland, corn, or N losses, were outside North America, or did not address field N losses. A total of 237 studies fulfilled all of our search criteria and were subjected to further review. We carefully assessed each of these papers in order to identify the subset that reported co-occurring measures of N fertilization rate, crop yields and either N_2O or NO_3 losses. A total of 27 studies contained N_2O data and 22 contained NO_3 loss data together with crop yield and N fertilization rate (one study reported both losses). The final database built from these 48 individual studies included 408 observations of N_2O emissions and 396 observations of NO_3 leaching losses (runoff losses were not considered). An observation is defined as the growing season or annual N loss reported for a single year at a specific location for a given treatment.

Locations across studies for the N_2O and NO_3 data rarely overlap, only one study reported losses of both N_2O and NO_3, and management practices for both types of field studies are diverse. For example, 60% of NO_3 observations and only 1.4% (reportedly) of the N_2O data were from tile-drained fields, plus 40% of N_2O observations and only 8% of NO_3 data points were in no-till systems. Thus we were unable to identify possible trade-offs between air and water quality N pollutant losses (i.e., N_2O vs NO_3 losses).

We constructed N fertilizer response curves for yield, N_2O emissions, and NO_3 leaching losses. Crop yield increases with N rate up until a saturation point, N_2O emissions respond in an exponential fashion, and NO_3 leaching losses tend to show a linear relationship. Fertilizer source comparisons were conducted in about half of the N_2O observations, but only one NO_3 study examined different sources. Placement was compared in 15% of N_2O and 9% of NO_3 observations; timing was compared in 2% and 13% of N_2O and NO_3 observations, respectively. Effect sizes were calculated for six fertilizer source contrasts affecting N_2O – four of which showed significant loss reductions – and for one timing contrast affecting NO_3 – which was not significant. All other treatments had fewer than nine side-by-side comparisons or were limited to a single location.

Multi-level regression models were developed for both N_2O and NO_3 with the full datasets, and also with a subset of observations at typical fertilizer N application rates. By considering multiple factors, these

models captured differential effects of management and location between studies – thus broadening results beyond the specific side-by-side comparisons. With many different fertilizer N rates used in the component observations, rate proved to be a key factor for both N_2O and NO_3 losses. Nitrous oxide emissions increased with average July temperature, but decreased with the use of nitrification inhibitors and when fertilizer was applied while the crop was growing (i.e., side-dress). Nitrate losses were lower in soils with more soil C, and also in dryer climates, except where irrigated. While aqueous ammonia resulted in higher NO_3 losses as compared with other fertilizer sources, this may have limited practical or policy impact since it is not a commonly used fertilizer source – so not a component of the typical baseline practice. With poor regional coverage, minimal overlap between N_2O and NO_3, and limited data on placement and timing impacts, we have limited knowledge about important management practices. As a result, the best-available process-based models, which only have these existing studies for calibration, may be the only somewhat reliable way to compare and contrast loss responses to 4R fertilizer management in the foreseeable future.

Introduction

Fertilizer is essential for food production to feed a growing global population. However, worldwide nitrogen (N) fertilizer recovery is usually less than 50% (Fageria and Baligar 2005), leaving significant room for loss. These N losses can contribute to serious environmental consequences such as coastal dead zones and fish kills, acid rain, climate change, and stratospheric ozone destruction (Galloway et al. 2008). These damages result most notably from nitrate (NO_3) leaching and runoff, and nitrous oxide (N_2O) emissions coming from nitrogen (N) not taken up by crops. This creates a need for improved fertilizer management that can both increase cropping efficiency and minimize the export of problematic nitrogen pollutants from farm fields. Such improvements can enhance both farm profitability and environmental sustainability.

Agricultural Nitrogen Losses

Fertilizer N losses increase crop production costs, and uncertain fertilizer use efficiencies may lead to over-application as a form of insurance. Reported recoveries of fertilizer N by corn range from 14 to 65% (Dinnes et al. 2002), and while some unused fertilizer stays within soil organic matter, significant losses are not uncommon. An improved understanding of the form of these losses can help target management efforts.

Key agricultural N loss pathways include gaseous emissions in the form of ammonia (NH_3) and nitrous oxide (N_2O) and aquatic losses that stem mainly from nitrate (NO_3) leaching. Studies in arable systems have found high variability in NO_3 loss rates, ranging between 3 and 54% of applied N (Di and Cameron 2002b). In comparison to NO_3, a typically smaller proportion of agricultural N is lost to the air. An extensive review by Bouwman et al. (2002) concluded that 7% of N in synthetic fertilizer is lost as NH_3 in industrialized countries. Ammonia losses from manure are about three times those from synthetic fertilizer. Denitrification, when soil microbes make use of NO_3 in oxygen-limited conditions, releases other gaseous forms of N, most notably dinitrogen gas (N_2), NO_x, and N_2O. The proportion of denitrified N released as N_2O varies. Agriculture accounts for 75% of total annual N_2O emissions in the U.S. (Cavigelli et al. 2012). Grace et al. (2011) estimated that about 2% of N applied to cereal corn in the North Central Region of the U.S. between 1964 and 2005 was lost as N_2O.

Nitrate in drinking water can have negative health impacts and is partly responsible for algal blooms and dead zones in impaired waterways and coastal areas (Camargo and Alonso 2006). Ammonia contributes to acid rain and can further harm sensitive natural habitats by adding N through deposition after being transported long distances through the air (Driscoll et al. 2001). The role of N_2O as a greenhouse gas (GHG) is especially significant, with more than 300 times the global warming potential than carbon dioxide (Forster et al. 2007). Nitrous oxide is also the most significant ozone depleting substance in the stratosphere (Ravishankara et al. 2009).

Whenever and wherever excess inorganic N is present in a farm soil in excess of plant demand there is a high potential for export of that excess N. There are many techniques by which farmers can avoid this situation (see Box A). They can reduce losses by simply adding less N, but this alone may lead to a reduction in crop yield. Maintaining or increasing yields along with improved N management is critical to avoid shifting production and associated N losses elsewhere. Better options are to add smaller but more strategic doses of right N source at the right times and in the right places so that supply is best matched to crop demand. These four key factors of fertilizer management – namely, the right rate, right time, right place, and right source – are promoted in agricultural extension as the 4R Nutrient Management framework.

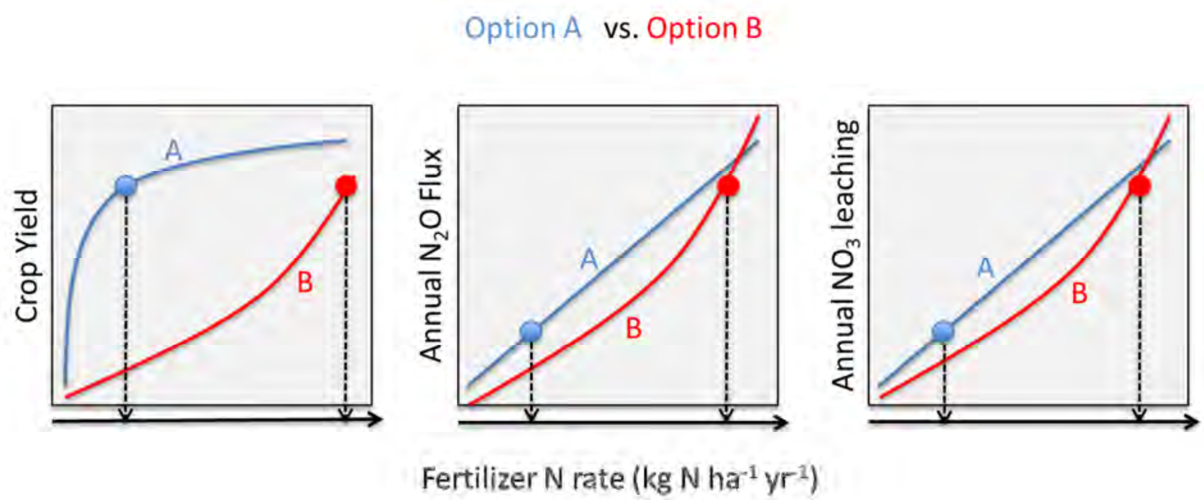

Figure 1. Schematic representation of yield and pollutant responses to fertilizer N rate. Options A and B illustrate possible management, soil, or climatic systems that exhibit differing potential to optimize yield and N loss responses.

It is very important to understand how both crop and pollutant outcomes vary with fertilizer management. Even though N losses are often calculated as a proportion of fertilizer applied,[1] such a linear response of yield and N losses to fertilizer application rate may not be the best model. There are also many reasons to doubt that NO_3, N_2O, and yield all have the same response to fertilizer source, timing, and placement. In fact, it is only if they don't respond in the same way that there is potential to reduce losses while maintaining or increasing yield. Responses are also unlikely to be invariant across space and time. The response to fertilizer rate could differ from one location to another, or from one management system compared to another. For example, the two options shown in Figure 1 could represent two different types of soil or two different tillage systems, with varying yield, N_2O, and NO_3 responses to fertilizer rate within a given range. In each option, an optimum point can be found at which the benefits of yield can be maximized in relationship to the costs of N pollution.

[1] Most regional and global models – as well as smaller-scale models used in existing environmental markets – assume a linear relationship to fertilizer application rate (i.e., N_2O and NO_3 losses represent a fraction of loading, or loading in excess of crop demand). The IPCC Tier-I model estimates that 1.0% of all mineral fertilizer is lost at the field as N_2O, plus 30% of fertilizer N is leached as NO_3 in wet or irrigated regions, of which 0.75% is then lost as indirect N_2O emissions (IPCC 2006). The Canadian Nitrous Oxide Emissions Reduction Protocol (NERP), adopted for the Alberta offsets system, uses an expert-derived model with linear N_2O response to N rate (Government of Alberta 2010). One protocol for voluntary carbon markets applies a non-linear empirical model to mid-western states for which sufficient data are available (see Hoben et al. 2011), but the rest of the U.S. is treated under linear IPCC Tier-I assumptions (Millar et al. 2012). The ADAPT model, commonly used for water quality trading programs, also assumes a set proportion of fertilizer N applied to be lost as leached NO_3 (Davis et al. 2000).

Study Objectives

Many studies on fertilizer management have evaluated responses of crop yield and N losses to a subset of management options and do so in a limited range of natural variability (soils and climate). However, rarely have they considered losses of *both* NO_3 and N_2O or how these relationships vary with soil, climate, or crop type. In this research, we compiled a database of fertilizer management field studies that measured yield, NO_3 leaching and N_2O emissions, and included all available information on fertilizer timing, placement, rate, and source, as well as climate, soils, and other management factors. By ensuring that all effects of management practices are suitably analyzed in meta-analysis of the data, this approach can provide more strategic and comprehensive information, reducing the risk of unintended negative consequences when recommending management change.

The goal of this project was to enhance our understanding of what is controlling nitrogen use efficiency (NUE) and N loss pathways. The project focused on the impact of 4R nutrient management on total N losses relative to yield from corn-based cropping systems in the United States. We asked the following questions:

> First, how do crop yield, NO_3 leaching, and N_2O emissions respond to N fertilizer application rate, timing, type, and placement? This would inform how best to manage for maximum yields while minimizing N losses in a particular place. For example, in a given situation, improved timing could enhance yield and nitrogen use efficiency (NUE) and reduce multiple forms of N loss.
>
> Second, how do these effects of fertilization practices depend on climate and soil factors? That is, how would management need to shift across climates and soils to achieve the same goals? If fertilizer rate adjustments affect N losses more in one region than another, perhaps resources for change can be directed in a targeted manner.
>
> And third, how do NO_3 and N_2O losses co-vary with management, climate, and soil? Where does management result in changes that are positively versus negatively correlated? If changing fertilizer source in a certain cropping system or soil type simply shifts losses from NO_3 to N_2O, actual benefits may be uncertain.

The resulting database also fulfills an additional project goal of compiling data in a form to which others can add and also use for comparing new study sites to existing research.

Fertilizer Management to Reduce N Losses

Crop yield and NUE implications of 4R management have been the subject of much field research and agricultural extension (Fageria and Baligar 2005; Raun and Johnson 1999). Although it is clear that better matching N addition to crop demand (with appropriate rate, source, timing, and placement) will maximize crop yields while reducing fertilizer costs, it is far less clear how the effects of various fertilization scenarios will alter the form, timing and magnitude of N pollutant losses.

Water quality concerns related to fertilizer management gained traction in field research in the 1960s (Jolley and Pierre 1977; Olsen et al. 1970). Research that summarizes management strategies for NO_3 pollution reduction has focused on the use of in-field activities including 4R management (Power et al. 2001) as well as edge-of-field or off-site activities such as biofilters and wetlands (Dinnes et al. 2002). In the first known meta-analysis of fertilizer management and nitrate leaching losses, Zhou and Butterbach-Bahl (2014) assessed the yield-scaled loss implications of agricultural N management practices in both maize and wheat cropping systems, finding that the lowest yield-scaled losses occurred at slightly suboptimal fertilization rates (e.g., at 90% of maximum yield for corn).

Box A: 4R Fertilizer Management – The Right Source applied at the Right Rate and the Right Time, in the Right Place.

Right Rate

In general, crop yields are expected to increase with fertilizer input up to the point of saturation, beyond which excess N is particularly susceptible to losses. The right rate of fertilizer application is the one that maximizes yield while minimizing economic and environmental costs. However, the rate of application that achieves these ends depends strongly on soil fertility and crop type, as well as the type, timing, and placement of fertilizer application and other management strategies. Climate and soil characteristics play very important roles, especially as they affect crop yield potential (and thus N uptake), as well as the hydrological and microbiological factors that govern NO_3 leaching and N_2O emissions. Fertilizer N application rate can in a way serve as an integrator of other management practices, with potential for downward rate adjustments as changing practices reduce losses and leave more N available to the crop.

Right Time

The right timing of fertilizer application can increase nitrogen use efficiency. In many regions, fall fertilizer applications have been common practice, allowing for fewer spring field operations and thus earlier planting. However, application rate recommendations are higher to account for the anticipated losses over winter months. Split applications, in which growing season fertilizer is applied at multiple times, can also reduce losses, although additional labor, fuel, and equipment costs may make such practice unattractive in practice. In drip and flood irrigated systems, multiple split fertilizer applications can accompany irrigation water, allowing farmers a great deal of control in timing nutrient additions to best meet crop needs. If fertilizer is applied when the crop is growing (either as part of a split application or as the only application), this is called “side-dressing”. Timing of fertilizer application can have differential effects by climate and soil type as well.

Box A (cont.)

Right placement

Fertilizer placement variations include broadcast versus incorporating (which can include banding), as well as different depths of application when injecting/banding. In irrigated systems, fertilizer can also be applied directly within the irrigation water (i.e., fertigation).

Placement affects N availability to the plant, as well as availability for loss pathways. Soil and climate factors can also affect loss and yield responses to placement. Lower temperatures may reduce the otherwise high NH_3 losses from broadcast urea-based fertilizer as compared with banded applications. Similarly, precipitation likely reduces NH_3 losses from broadcast fertilizer, but increases denitrification and leaching (and thus N_2O and NO_3 losses).

Right source

The type or source of fertilizer influences the timing of availability and susceptibility to loss because different N sources perform better in terms of NUE in different locations. Common fertilizer types include urea, anhydrous ammonia, and UAN. Controlled- or slow-release fertilizer have been developed to improved NUE, and additives such as nitrification inhibitors or urease inhibitors, while not fertilizer sources per-se, also serve to improve NUE in certain soil and climate conditions. Along with slow-release mechanisms, inhibitors are incorporated in a number of commercial brand-name fertilizer formulations that advertise their efficiency gains (e.g., Super U by Koch Agronomic Services, LLC). Manure is an important nutrient source in many cropping systems, globally accounting for around 30% of N applied to crops, but because of highly variable N availability and significant losses, tends to achieve lower NUE rates.

With increasing concerns about ozone depletion and climate change by the late 1970s, researchers began to measure in-field N_2O emissions as affected by fertilizer management (Breitenbeck and Bremner 1986; Breitenbeck et al. 1980; Bremner et al. 1981; Bronson et al. 1992). Comprehensive reviews of this research have noted that 4R fertilizer management practices can reduce N_2O emissions by better matching supply to crop needs, even though the directly measured evidence for some practices is only available in certain regions or cropping systems (Dalal et al. 2003; Snyder et al. 2009).

Quantitative summaries that include multiple practices are limited. With a focus on US cropping systems, Eagle and Olander (2012) used field data to determine the GHG mitigation potential of many different agricultural practices, including average N_2O emission reduction from 4R fertilizer management. They calculated average emission reduction potential for different fertilizer management activities ranging from 0.12 to 0.59 t CO2e ha^{-1} (0.26 to 1.3 kg N ha^{-1}). Using meta-analysis methods, Linquist et al. (2012) documented the GHG emission response to fertilizer management in rice systems, including N_2O emission reductions of 29% with nitrification inhibitors and increased N_2O emissions from deep fertilizer placement. A more focused meta-analysis on corn systems in the U.S. Midwest examined the N_2O emission implications of multiple fertilizer management practices, finding that N_2O response to rate varied by region and that the use of nitrification inhibitors decreased N_2O losses by almost 40% (Decock 2014).

Other studies (including some meta-analyses) assess single practice implications for both N_2O emissions and NO_3 leaching. Fertilizer application rate has been considered more than any other 4R practice. Emission responses to N rate are predominantly non-linear and exponential in arable cropping systems throughout the world, especially at application rates above the crop N uptake (Kim et al. 2013; Shcherbak et al. 2014; van Groenigen et al. 2010). Recent models of NO_3 leaching find that the losses per unit applied may increase with rate as well (Qi et al. 2012).

Many studies have compared the N_2O production consequences of two or more fertilizer types. In comparing studies across regions, soil temperature and moisture conditions seem to dominate emission differences between commonly-used fertilizer sources such as urea, anhydrous ammonia (AA), ammonium sulfate so that there is not always a clear “winner” (Snyder et al. 2009). For example, Venterea et al. (2010) reported from a Minnesota side-by-side field study that emissions with AA were double those of urea, but a global summary by Stehfest and Bouwman (2006) of over 150 agricultural observations found no average emission difference between these two sources. Polymer-coated urea reduced N_2O emissions by 10 to 45% in wheat, corn, and potato cropping systems (Burton et al. 2008a; Halvorson et al. 2010a; Hyatt et al. 2010), Emissions from coated urea in the first weeks after fertilization were 71% lower in barley, but continued N_2O release after that point was higher than for urea (Delgado and Mosier 1996). In at least one case, polymer-coated urea produced lower crop yield (Venterea et al. 2011b), so that calculating emissions on a yield-scaled basis negated any possible benefit.

Few fertilizer source comparisons for NO_3 leaching implications have been reported, with the exception of sources that are combined with nitrification inhibitors (NIs). Even though NIs are not a source of fertilizer per-se, we treated them along with other sources, as they are formulated into some high-efficiency N fertilizers, and even when not part of the formulation they are applied at the same time as N fertilizer. Nitrification inhibitors can reduce total NO_3 losses in grassland and vegetable cropping systems by up to 76% and 59%, respectively (Cui et al. 2011; Di and Cameron 2002a). A meta-analysis of field studies around the world determined average N_2O emission reductions of 38% with nitrification inhibitors (Akiyama et al. 2010), with more pronounced effect in grasslands (54% reduction) than on upland (34%) or rice fields (30%).

There have been considerable efforts to document the losses associated with the common practice of fall fertilization. Nitrous oxide losses from fall N fertilizer application may be more pronounced in regions where the spring thaw generates a strong denitrification pulse (Johnson et al. 2011). Shifting application from fall to spring reduced N_2O emissions by over 70% in a Canadian wheat/canola rotation (Hao et al. 2001) and decreased NO_3 losses by about 40% in a New Zealand grassland (Di and Cameron 2002c; Hao et al. 2001). Nitrate implications in corn seem to be variable, with a 14% loss reduction in Minnesota (Randall and Vetsch 2005), but no impact in an Iowa study (Lawlor et al. 2011).

Nitrous oxide emission reductions of up to 40% have also been achieved by changing fertilizer timing from pre-plant to split in potatoes (Burton et al. 2008b). However, in corn, neither splitting fertilizer N application instead of all pre-plant in Indiana (Smith et al. 2011) nor side-dressing instead of pre-plant in North Dakota (Phillips et al. 2009) and New Brunswick (Zebarth et al. 2008) affected N_2O emissions in a consistent manner. Split fertilizer applications are also recommended for reducing NO_3 losses (Dinnes et al. 2002), although supporting field data are limited.

Far fewer studies have explicitly examined the impact of fertilizer placement on N_2O and NO_3 losses. However, other management factors have proven important. Soils with stratified organic matter (i.e., significantly greater amounts near the surface as typified in no-till systems) lost less N to denitrification when fertilizer is placed at greater depth (Khalil et al. 2009). In a meta-analysis of over 200 direct comparisons between conventional tillage and reduced or no-till systems (RT/NT), van Kessel et al. (2013) found that RT/NT reduced N_2O emissions only when fertilizer was placed at ≥ 5 cm depth. Walters & Malzer (1990) tested incorporation of urea fertilizer, with no significant effect on NO_3 losses or water percolation. Bakhsh et al. (2010) tested localized compaction and doming as opposed to conventional knife injection for UAN over a five-year period, with some reduced NO_3 concentrations but increased total flow, so that total leaching losses between the two systems were not significantly different.

No known reviews quantitatively examine the fertilizer management effects on both N_2O and NO_3 simultaneously. One recent field study of corn in Minnesota measured N losses with pre-plant polymer-coated urea and SUPERU™ as well as urea applied with a split application; finding some loss reductions in both N_2O and NO_3 losses with the improved treatments (Maharjan et al. 2014). Some combined N_2O and NO_3 measurements have also been reported from non-corn cropping systems. Studies of nitrification inhibitors in grassland and vegetable cropping systems have measured co-existent N_2O and NO_3 loss reductions (Cui et al. 2011; Di and Cameron 2002a), and some source-timing combinations on potato were also beneficial for both loss pathways (Venterea et al. 2011a).

Therefore, while field research is beginning to assess the responses of both N_2O and NO_3 to fertilizer and other management, the relationship between the two is largely undefined. This research fills that gap with a combined analysis, beginning to assess the potential for synergies and trade-offs that have significant implications for market trading systems in both water and air quality.

Methods

This research project expanded an existing literature database on field research (as of 2011) in the U.S. and Canada regarding N_2O emissions and 4R nutrient management (Eagle and Olander 2012). In this case, however, focus was limited to field studies in corn systems in North America. Corn takes up about one-quarter of all harvest cropland in U.S. and accounts for 40% of the fertilizer used. Because of its significance to U.S. agriculture, much existing field research pertains to rotations dominated by corn. Limiting the analysis of data to one crop also has potential to reduce some unknown and unquantified differences between crops that could be further complicated by climate, soil, and other management characteristics. Even with this focus, there is potential for information exchange to and from other cropping systems.

Data Compilation

Data collection began with a comprehensive literature search for field studies of North American corn-based systems published through July 2014. This search was performed using ISI Web of Science with terms related to agricultural N losses (fertilizer, nitrogen, nutrient management, agriculture, nitrous oxide, nitrate, leaching, and emissions), and identified 4400 papers in the scientific literature. Of these studies, over 1800 were discarded upon initial title review because they were not about cropland systems or N losses, they dealt with non-N types of fertilizer (e.g., phosphorus or potassium), or they addressed N losses and transport after the field. The remaining 2590 papers were put through a more detailed review of article abstracts to select those with potential for useful data. This triage discarded about 1500 papers that were either outside of North America or not in corn systems, plus a selection of others that were about manure only or did not address fertilizer management. Laboratory and greenhouse studies were also excluded.

In the end, 237 papers were selected with potential for field data on fertilizer N management and losses. Review articles from the initial search were also utilized for a snowball search to capture additional research, but of 150 possible papers identified in this way, only one contained useable data. Each study was reviewed for topical relevance and data availability, with results of this review recorded in a relational Access database designed for the project. The database collected information at three different levels: first the study/citation, then locations within that study, and then observations at each location. To reduce data entry errors, drop-down lists with potential options were created for all relevant variables (e.g., fertilizer source). If a research study did not contain useful data, the reason for exclusion was noted, and it was set aside without recording any further information. Each citation retained for the database recorded either or both N_2O and NO_3 losses from field experiments of corn-based cropping systems in the United States or Canada for which 4R fertilizer management treatments were applied.[2] Manure and other organic fertilizer were not included because of the additional complexity this would have added.

Location-level data including climate and soil characteristics were recorded once, to be connected with all related observations. At the observation-level, all available details were collected regarding crop type, crop rotation, irrigation and water management, tillage, fertilizer management (rate, timing, placement, and source), crop yield, N uptake, soil N, and N losses as N_2O and NO_3. Nitrification inhibitors were included in the source category. In many cases, it was also possible to record fertilizer rate and crop type from the previous year, as well as variability measures for yield and N losses. All observations included in the database were in the corn phase of the rotation and recorded 1) losses of either or both N_2O and NO_3 over at least 55 days during the growing season, 2) crop yield, 3) N application rate, 4) number of replicates. An observation is defined as the growing season or annual N loss reported for a single year at a specific location for a given treatment.

[2] While Mexico was included in the search (being in North America), no available loss data were identified.

If crop yield, N losses, soil water NO_3, and soil extractable N values were only presented in graphical form without directly reporting in the literature, numerical values were requested from individual data owners for research published post-2004. For the same date range, clarifications about and numerical values of data from combined treatments were also requested from individual data owners. Any remaining data available only in graphical form were quantified using DataThief III (B. Tummers, http://datathief.org).

Climate data included the long-term (30-year) averages for total annual precipitation, mean annual temperature, and mean July temperature, as well as annual precipitation for the study year. Where not provided within the published studies, these data were retrieved from the nearest National Climatic Data Center (NCDC) weather station. Soil characteristics included organic matter content, drainage, and texture.

Details pertaining to N loss measurements were also documented, such as the relevant time-frame, frequency of measurement, peak N loss rates, and method of measurement. Nitrous oxide emissions were measured solely with in-field chamber methods. Leaching losses in the database were calculated from NO_3^- concentrations measured in porous-cup-collected soil water in combination with hydrological modeling (14%), by measuring tile water drainage volume and NO_3^- concentrations (61%), or by using lysimeters (25%).

All corn yield values were reported at (or corrected to) 15.5% moisture. In the small number (7%) of cases where yield was reported as total biomass, it was converted to grain yield using a harvest index of 0.53.[3] A number of observations were removed from further analysis because drought reduced yield or the yield reported was not reported separately for distinct treatments. Given that the objective is NUE (i.e., yield per unit of nutrient) rather than yield increases or N loss reduction alone, yield-scaled N losses are used in this study as in more recent research addressing both N_2O (Johnson et al. 2011; van Groenigen et al. 2010; Venterea et al. 2011b) and NO_3 (Zhou and Butterbach-Bahl 2014).

Some data for N uptake were imputed for use in the analysis. Total aboveground N uptake was reported in 21% of observations, and grain N uptake in an additional 10%. Where only grain N uptake was reported, this was converted to aboveground N using an estimated N harvest index for corn of 0.64 (i.e., total N in grain as fraction of total aboveground N, see van Groenigen et al. 2010). For observations where N uptake was not reported directly, it was estimated from crop yield (converted to total biomass with a harvest index of 0.53), and an average whole plant N content of 0.96% (the median for all observations that recorded N uptake).

Data variance is commonly used as a weighting factor in meta-analysis, especially when there are many observations within each study. Some researchers have assigned an average standard deviation from other studies to the observations that didn't report variance (Akiyama et al. 2010). In the current database, variance was reported for only 33% of yield observations, 58% of N_2O observations, and only 19% of NO_3 observations. Since each study had an average of three observations per treatment, and

[3] Based on a literature review, a harvest index (grain as a fraction of total above-ground biomass) of 0.53 for corn seems a reasonable estimate (see Appendix A for details).

there were numerous studies per location, variance-weighting was determined to be inferior to location-based weighting.

Data Analysis

The core relationships are the responses of crop yield and N losses to both absolute fertilizer rate and fertilizer excess (i.e., fertilizer N applied minus N uptake). Where three or more N rates – including a control with zero or near-zero N[4] – were used for a specific cropping system in a single year and location, we constructed N fertilizer response curves for yield, N_2O emissions, and NO_3 leaching losses. Due to lack of N uptake data (only available for 21% of observations), N excess curves were less robust and subsequently discarded.

A more traditional meta-analysis of side-by-side comparisons was then used to detail loss responses to those management practices for which sufficient observations are available. Results were weighted so that locations with many observations did not overwhelm those with less available data. Each observation was weighted by the following factor: 1/ ln (# of observations in location).

Multi-level (hierarchical) models constructed for both N_2O and NO_3 losses use all observations, including those that did not measure a rate response, with location as the group level. Such models are also called mixed-effects, because they include a combination of both random and fixed effects within the model. By determining effects at the group level, this type of model determines the relationships between losses and the driving factors for each location, while allowing effects at each location to explain the overall effects.

The multi-level model initially looks like your typical regression,

$$(1) \qquad y_{ij} = \beta_1 + \beta_2 x_{2ij} + \cdots + \xi_{ij} \qquad\qquad \xi_{ij} = \sigma_j + \epsilon_{ij}$$

where y_{ij} is nitrogen loss of observation *i* at location *j*, β_1 is the constant or intercept, and x_2 and following are covariates with coefficients of β_2 and following. The residual error term is ξ_{ij}. Since we expect the losses at a certain location to be correlated with each other in this model, the error term, ξ, can be split into two components – σ_j, the error shared by all observations at the same location; and ϵ_{ij}, the remaining residual unique for each observation. This error term at the group level (σ_j) represents the combined effects of omitted characteristics or unobserved heterogeneity for each location.

A likelihood ratio test examines the total variance at the group level (i.e., between groups) and the total variance at the individual level (i.e., within groups). This test determines whether group level variance is sufficient to favor the multi-level regression over an ordinary regression model. For both N_2O and NO_3 losses, the multi-level structure improves the models, largely because background losses and response to treatment vary by location.

[4] Fertilizer N control application rates were generally zero, but also included treatments with up to 30 kg N/ha applied at planting – often called a “starter” – or treatments up to 45 kg N/ha that included with a known amount of N in irrigation water.

In addition to correcting for unknown or un-quantified differences between groups, the multi-level model addresses unbalanced observations, so that the overall effect of an independent variable (or $\hat{\beta}$) is a weighted mean of the cluster means. Therefore,

$$(2) \qquad \hat{\beta} = \frac{\sum_{j=1}^{J} \mathrm{w}_j \bar{y}_{.j}}{\sum_{j=1}^{J} \mathrm{w}_j} \quad , \qquad \text{where} \quad \mathrm{w}_j = \frac{1}{\hat{\varphi} + \hat{\theta}/n_j}$$

and $\bar{y}_{.j}$ is the average for each cluster, now weighted by the weighting factor w_j. The weighting factor, or the contribution of a specific group to the overall mean, increases with increasing group size (i.e., n_j is greater. Similarly, as the variance increases, both within groups ($\hat{\theta}$) and between groups ($\hat{\varphi}$), the weighting factor decreases and the contribution of that group is reduced.

Results

The final dataset consisted of 408 observations of N_2O field losses from 27 studies (18 distinct locations) and 396 observations of NO_3 field losses from 22 studies (16 distinct locations). Only one study (16 observations) reported simultaneous N_2O and NO_3 losses. The data cover much of the area of North America for which corn is a primary crop (Figure 2), but there are notable areas with very different soils and climates where corn is grown with little information in either one or both loss pathways. All of the following results focus only on this first tier of data, i.e., those with corn as the crop and yield data available.

Additional data in the database that were not used in this analysis includes observations of N loss measurements for which no corresponding corn crop yield data were available. With this the total dataset grew to a total of 495 N_2O and 456 NO_3 records, expanding geographic cover to an additional five locations for N_2O and two locations for NO_3. Nitrogen losses from other crops within the corn-based rotations were also included in the dataset, for a total of 54 NO_3 observations and 166 N_2O observations. Only about half of these non-corn observations included crop yield. None of these data were used in the analysis shown below.

Data description

Data originated from studies with divergent geographic, management, and other characteristics. Only one study measured the two loss pathways simultaneously. For N_2O emissions, data come from 27 studies, plus eight studies reporting loss measurements but not crop yield (see Table 1). In terms of regional distribution, 30% of observations came from Colorado, 21% from Eastern Canada, 17% from Minnesota, 14% from Michigan, and the remaining 18% from eight other U.S. states. Field data were collected between 1994 and 2012, with a median of 2007. Urea was the most commonly used form of fertilizer N (26% of non-control observations), followed by UAN (22%), SUPERU™, Ammonium nitrate, ESN® (each of the latter three with 9% of observations), and polymer coated urea (7%). Other sources were less common. No-till was practiced with 38% of observations, 41% were irrigated,[5] and only 1.6% of the N_2O observations were reportedly in tile-drained fields. More than 95% of N_2O measurements

[5] Note that some "irrigated" systems were only irrigated sparingly to address drought conditions.

covered the entire growing season or more, with 69% of observations monitoring for up to six months of the year, 19% for between six and nine months, and 7% of observations including data from a full year.

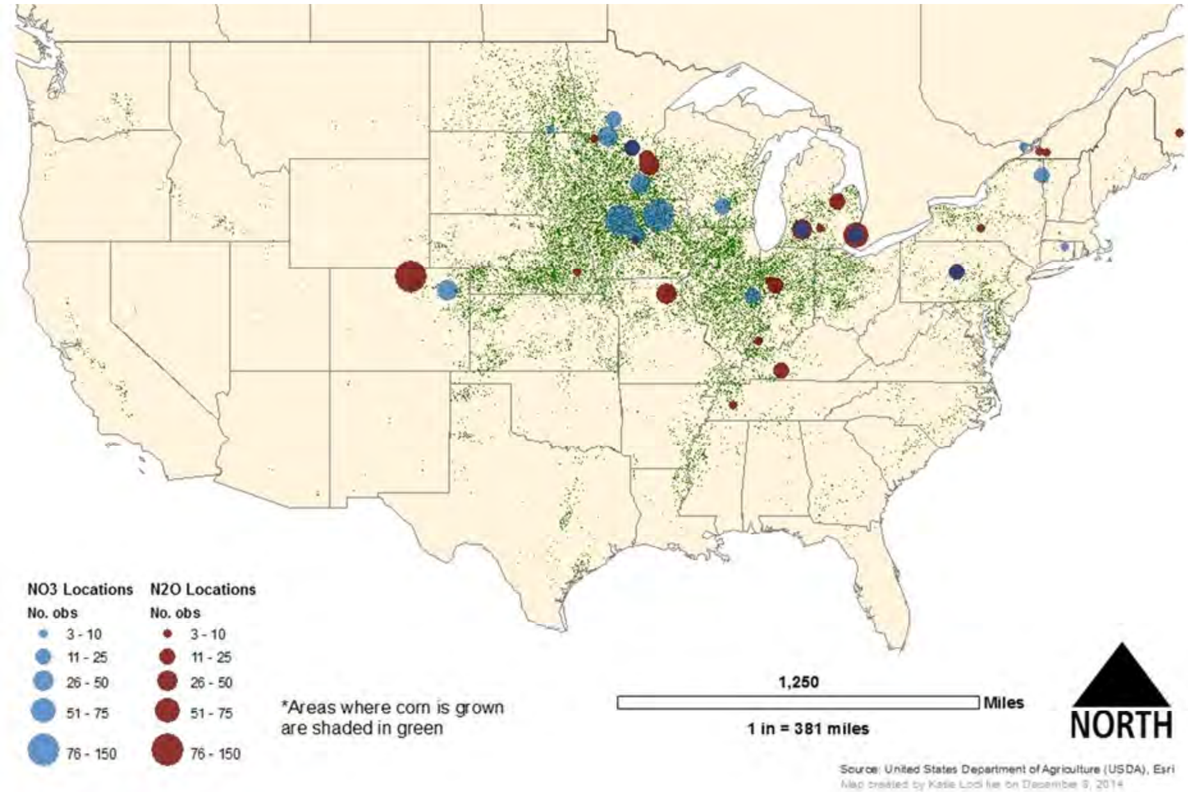

Figure 2. Geographic distribution of agricultural N loss dataset

Data for NO_3 leaching losses come from 22 studies, plus five reporting NO_3 losses but not crop yield (Table 2). The vast majority of these come from Iowa (51%) and Minnesota (21%), with 9% from Colorado and the remaining 19% from six other U.S. states and Eastern Canada. These studies tended to be earlier than those for N_2O, with field data collected between 1980 and 2010 and a median of 1994. UAN comprised 48% of non-control observations, followed by anhydrous ammonia (15%), urea (14%), ammonium nitrate (8%), and ammonium sulfate (8%). Other sources were used in only a small number of cases. Most were conventional tillage (67%), with no-till for only 8% of observations. Nitrate loss measurements were primarily (60%) in fields with tile-drainage, and 26% of observations were in irrigated systems. More than 98% of NO_3 measurements covered the entire growing season or more, with 39% of observations monitoring for up to six months of the year, 53% for between six and nine months, and 7% of observations including data from a full year.

Table 1. Studies used in the N_2O meta-analysis. Location, soil texture, fertilizer management, tillage and water management, and range of N_2O emissions.

Ref.	No of obs.	Location	Year(s)	Soil texture	Tillage	Water mgmt	Fert N source and Inhibitors	Fert N rates (kg N/ha)	Fertilizer timing	Fertilizer placement	N_2O losses (kg N/ha)	Plant N uptake
Adviento-Borbe et al. (2007)	10	NE/USA	2003-05	Silty clay loam	CP	IRR	AN, Combo	0–310	Sp, Sp (PP+SD)	Brd/Inc	1.4–9.2	N
Almaraz et al. (2009)	4	QC/Canada	2002-03	Clay loam	MB, NT	None	AN	0–180	Sp	Bnd	2.3–5.5	N
Dell et al. (2014)	25	PA/USA	2009-12	Silt loam	NT	None	ESN, PiNT, SupU, UAN+AP, UAN, Urea	0–154	SD+St	Bnd, Brd	0.1–2.9	N
Drury et al. (2006)	18	ON/Canada	2000-02	Clay loam	NT, CsT, MB	None	AN	182	SD+St	KI(d), KI(s)	1.3–9.0	N
Drury et al. (2012)	36	ON/Canada	2004-06	Fine sandy loam	NT, CsT, MB	None	PCU, Urea	152	AP, SD+St	Bnd	1.2–9.2	N
Fujinuma et al. (2011)	8	MN/USA	2009-10	Loamy sand	MB	IRR	AA, Urea	37–223	Sp+St	Brd/Inc, KI(d), KI(s)	0.1–1.6	Y
Halvorson and Del Grosso (2012)	14	CO/USA	2009-10	Clay loam	NT	IRR	ESN, SupU, UAN, UAN +AP, Urea	0–202	SD	Bnd SR, Bnd SS	0.2–1.8	Y
Halvorson and Del Grosso (2013)	20	CO/USA	2010-11	Clay loam	CsT, NT	IRR	ESN, Urea, SupU, UAN	202	SD	Bnd SR, Brd	0.3–1.7	Y
Halvorson et al. (2008)	20	CO/USA	2005-06	Clay loam	MB, NT	IRR	Combo	0–246	AP, Sp	Bnd SR	0.2–1. 8	N
Halvorson et al. (2010b)	14	CO/USA	2007-08	Clay loam	NT	IRR	ESN, PCU, SupU, UAN+AP, UAN, Urea	0–246	SD	Bnd SR	0.2–0.9	Y
Halvorson et al. (2010a)	21	CO/USA	2007-08	Clay loam	MB, NT	IRR	ESN, SupU, Urea	0–246	SD	Bnd SR	0.1–2.6	Y
Halvorson et al. (2011)	16	CO/USA	2009-10	Clay loam	CsT	IRR	ESN, UAN+Nf, SupU, UAN+AP, UAN, Urea	0–202	SD	Bnd SR, Bnd SS	0.1–1.7	Y
Hoben et al. (2011)	36	MI/USA	2007-08	Loam, Sand	CP	None	Urea	0–225	PP	Brd	0.3–5.2	N
Maharjan and Venterea (2013)	15	MN/USA	2011-12	Silt loam	Unk	None	Combo, ESN, SupU, Urea	0–180	SD	Bnd MR, Brd/Inc	0.4–6.2	N
Maharjan et al. (2014)	16	MN/USA	2009-10	Loamy sand	CP	IRR	ESN, SupU, Urea, UAN	6–186	PP+St, Sp+St	Brd/Inc	0.2–0.4	Y
McSwiney and Robertson (2005)	18	MI/USA	2001-03	Loam	CP	IRR	Combo, UAN	0–291	Sp (PP+SD)	Brd/Inc, KI	0.02–6.9	N

Mosier et al. (2006)	28	CO/USA	2002-04	Clay loam	MB, NT	IRR	UAN	0–224	PP	KI(s)	0.2–3.6	N
Nash et al. (2012)	6	MO/USA	2009-10	Silt loam	NT, CsT	IRR	Urea, PCU	0–140	AP	Bnd, Brd	1.1–6.1	N
Parkin and Hatfield (2010)	4	IA/USA	2006-07	Silty clay loam	CsT	None	AA, AA+NP, Combo, Combo+NP	125–168	F	KI	5.3–7.0	N
Pelster et al. (2011)	6	QC/Canada	2004	Clay loam	NT, MB	TD	AN	0–160	Sp	Bnd	0.8–2.8	N
Phillips et al. (2009)	2	ND/USA	2008	Clay loam	NT, MB	None	Urea	70	PP, SD	Brd	0.27–0.33	N
Sistani et al. (2011)	14	KY/USA	2009-10	Silt loam	NT	None	AN, ESN, Urea, SupU, UAN, UAN+AP	0–168	SD	Brd	1.0–5.9	N
Smith et al. (2011)	15	IN/USA	2005-07	Silt loam	CsT, CP, NT	None	UAN	0–168	AP, Sp (PP+AP), Sp (PP+SD)	Bnd, KI	0.5–11.2	N
Thornton and Valente (1996)	3	TN/USA	1993	Silt loam	NT		AN	0–252	SD	Brd	1.9–8.5	N
Thornton et al. (1996)	3	TN/USA	1994	Silt loam	NT	None	AA, Urea	0–168	SD	Bnd MR	1.4–13.8	N
Venterea et al. (2010)	12	MN/USA	2006-07	Silt loam	CP	None	AA, Urea	0–146	PP	Brd, KI	0.6–3.4	N
Venterea et al. (2011b)	24	MN/USA	2008-10	Silt loam	MB, NT	None	Urea, PCU, SupU	5–151	SD+St	Brd	0.4–1.1	Y
Bronson et al. (1992)	10	CO/USA	1989-90	Clay loam	MB, CP	IRR	Urea, Urea+NP, Urea+ECC	0–218	SD	KI	0.1–3.4	N
Duxbury and McConnaughey (1986)	3	NY/USA	1981	Silt loam	Unk	None	CN, Urea	0–140	SD+St	KI	0.3–2.5	N
Hernandez-Ramirez et al. (2009)	4	IN/USA	2005-06	Silty clay loam	CP	None	UAN	0–157	SD	KI	4.4–6.9	N
Johnson et al. (2010)	9	MN/USA	2004-06	Silty clay loam	CsT, MB	None	AN, AA	0–150	SD+St	Brd, KI	4.2–6.4	N
Mitchell et al. (2013)	3	IA/USA	2011	Loam	NT	None	UAN	0–225	SD	Bnd SR	1.3–5.1	N
Omonode and Vyn (2013)	8	IN/USA	2011-12	Silt loam	NT, CsT	None	UAN, UAN+NP	200	SD	KI	0.3–16.3	N
Paniagua (2006)	24	MO/USA	2004-05	Silt loam	CP	IRR, SIRR, TD	ESN, Urea	13–299	PP+St	Brd/Inc	2.0–43	N
Smith et al. (2011)	5	IN/USA	2004	Silt loam	CsT, CP, NT	None	UAN	168	AP, PP, Sp (PP+SD)	Bnd, KI	2.9–3.8	N
Venterea et al. (2005)	12	MN/USA	2003-04	Silt loam	CP, MB, NT	None	AA, UAN, Urea	120	PP, SD	Brd, KI	0.4–4.2	N
Zebarth et al. (2008)	8	NB/Canada	2004-05	Silt loam	Unk	None	AN	45–209	Sp (PP+SD)	Bnd, Brd/Inc	1.0–3.3	N

Abbreviations

Tillage Practice: CP - Chisel plow; CsT - Conservation tillage (reduced, strip, ridge, precision, vertical); MB - Moldboard plow; NT - No till; Unk - Unknown

Water Management: IRR - Irrigated; SIRR - Subirrigation; TD - Tile drainage

N source, as well as co-applied nitrification and urease inhibitors: AA - Anhydrous ammonia; AN - Ammonium nitrate; AP - AGROTAIN® PLUS; AS - Ammonium sulfate; CN - Calcium nitrate; Combo -combination; ECC - encapsulated calcium carbide; ESN - ESN®, Environmental Smart Nitrogen; Nf - NITAMIN NFUSION®; NP - Nitrapyrin; PCU - Polymer coated urea; PiNT - a cation-stabilized amine N product; SupU - SUPERU™; UAN - Urea ammonium nitrate

Fertilizer timing: AP - At planting; F - Fall; PP - Preplant; SD - Side dress; Sp - Split; St - Starter

Fertilizer placement: Brd – Broadcast; Brd/Inc - Broadcast and incorporated; Bnd - Banded; Bnd MR - Banded midrow; Bnd SR - Banded siderow; Bnd SS - Banded subsurface; KI - Knife injected; KI(d) - Knife injected deep; KI(s) - Knife injected shallow

Table 2. Studies used in the NO_3 meta-analysis. Location, soil texture, fertilizer management, tillage and water management, and range of NO_3 leaching losses. Records listed below the bold solid line do not include crop yield.

Ref.	No of obs.	Location	Year(s)	Soil texture	Tillage	Water mgmt	Fert N source and Inhibitors	Fert N rates (kg N/ha)	Fertilizer timing	Fertilizer placement	NO_3 losses (kg N/ha)	Plant N uptake
Bakhsh et al. (2002)	24	IA/USA	1993-98	Silty clay loam	CP, NT	TD	UAN	93–195	PP, SD	KI	3–46	N
Bakhsh et al. (2010)	10	IA/USA	2001-05	Silt loam	CP	TD	UAN	168	AP, PP	KI, LCD	0.4–27.5	N
Basso and Ritchie (2005)	12	MI/USA	1994-99	Loam	MB	None	Urea	0–120	Sp (PP+SD)	Brd	11–89	N
Guillard et al. (1999)	6	CN/USA	1995-96	Fine sandy loam	MB	None	AN	34–196	PP, SD, Sp (PP+SD)	Unk	4–61	N
Heilman et al. (2012)	49	IA/USA	1990-03	Silty clay loam	CP, RT, MB, NT	TD	AA, UAN	110–202	PP, Sp (PP+SD)	KI, LCD	0.4–119	N
Helmers et al. (2012)	35	IA/USA	1990-93	Clay loam	CP	TD	UAN	0–224	SD, AP	KI	4–88	N
Jayasundara et al. (2007)	2	ON/Canada	2003	Silt loam	NT, MB	None	UAN, Urea	60–150	AP, SD	Brd/Inc, KI	1.9–2.1	Y
Jaynes (2013)	6	IA/USA	2006, 2008	Clay loam	CP	TD	UAN	134–157	SD, Sp	KI	21.6–46.3	N
Jaynes and Colvin (2006)	8	IA/USA	2002, 2004	Clay loam	CP	TD	UAN	69–199	SD, Sp	KI	10.7–36.9	N
Jaynes et al. (2001)	6	IA/USA	1996, 1998	Loam	MB, CP	TD	AA, UAN	57–202	PP, SD	KI	37–61	Y
Jemison and Fox (1994)	9	PA/USA	1988-90	Silt loam	CP	IRR	AN	0–200	AP	Brd	23.6–133	N
Kucharik and Brye (2003)	12	WI/USA	1995-00	Silt loam	CP, NT	None	AN	0–180	PP	Brd	3.2–102	Y
Lawlor et al. (2008)	49	IA/USA	1990-04	Clay loam	CP	TD	UAN	0–252	AP, PP	KI	0–88	N
Lawlor et al. (2011)	16	IA/USA	2001-04	Clay loam	CP	None	AqA	168–252	F, PP, SD	KI	25–86	N
Maharjan et al. (2014)	16	MN/USA	2009-10	Loamy sand	CP	IRR	ESN, SupU, Urea, UAN	5.6–185.6	PP+St, Sp+St	Brd/Inc	1.4–47.4	Y
Porter (1995)	36	CO/USA	1991-93	Silty clay loam	Unk	IRR	AS	0–376	Unk	Brd/Inc	0.8–24.5	Y
Prunty and Greenland (1997)	4	ND/USA	1993, 1995	Loamy sand	Unk	IRR	UAN, Urea	82–136	Sp (PP+SD)	Bnd SS	3–118	N
Randall and Vetsch (2003)	24	MN/USA	1987-88, 1990-93	Clay loam	CsT	TD	AA, AA+NP	150	F, PP, Sp (PP+SD)	KI	2–122	Y
Randall and Vetsch (2005)	24	MN/USA	1994-99	Clay loam	CsT	TD	AA, AA+NP	135	F, PP	KI	4–63	Y
Sexton et al. (1996)	16	MN/USA	1991-92	Sandy loam	MB	IRR	Urea	20–180	Sp+St	Brd	15–141	Y

Sogbedji et al. (2000)	17	NY/USA	1992-94	Clay loam, Loamy sand	MB	None	UAN	22–134	AP, SD+St	Bnd, KI	5.9–34.9	Y
Tan et al. (2002)	12	ON/Canada	1998-00	Clay loam	MB	TD	AN	0–128.8	Sp	Brd	2.5–47.9	N
Walters and Malzer (1990)	27	MN/USA	1980-82	Sandy loam	Unk	IRR	Urea, Urea+NP	0–180	AP	Brd, Brd/Inc	6.9–140.7	Y
Kalita et al. (2006)	16	IL/USA	1992-00	Silty clay loam	CsT	TD	Unknown	0–254	PP	Brd	3.3–72.6	N
Kaluli et al. (1999)	5	QC/Canada	1994	Sandy loam	CP	TD, SIRR	AN	0–270	SD + St	Brd	2.6–21.9	Y
Toth and Fox (1998)	9	PA/USA	1991, 1994	Silt loam	CP	IRR	AN	13–213	PP+St	Brd	4.5–91.7	N
Zhu and Fox (2003)	6	PA/USA	1997, 1999	Silt loam	NT, CP	None	AN	0–200	PP+St	Unk	8–135	N

<u>Abbreviations</u>

Tillage Practice: CP - Chisel plow; CsT - Conservation tillage (reduced, strip, ridge, precision, vertical); MB - Moldboard plow; NT - No till; Unk - Unknown

Water Management: IRR - Irrigated; SIRR - Subirrigation; TD - Tile drainage

N source, as well as co-applied nitrification and urease inhibitors: AA - Anhydrous ammonia; AN - Ammonium nitrate; AS - Ammonium sulfate; AqA - Aqueous ammonia; ESN - ESN®, Environmental Smart Nitrogen; NP - Nitrapyrin; SupU - SUPERU™; UAN - Urea ammonium nitrate

Fertilizer timing: AP - At planting; F - Fall; PP - Preplant; SD - Side dress; Sp – Split; St - Starter

Fertilizer placement: Brd - Broadcast; Brd/Inc - Broadcast and incorporated; Bnd - Banded; Bnd SS - Banded subsurface; KI - Knife injected; LCD - Localized compaction and doming; Unk - Unknown

Fertilizer N rate yield, nitrous oxide, and nitrate

Fertilizer N rate effects on yield and N losses were determined from experiments that explore multiple rates, using rate response on a continuous basis. Compared to placement, source, and timing, fertilizer rate effects on N losses have been studied in far more field experiments and are the subject of numerous reviews and syntheses. In corn studies where crop yield was also reported (i.e. the database collected for this study), 129 (32%) of the N_2O observations and 162 (41%) of the NO_3 observations were from experiments that measured at least three fertilizer rates including a control.

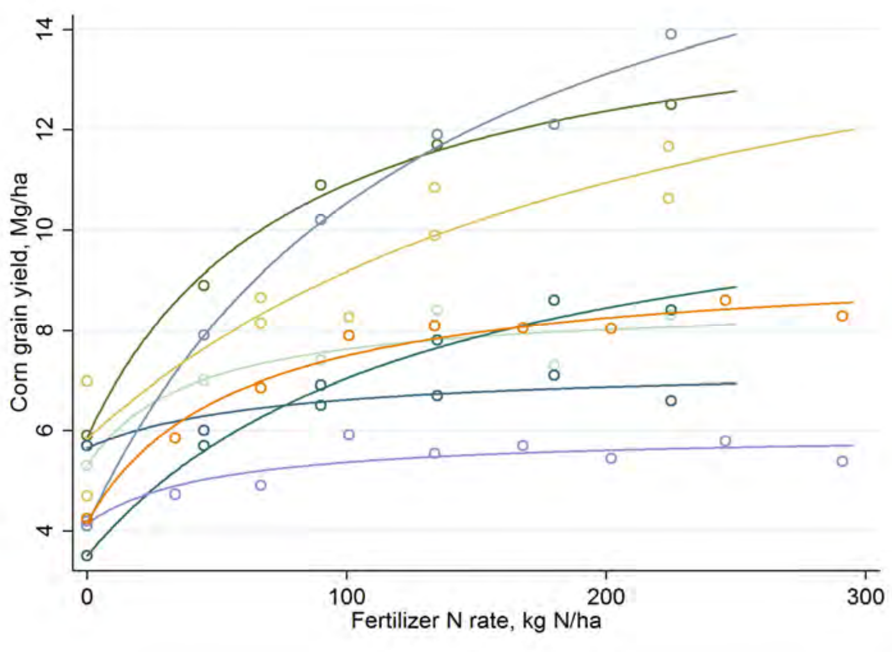

Figure 3. Corn grain yield response to fertilizer N rate in eight different example site-years

With these observations, models of each individual site-year were generated to look at the responses of corn grain yield, N_2O emission, and NO_3 leaching as compared to fertilizer N rate. Figure 3 shows an example of corn grain yield response of eight different site-years. Not surprisingly, yield increased with fertilizer N rate over a certain range, and then leveled off. Yield response for each site-year generally followed a Michaelis-Menten type saturation curve, but with an added y-intercept (instead of starting at zero). The equation is

(3) $$Y_{fert} = Y_0 + \frac{Y_{inc}[N_{fert}]}{K_m}$$

where Y_{fert} is yield at a given fertilizer rate, Y_0 is yield without fertilizer N, Y_{inc} is the maximum yield increase possible, N_{fert} is N fertilizer rate, and K_m is a saturation constant equal to the fertilizer rate at which half of the maximum yield increase has been achieved. There were some significant differences between locations and some variation between years at the same location. The yield in plots without fertilizer N varied (i.e., different y-intercepts), and so did the yield response to fertilizer N additions (affecting the shape of the curve).

Nitrous oxide emissions responded differently to fertilizer N rate than corn grain yield. For these data, the best relationship between rate and losses was exponential (see Figure 4 for an example). In some locations, the N2O emissions remained much lower than in others, even at high fertilizer application rates. As with yield, the baseline (or emissions in the case of zero N fertilizer) varied by location and by year within the same location. The rate of increase also varied somewhat. Nitrate, on the other hand, was highly variable, and while NO_3 losses did increase with rate, there was not always a clear relationship. The best fit for this relationship was linear (Figure 5).

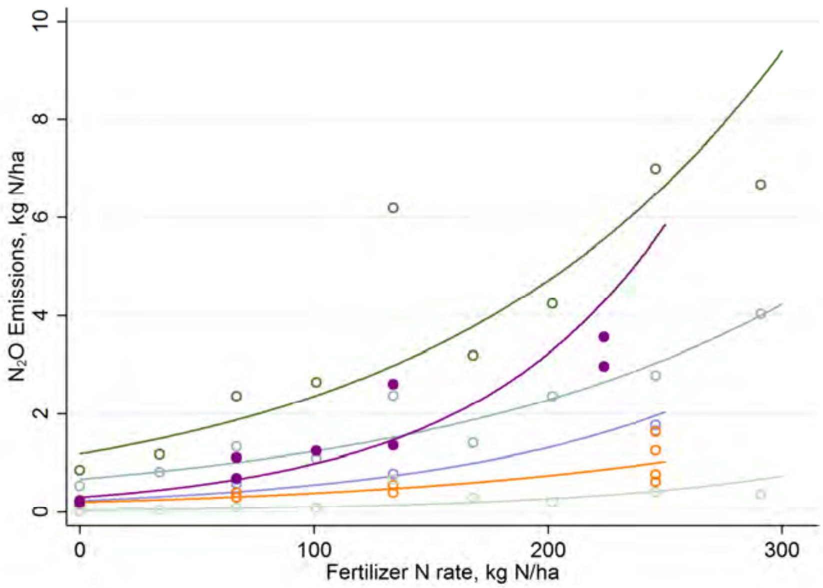

Figure 4. Nitrous oxide emission response to fertilizer N rate in six example site-years

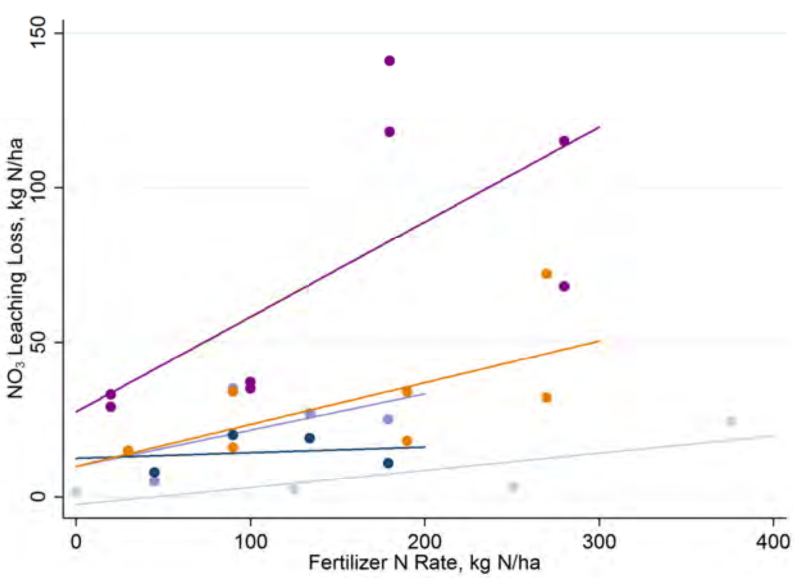

Figure 5. Nitrate leaching response to fertilizer N rate in five example site-years

Side-by-side comparisons of 4R management factors

The effects of 4R management on N losses to N_2O and NO_3 can be assessed from these data with at least two different approaches. First, all available side-by-side comparisons (i.e., experiments where the only difference was the specific management practice) can be summarized and used to calculate the average effect (or effect size) of that treatment. Because other factors are not considered, conclusions about these practices based on this standard meta-analysis method should be limited to the locations and other characteristics of the specific experiments. In the corn fertilizer management field data, effect sizes are not representative of the climatically diverse overall corn producing area in North America, nor do they characterize the varied management in tillage, crop rotations, drainage, and other factors. The second approach involves using data from diverse experiments to create an empirical model that considers all available climatic and management factors to test for effects of management differences that are not limited to the side-by-side experiments. The second approach, a multi-level model, is described in the following section.

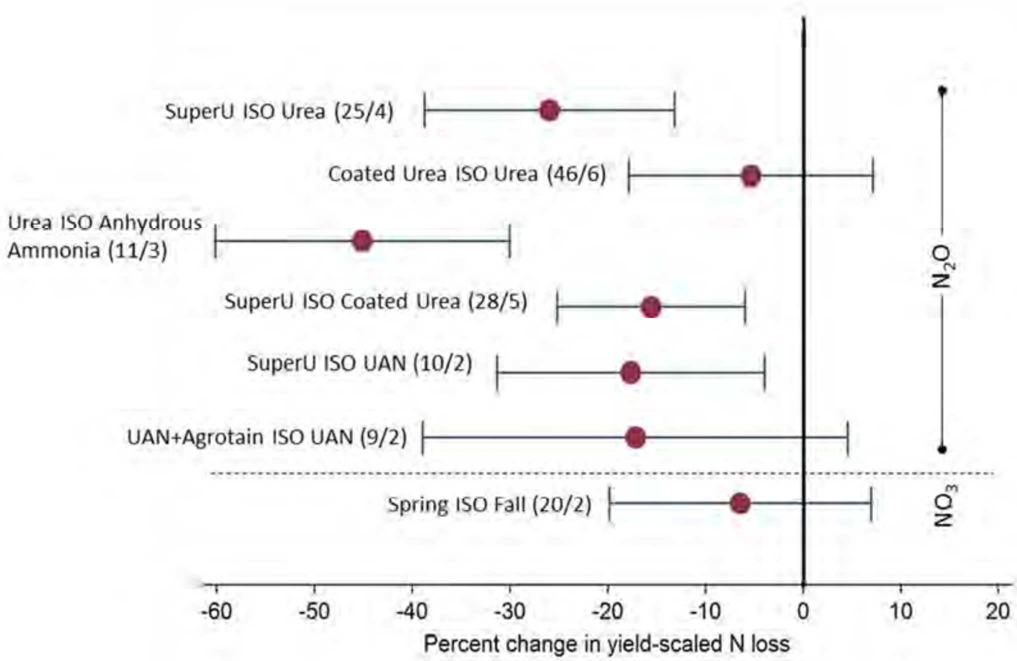

Figure 6. Effect sizes of N_2O and NO_3 losses from selected fertilizer management treatments, yield-scaled percent change with 95% confidence intervals. ISO = "Instead of" and values in parentheses are (# of comparisons/# of locations).

We calculated effect sizes of source and timing for treatments where the data contained at least nine side-by-side comparisons from at least two different locations. Effect sizes are weighted means, with the contribution of individual observations affected by the number of observations per location. Reported in this manner, effect sizes do not consider factors other than the specified contrasted 4R management treatments. Therefore, the interpretation would be that the reported effect sizes are the average response to the specific treatment across all locations and in the given fertilizer source, crop rotation, and other characteristics of the studies supplying the data.

For the number of side-by-side comparisons for all available source, placement, and timing contrasts, see Appendix B. Of the source, placement, and timing data, the best available information compares N_2O losses between fertilizer sources. Effect sizes are presented in Figure 6. Replacing anhydrous ammonia with urea in Minnesota and Tennessee reduced yield-scaled N_2O emissions by an average of 45% (Fujinuma et al. 2011; Thornton et al. 1996; Venterea et al. 2010). The combination of data from Colorado, Pennsylvania, and Minnesota found that SUPERU™ reduced N_2O emissions by 26% when compared to urea (Dell et al. 2014; Halvorson and Del Grosso 2012; Halvorson and Del Grosso 2013; Halvorson et al. 2010a; Maharjan and Venterea 2013; Venterea et al. 2011b). The loss reduction with SUPERU™ was most likely owing to nitrification and urease inhibitors within its formulation. SUPERU™ also reduced emissions by 15% when compared to polymer-coated urea in Minnesota and Colorado (Halvorson et al. 2010b; Venterea et al. 2011b) and to UAN in Colorado and Kentucky (Halvorson and Del Grosso 2012; Halvorson and Del Grosso 2013; Halvorson et al. 2010b; Halvorson et al. 2011; Sistani et al. 2011). Data from Minnesota, Ontario, and Missouri found that emissions from polymer-coated urea were not significantly different from urea (Drury et al. 2012; Nash et al. 2012; Venterea et al. 2011b), and AGROTAIN® PLUS also did not generate consistent emission reductions when added to UAN in Colorado and Pennsylvania (Dell et al. 2014; Halvorson and Del Grosso 2012; Halvorson et al. 2010b; Halvorson et al. 2011).

Various fertilizer placement and timing options have also been tested for N_2O emission impact, although none were replicated in more than one study within the database. Nitrous oxide emissions have been reduced by deeper placement of ammonium nitrate on a clay loam soil in Ontario (Drury et al. 2006), and by deeper placement of anhydrous ammonia on a loamy sand soil in Minnesota (Fujinuma et al. 2011). In contrast, sub-surface placement of ESN lost more N2O than surface band placement in both no-till and strip-till corn on a clay loam soil in Colorado (Halvorson and Del Grosso 2012; Halvorson et al. 2011). Band placement also increased emissions in contrast to broadcast for a number of different sources in both Colorado and Minnesota (Halvorson and Del Grosso 2013; Maharjan and Venterea 2013). Even though there were insufficient timing observations to calculate effect sizes, individual study results from North Dakota and Indiana found that changing from early to late spring application (Phillips et al. 2009) and splitting fertilizer into two applications (Smith et al. 2011) had little impact on either area or yield-scaled N_2O emissions in corn.

The combined effect size of nitrate leaching losses calculated for spring anhydrous ammonia application as opposed to fall was not significantly different from zero (Figure 6), when considering jointly a Minnesota study that measured a 14% reduction (Randall and Vetsch 2005) and an Iowa experiment that found no impact (Lawlor et al. 2011). Too few observations were available to calculate any multi-study effect sizes for other timing changes or for the other 4R management practices, so the information available is limited to individual experiments. Splitting fertilizer application during the growing season did not affect NO_3 leaching losses in Iowa (Jaynes 2013; Jaynes and Colvin 2006). Individual experiments found no significant difference in NO_3 leaching between SUPERU™ and polymer-coated urea (Maharjan et al. 2014), or with nitrification inhibitors added with either urea (Walters and Malzer 1990) or anhydrous ammonia (Randall and Vetsch 2005). Tests of fertilizer N placement in Iowa and Minnesota found no significant effects on NO_3 leaching losses (Bakhsh et al. 2010; Walters and Malzer 1990).

Hierarchical models of nitrous oxide and nitrate loss

To combine data from all the different studies into a large dataset for analysis, it was necessary to develop something more complex than a simple regression model. Multi-level, or hierarchical, regression models were used to figure out relationships at each location while allowing each location to explain the overall effect. These models were developed for both nitrous oxide emissions and nitrate leaching losses.

Nitrous oxide model results

The first nitrous oxide model includes all observations, and a restricted model includes only observations that fall within the typical fertilizer N application rates of 150 to 250 kg N/ha (Table 3). For both models, the dependent variable is the log of emissions and the significant explanatory variables include rate, yield, July temperature, nitrification inhibitors, injected fertilizer placement, and side dress fertilizer application timing. Interactions between variables were not significant in the model.

Table 3. Hierarchical (multi-level) regression models of N_2O emissions in North American corn cropping systems. The dependent variable is the natural log of N_2O emissions (kg N/ha). Model 2 restricts observations to those with fertilizer N application rates between 150 and 250 kg N/ha.

	Model 1 (n=408, 19 clusters)		Model 2 (n=267, 17 clusters) [N Rates: 150–250 kg/ha]	
	Coeff	Std Err	Coeff	Std Err
N Rates (kg N/ha)	0.0059***	0.0005	0.0035*	0.002
Yield (Mg grain/ha)	0.048***	0.013	0.061***	0.019
July Temp (°C)	0.224**	0.094	0.256**	0.101
Nitrification inhibitors	-0.330***	0.086	-0.365***	0.091
Inject fertilizer	0.520***	0.073	0.421***	0.102
Sidedress fertilizer	-0.372***	0.078	-0.503***	0.100
Constant	-5.784***	2.137	-6.209***	2.307
ψ – variance between clusters	0.512		0.443	
θ – variance within clusters	0.265		0.271	

*, **, and *** indicate statistical significance at the 0.10, 0.05, and 0.01 levels, respectively.

The model shows a positive effect of rate, yield, July temperature, injected fertilizer, and a negative effect of nitrification inhibitors and side-dressed fertilizer. Coefficients can be interpreted as the percent change in emissions from a change in one unit (for N rate, yield, and temperature) or from adoption of a practice (for nitrification inhibitors, injecting fertilizer, and side-dress timing of fertilizer). For example, we can see that each additional 10 kg of N/ha contributes to an average 5.9% increase in N_2O emissions. This non-linear response means that the absolute response depends on the baseline loss rate). Also, each one degree rise in average July temperature results in a 22.4% increase in emissions. The total variance between clusters (psi) and within clusters (theta) are also shown in Table 3. A significant amount of total variance occurs between clusters, indicating that using location as a group explains much of the model error. Therefore, a significant amount of the variance in the data is explained by differences between research sites.

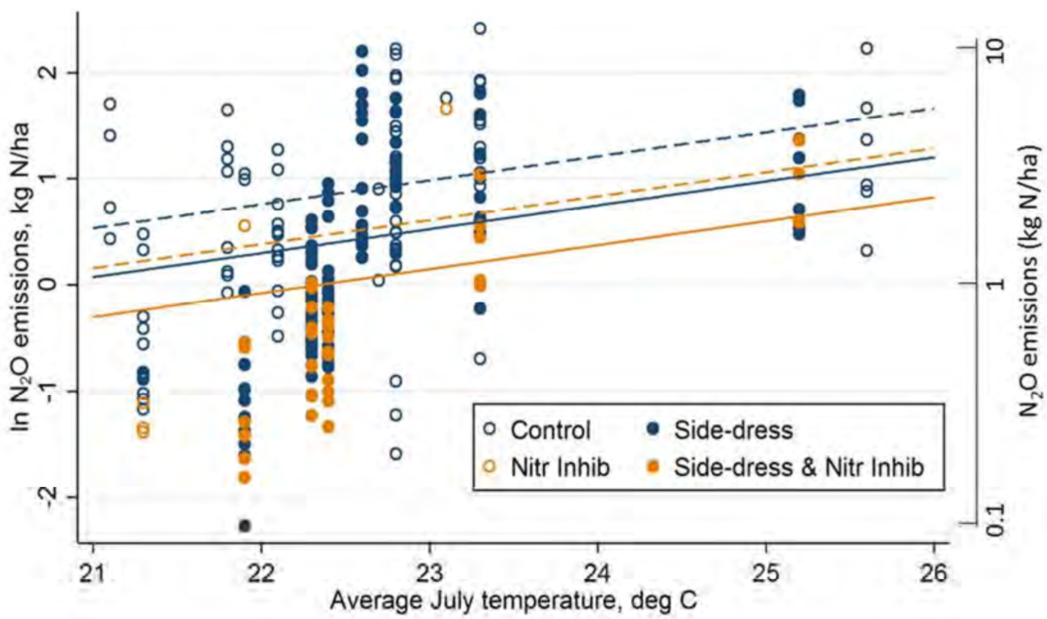

Figure 7. Nitrous oxide emissions response to July temperature, nitrification inhibitors, and side-dress timing of N fertilizer. Model results and observations come from corn/maize field experiments in North America for which yield data are available, and are limited to those with fertilizer N application rates between 150 and 250 kg N/ha.

When restricting the model to observations at typical fertilizer N application rates, the impact of rate decreased somewhat, so that growing season temperature and three fertilizer management practices were more important in comparison. Injecting fertilizer had a positive effect on N_2O emissions, although this seemed to be complicated with fertilizer source (since only certain sources were injected). Figure 7 illustrates the response of N_2O to temperature, nitrification inhibitors, and side-dress fertilizer application, with the lines showing the model results and individual observations depicted as circles. For example, at the average July temperature of 23 degrees C, side-dressing fertilizer (instead of pre-plant) and using nitrification inhibitors reduced N_2O losses from 2.7 kg N/ha to 1.2 kg N/ha.

Nitrate model results

The NO_3 models were also developed for both full and restricted (110-270 kg fertilizer N/ha) datasets. These models (Table 4) found a positive effect of rate, yield, annual precipitation, and aqueous ammonia compared to other sources. As with N_2O, interactions between variables were not significant in the model. Higher fertilizer N application rates as well as less crop N uptake (associated with lower yield) are also associated with increased nitrate losses. On average, at typical application rates, 10% of each additional unit of fertilizer N was lost via NO_3 leaching. Figure 8 shows actual observations (circles) and the modeled NO_3 loss response (lines) for typical fertilizer application rates, that is, the restricted model.

Table 4. Hierarchical (multi-level) regression models of NO_3 leaching losses in North American corn cropping systems. Dependent variable in both models is NO_3 losses (kg N/ha). Model 2 restricts observations to sites with fertilizer N application rates between 110 and 270 kg N/ha.

	Model 1 (n=396, 16 clusters)		Model 2 (n=275, 16 clusters) [N Rates – 110–270 kg/ha]	
	Coeff	Std Err	Coeff	Std Err
N Rates (kg N/ha)	0.080***	0.017	0.102***	0.036
Yield (Mg grain/ha)	1.801***	0.595	2.082***	0.775
Annual Precip (mm)	0.079***	0.009	0.091***	0.011
Irrigated (0 or 1)	--	--	15.746	10.905
Soil Carbon (g/kg soil)	-1.135***	0.344	-1.355***	0.477
Yr of study (mean=0)	-1.199***	0.340	-1.016***	0.387
Aqueous ammonia fertilizer source	18.193***	6.082	18.293***	6.559
Constant	-34.537***	9.996	-47.634***	16.042
ψ – variance between clusters	188.4		217.5 .	
θ – variance within clusters	389.1		410.6	

*** indicates statistical significance at the 0.01 level

Greater amounts of precipitation resulted in higher leaching losses. For example, a given year or a location that had 100 mm more precipitation than a drier year or location experienced an average of 9 kg N/ha more NO_3 lost via leaching. Irrigation increased nitrate losses in a similar manner to a wet climate. Therefore, the important relationship between irrigation and precipitation dictated inclusion of irrigation in the final restricted model, even though the irrigation variable was only significant at p=0.15.

The end result was that of much more accurate modeled estimates for the irrigated observations (which were located in low precipitation regions that would otherwise suggest low loss rates).

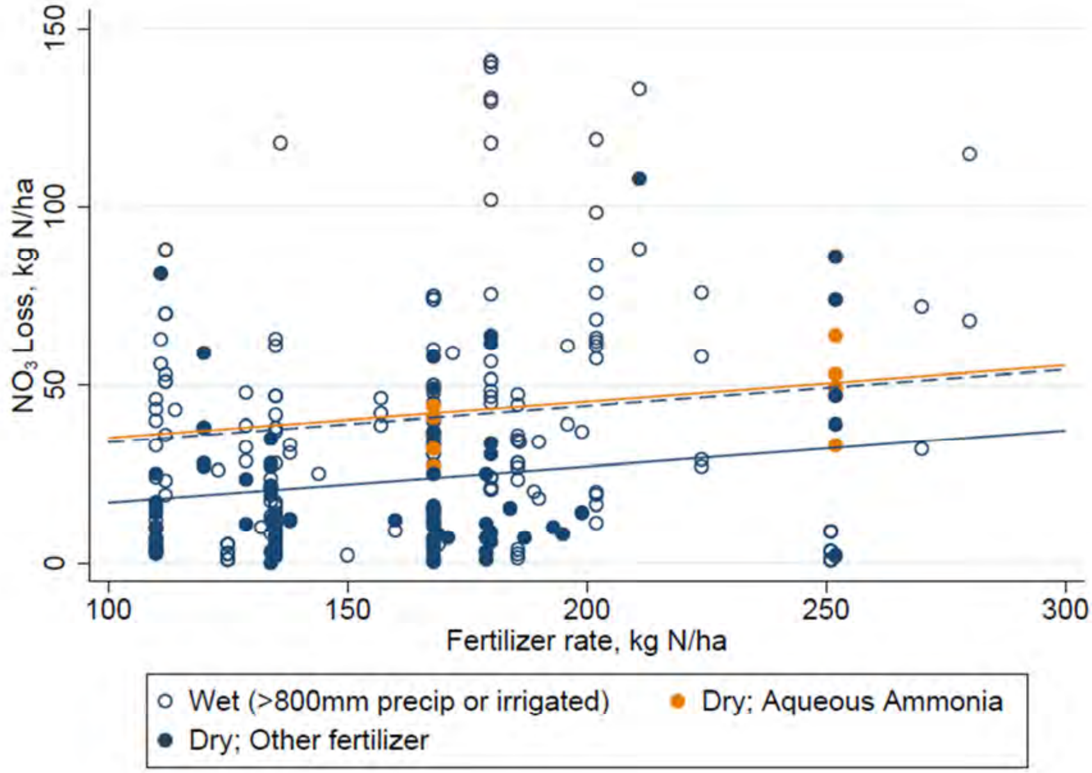

Figure 8. Nitrate loss response to fertilizer N rate, precipitation or irrigation, and fertilizer source. Model results and observations come from corn/maize field experiments in North America for which yield data are available, and are limited to those with fertilizer N application rates between 110 and 270 kg N/ha. Observations and modeled lines are separated into two precipitation categories; those denoted “wet” climate were irrigated or had annual precipitation greater than 800 mm, and those that are “dry” received less than 800 mm in annual precipitation.

While increased precipitation or other water addition increased losses, soil carbon on the other hand (itself often associated with wetter climates) had a negative impact. The model shows that on average, each additional 10 g/kg of soil C (or 1%) reduced average NO_3 loss by 14 kg N/ha. Since leaching rates are higher in more coarse-textured soils (e.g., sand) that also do not retain as much organic matter, the impact of soil C may be related to texture, but soil texture as a factor on its own was not significant in any models tested.

The only management practice with any significant impact was N fertilizer source, with increased NO_3 leaching (an average of 18 kg N/ha) when aqueous ammonia was used in place of other fertilizer sources – primarily UAN in these data. Since aqueous ammonia comprised only a small number of total observations, the effect on nitrate losses due to source was tested in a model further restricted to the two locations that had aqueous ammonia observations (model not shown). This test confirmed that losses for aqueous ammonia were significantly higher than from UAN in these locations.

Finally, the year of study also had a negative impact, suggesting that losses were affected by some undocumented or otherwise uncaptured change in management, crop variety, location of study, or measurement technique. Over the entire time period of 30 yrs, this resulted in difference of about 30 kg N/ha. It should be noted that this is most likely a factor of the types of studies that were conducted in different time periods, so should not be interpreted as indication that losses are actually decreasing over time. The models find that grouping the data by location explains a significant amount of the variability (i.e., ψ, the variance between clusters, accounts for 35% of the variance). However, in this case, we also see a high degree of variability remaining within the clusters, indicating that within locations there was a large amount of scatter in the data.

Discussion

Generalizing the impacts of 4R management activities across regions and varied management systems is not possible with individual studies or even standard meta-analysis effect sizes. Therefore, the combination of data from many different studies with a hierarchical model contributes some valuable insights. The hierarchical models allow us to assess the N_2O and NO_3 loss implications of different fertilizer management activities, not only those compared within specific studies, but also those that vary between studies by controlling for climate, soil, and other management factors. With the available data, these empirical models were not able to detect significant interactions between relevant factors. However, any results should be interpreted keeping in mind that temperature, soil C, soil texture, and other factors in these systems are related to one another, and such relationships are bound to have implications for N cycling.

While insufficient input data for our models make it impossible to develop broader conclusions for some of the 4Rs (or aspects thereof), other activities show clear results. For example, our models show that using nitrification inhibitors and delaying fertilizer application by side-dressing both reduce N_2O emissions. Further, replacing aqueous ammonia with UAN or other sources – at least in certain locations – reduced NO_3 losses by an average of 18 kg N/ha. Also, greater levels of soil C reduce NO_3 losses. Most of these factors were not tested (or could not be tested) in side-by-side comparisons, and when they were assessed, results from individual studies tended to be uncertain. Therefore, by including data from multiple experiments that represent broad climatic and soil conditions, the models begin to find results that have policy-relevance.

As expected (and because it was tested in the majority of studies), fertilizer N rate was one of the most important factors affecting N_2O emissions and NO_3 losses, both on an area-basis and when yield-scaled. Some loss reductions without corresponding yield declines are likely achievable through adjusting current fertilizer application rates. This is because of the exponential response of N_2O emissions to increasing rates that corresponds to a linear response of NO_3 and a saturation curve response of yield. If these relationships can be established for individual regions or fields with given soil C, temperature, and expected moisture regimes, it should be possible to optimize rates for the desired yield and N loss results.

While yield, nitrous oxide emissions, and nitrate leaching losses all increased with fertilizer N rate, there were also some other important differences. For example, the typical losses from nitrous oxide emissions were between 1 and 2 kg N/ha while the typical losses from nitrate leaching were between 20 and 50 kg N/ha. Marginal damage cost estimates are similar for both N_2O and NO_3, around $7 to $20 per kg of N (Compton et al. 2011; van Grinsven et al. 2013), with projected abatement costs for one effort (the Chesapeake Bay) in a comparable range ($8-15 per kg N). Therefore, with average NO_3 losses exceeding those of N_2O, and even though there are some unquantified tradeoffs, nitrate may be more of an environmental concern than nitrous oxide emissions in the average system.

Fertilizer source comparisons suggest the following order of preferred sources for minimizing N_2O emissions within the studies included, with the best options listed first: SUPERU™ > UAN+ AGROTAIN® PLUS ≥ UAN ≥ polymer-coated Urea ≥ Urea > Anhydrous Ammonia. The multi-level model finds that nitrification inhibitors (such as those found in SUPERU™) significantly lowered N_2O emissions. Given this consistency, we can conclude that this would apply across many, although perhaps not all, of the broader climatic and management systems for corn production in North America. With very minimal side-by-side comparisons testing fertilizer source impacts on nitrate losses, and in fact, little variation in fertilizer sources overall in these experiments, the only source that had any significant impact on NO_3 was aqueous ammonia. That this uncommon fertilizer source is associated with higher losses has little practical implication, however, since the baseline use is very low. Further testing is therefore necessary to determine if there may be a similar pattern in terms of both N_2O and NO_3 losses when it comes to fertilizer source.

In side-by-side comparisons and traditional meta-analysis, timing and placement do not have sufficient data to make any conclusions as to their impact on either N_2O or NO_3 losses. However, side-dressing fertilizer, or applying while a crop was growing, was found to have significant N_2O emission reduction potential within the hierarchical model. This finding is in agreement with expectation and the theory that delaying fertilizer N availability in mineral form until needed by the crop is optimum for both production and environmental reasons.

Temperature, soil C, and precipitation are also important in certain cases, but whether they can be managed is another question. Our model finds a negative relationship between NO_3 losses and soil organic matter (i.e., soil C), a soil characteristic that can be managed with tillage, residue return, and cover crops, among other things. However, one potential problem with increasing soil C for the sake of reducing NO_3 losses is that N_2O emissions tend to be higher in soils with more organic matter. There is thus the possibility of a trade-off between the two N loss pathways. The greater rates of N loss associated with temperature and precipitation may mean that efforts to reduce losses may be most effective, or most needed, in warmer and wetter regions. This information could be useful in targeting policy or programs.

For NO_3 losses, losses at the field scale may not always translate into NO_3 pollution of ground or surface waters, as off-site or field-border management such as riparian buffers, constructed wetlands, and perennial filter strips can be used to remove NO_3 from runoff or leachate water (Fennessy and Craft 2011). However, these also tend to generate indirect N_2O emissions (Fisher et al. 2014). Much research and numerous water quality trading protocols study and implement these management practices. The three main mechanisms for removal in these systems are plant uptake, microbial N immobilization into soil organic matter, and denitrification to N_2O or N_2 (Matheson et al. 2002 and Hefting et al. 2005, cited in Iqbal et al. 2015). Different soils and vegetation systems can vary in N gas product ratios (i.e., N_2O/N_2); N_2O fluxes decrease with greater levels of potentially mineralizable carbon (C). While efforts could be made to encourage complete denitrification to N_2 (thus reducing the proportion of N_2O released), the potential environmental tradeoffs of air quality issues for the sake of improved water quality could be significant. IPCC GHG protocols include indirect N_2O emissions of 22.5% of all applied N in wet or irrigated areas to account for this off-site denitrification. In addition, plant and SOM uptake of NO_3 could reach a saturation point so that losses are reduced only in the short term (Iqbal et al. 2015; Mitchell et al. 2015). Therefore, in-field NO_3^- loss reductions are an essential front line in the effort to reduce negative environmental impacts to water and air quality.

Gap Analysis

It is difficult to derive accurate estimates of anticipated yield impacts and N loss reductions for many of the recommended 4R fertilizer management practices, as these questions have either not been sufficiently tested in field experiments or the data have not been published for various reasons. Key corn-producing regions in North America are also missing from the available field data. In addition to the possible treatment-response bias within the data that are available (a problem not uncommon for meta-analysis), the paucity of information means that while we may have an idea of directionality, it is difficult to design policy and programs without better quantification. For example, which practice will achieve the most N_2O loss reductions in a specific watershed, without negative yield or NO_3 leaching consequences? To better target future research, it is advisable to start with better understanding relationships between N_2O and NO_3 losses in different soils and climates. After that, field research can be directed to key regions and farming systems, to test practices that could be reasonably adopted by producers in those regions.

Future field experiments on N losses, and the presentation thereof, need to be more standardized and provide additional measurements in order to better advise biophysical models and policy. First, losses of NO_3 and N_2O should be co-measured to ensure relationships are clearly understood. This could be done efficiently by adding one or the other to field experiments for which management is already being compared for one loss pathway. Second, more complete data on N pools and transformations are needed. Data on crop N uptake and pre-plant available soil N in the current study were seriously lacking, so that N excess could not be calculated with the existing field research. In cropping systems, we tend to assume that crop plants have first priority for available N. Therefore, the relationship between absolute N rate and N losses will likely show more variability and noise than those that relate N excess to losses. The contrast of the two variables is therefore valuable. These relationships are important to understand before determining the overall loss response to rate, when controlling for other factors. And finally, an indication of variability within the data collected needs to be reported. Because these data are not generally reported, current models cannot (and do not) include variability in final loss estimates.

Measures of Excess Nitrogen

If indeed soil sampling for available nutrients is beneficial for management, more accurate predictions of crop yield and N loss should be achieved with measures of N excess as opposed to simple fertilizer N application rates. For this reason, we set out to calculate N excess in two different ways: 1) N fertilizer applied minus aboveground N uptake, and 2) N fertilizer plus pre-plant available N minus aboveground N uptake. Very few studies recorded available N (NO_3 and NH_4) at either pre-plant or side-dress dates; only 11.3% of N_2O observations and 5.3% of NO_3 observations. Therefore, the second equation for determining N Excess was not feasible with the data limitations. Data limitations also contributed to high level of uncertainty in estimations of aboveground N uptake. Complete whole plant data were available for 21% of N_2O observations and 24% of NO_3 observations. Estimations calculated from grain N uptake, total aboveground biomass, and total grain yield were used to derive N uptake for remaining observations. Uncertainty was therefore introduced in numerous places: in estimates of N harvest index, in estimates of whole plant N content (which ranges from 0.5% to 2.0% in the samples recorded in the database), and in estimates of harvest index. This high uncertainty – caused by lack of sufficient data – meant that the calculated N excess did not improve any relationships (and in fact was worse) for N losses or yield.

Interactions between NO_3 and N_2O

Regarding both NO_3 and N_2O losses, what can we say about unintended shifting of N loss across forms? With very little overlap in the field data reporting losses in both pathways, it is nearly impossible to make any measurement-based conclusions about trade-offs or interactions between NO_3 and N_2O losses as affected by different management practices, soil type, or climate. Only one study reported both types of loss, and the systems for which NO_3 and N_2O were monitored are also very different. For example, most NO_3 but only a few N_2O observations were reportedly in tile drained fields, far more N_2O observations were in no-till systems, and the primary-focus fertilizer management practices differed between NO_3 and N_2O studies.

Therefore, the only currently plausible method of comparing and contrasting loss responses of both to different treatments may be through process-based models that have been calibrated and validated as best as possible to the soil, climate, crop, and management conditions. Unfortunately, COMET-Farm one of the most commonly used of these models is built primarily on data from only the state of Michigan (Davidson et al. 2014), which are a subset of the incomplete data that comprise this study. Additional field data are essential, but so is the incorporation of those data into existing process-based models. In the end, the models face the same limitations as this meta-analysis, in that key data from certain soil types and climates are missing and there are also no calibration data available for some of the 4Rs that have been poorly studied.

We hoped to apply the refined understanding of NUE and N losses as affected by management across soils and climates to estimates of 4R management benefits with widespread adoption. However, the lack of geographic, climatic, and soil type coverage for many practices, and the lack of overlap between N_2O and NO_3 make this difficult. While we are able to answer a few questions about management, the conclusions are most often limited to the study locations and interactions between multiple factors difficult to tease out.

Conclusions

Even though data are limited, the field research on N_2O and NO_3 losses in corn-based systems in North America points to potential system improvements and reduced losses with a selection of 4R nutrient management practices. Fertilizer rate reductions without yield decline, combined with appropriate source, timing, and placement can provide air and water quality benefits. The total benefit possible depends on baseline practices and current loss rates in each region and for each cropping system. If the standard practices tested in the field data from this meta-analysis appropriately represent those in production agriculture, nitrification inhibitors could reduce average N_2O emissions by 36%, and side-dressing fertilizer instead of applying at or before planting could reduce emissions by 50%. In addition, where aqueous ammonia fertilizer is in use, average NO_3 leaching reductions of 18 kg N/ha may be achieved by using UAN or urea fertilizer instead. These reductions would likely vary with climate and soil characteristics, so any policies or programs incentivizing practices and estimating environmental benefit should use appropriately validated models.

However, these conclusions cannot be applied with certainty across the broad climatic and soil conditions in North America, especially those for which no field data are available. Cross-site comparisons are needed to capture both N_2O and NO_3 loss potential and how they relate to each other in varying regions and with different management. Such initial work could then be used to more efficiently target more strategically-coordinated field research. Current networks of research sites compiled for GRACEnet and LTAR studies may be good starting points for some of this work.

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Appendix A – Selection of Harvest Index for Estimation Purposes

1. Michigan State University Extension plots (2009-2012)[6] in various locations throughout the state, average 0.56, range from 0.35 to 0.79. Using 95% confidence interval for the mean, this is 0.56 (+-0.06).
2. Iowa State University[7], graph of numerous studies over time (era studies from 1920 to 1990), shows gradual increase in HI, studies between 1970 and 1990 have mean of 0.517 (+-0.015).
3. Lincoln, NE Ecological Intensification Trial, 1999 to 2001,[8] 0.50 observed, simulations ranged from 0.50 to 0.53
4. Default value used in RUSLE2 model is 0.50 for corn and 0.42 for soybeans.[9] Based on observations from T. Barten, unpublished, Edgerton et al. 2010 used a regression model for yields between 175-225 bushels per acre (HI=0.0007x + 0.4168), where x = yield in bushels/acre. This resulted in HI ranging from 0.54 to 0.57. A lookup table for field use assumed HI=0.55.
5. University of Minnesota Extension publication assumes HI=0.53.[10]
6. In Table 1, Johnson et al. 2006[11] cite Prince et al. 2001[12] for harvest index of corn (as of 2000). Prince et al., in turn, cited Kiniry et al. 1997[13], who determined the HI for corn of 0.53 from experiments, then compared their economic yield model with measured yields from Illinois, Iowa, Kansas, Louisiana, Minnesota, Missouri, Nebraska, New York, and Texas. The model performance was "quite good", according to Prince et al. This corn HI of 0.53 was used for the Prince et al. study because of the broad geographic application. Now Johnson et al. use it, and so do others.

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Examples from the literature (the first 9 observations are as reported by Kiniry et al. 1997, mean of 0.52):

Location	HI (mean)	HI (SD)	Reference(s)
New York, USA	0.46	0.04	Francis et al., 1978
Colorado	0.47	0.05	Fairbourn et al., 1970
Ontario, Canada	0.47	0.04	Daynard and Muldoon, 1981 (highest 9 trials)
Hungary	0.50	--	Zoltán, 1988, 1990
Florida	0.50	0.03	Bennett et al., 1989 (highest 4 treatments)
Minnesota	0.56	0.04	Voorhees et al., 1989
Argentina	0.57	--	Sobriano and Ginzo, 1975
Georgia	0.58	0.05	Brown et al., 1970
Nebraska	0.58	0.02	Raun et al., 1989
USA – maybe Texas	0.53	--	Kiniry et al. 1997

Based on the above information, we assumed a HI of 0.53 for estimations of N Excess.

Appendix B – Tabulated side-by-side comparisons of fertilizer source, placement, and timing as affecting N_2O and NO_3

Table A1. Fertilizer source studies of N_2O emissions. Number of side-by-side comparisons, with number of locations in parentheses.

	Polymer coated urea	SUPERU™	UAN	UAN+ AGROTAIN® PLUS	UAN+NITAMIN NFUSION®	Anhydrous ammonia	Anhydrous ammonia + Nitrapyrin	Ammonium nitrate	Urea: SUPERU™, 50:50	PiNT	Other
Urea	46 (6)	25 (4)	4 (1)	2 (1)	.	11 (3)	.	.	1 (1)	.	.
Polymer coated urea	.	28 (5)	6 (1)	4 (1)	.	.	.	.	1 (1)	.	.
SUPERU™	.	.	10 (2)	6 (1)	2 (1)	.	.	2 (1)	1 (1)	.	2 (1)
UAN	.	.	.	9 (2)	2 (1)	.	.	2 (1)	.	3 (1)	2 (1)
UAN+AGROTAIN® PLUS	.	.	.	.	2 (1)	.	.	.	.	3 (1)	.
Anhydrous ammonia	.	.	.	.	.	.	2(1)	.	.	.	.
Ammonium nitrate	.	.	.	.	.	.	.	.	.	.	2 (1)

Note 1. Observations for this table reported by: Venterea et al. (2010), Thornton et al. (1996), Fujinuma et al. (2011), Halvorson and Del Grosso (2012), Venterea et al. (2011b), Halvorson et al. (2010a), Parkin and Hatfield (2010), Halvorson and Del Grosso (2013), Halvorson et al. (2010b), Halvorson et al. (2011), Sistani et al. (2011), Drury et al. (2012), Nash et al. (2012), Maharjan and Venterea (2013), Dell et al. (2014), and Maharjan et al. (2014).

Table A2. Fertilizer source studies of NO_3 leaching losses. Number of side-by-side comparisons, with number of locations in parentheses.

	Urea + Nitrapyrin	Anhydrous ammonia + Nitrapyrin	ESN®
Urea	12 (1)	.	.
Anhydrous Ammonia	.	18 (1)	.
SUPERU™	.	.	4 (1)

Note 1. Observations for this table reported by: Maharjan et al. (2014), Randall and Vetsch (2003), Randall and Vetsch (2005), and Walters and Malzer (1990).

Table A3. Fertilizer placement studies of N_2O emissions. Number of side-by-side comparisons, with number of locations in parentheses.

	Band - sub-surface	Broadcast	Broadcast & Incorporate	Knife Inject - Deep
Band (general)	.	.	6 (1)	.
Band - surface	4 (2)	10 (1)	.	.
Knife Inject - Shallow	.	.	.	11 (2)

Note 1. Observations for this table reported by: Drury et al. (2006), Fujinuma et al. (2011), Halvorson et al. (2011), Halvorson and Del Grosso (2012), Halvorson and Del Grosso (2013), and Maharjan and Venterea (2013).

Note 2. While both Drury et al. (2006) and Fujinuma et al. (2011) tested shallow versus deep knife injection, the definition of shallow and deep were different for these two studies: 1) 2 cm versus 10 cm placement of ammonium nitrate, and 2) 10 cm versus 20 cm placement of anhydrous ammonia, respectively. Therefore, these are not similar enough to calculate meta-analysis effect sizes for the treatment.

Table A4. Fertilizer placement studies of NO_3 leaching losses. Number of side-by-side comparisons, with number of locations in parentheses.

	Broadcast	Broadcast & Incorporate	Localized Compaction & Doming
Broadcast	.	12 (1)	.
Knife Inject	.	.	5 (1)

Note 1. Observations for this table reported by: Bakhsh et al. (2010) and Walters and Malzer (1990).

Table A5. Fertilizer timing studies of N_2O emissions. Number of side-by-side comparisons, with number of locations in parentheses.

	Pre-plant only	Side-dress only	Split (pre-plant and side-dress)
Pre-plant only	.	1 (1)	3 (1)

Note 1. Observations for this table reported by: Smith et al. (2011) and Phillips et al. (2009).

Table A6. Fertilizer timing studies of NO_3 leaching losses. Number of side-by-side comparisons, with number of locations in parentheses.

	Spring only	Split (V2 +V18)	Split (V2+ V12)	Split (V2+V6)
Fall only	20 (2)	.		.
Side-dress only	.	2 (1)	1 (1)	1 (1)
Split (V2+V12)	.	.	.	2 (1)

Note 1. Observations for this table reported by: Jaynes (2013), Lawlor et al. (2011), Jaynes and Colvin (2006), and Randall and Vetsch (2005).