Nutr Cycl Agroecosyst (2017) 107:335–355
DOI 10.1007/s10705-017-9836-z

ORIGINAL ARTICLE

N_2O emissions from grain cropping systems: a meta-analysis of the impacts of fertilizer-based and ecologically-based nutrient management strategies

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Received: 18 May 2016 / Accepted: 18 February 2017 / Published online: 1 April 2017

Abstract Understanding how agricultural management practices impact nitrous oxide (N_2O) emissions is prerequisite for developing mitigation protocols. We conducted a meta-analysis on 597 pairwise comparisons (129 papers) to assess how management affects N_2O emissions. Pairwise comparisons of practices aimed at improving fertilizer use efficiency (39%) and tillage (30%) dominated the dataset, while ecologically-based nutrient management (ENM) practices constituted 15% of the pairs. In general, across management practices, the quantity of N added was a more significant driver of N_2O fluxes than was the form of N (fertilizer, legume biomass or animal manures). Manure interacted with soil texture so that in coarse soils, N_2O emissions from manures tended to be higher compared to inorganic N fertilizers. The studies of ENM strategies frequently involved overapplication of N inputs in the ENM treatments. Cover crops reduced N_2O emissions compared to bare fallows. However, during the cash crop growing season, when differences in N added and N source were confounded, the extra N inputs from cover crops were significantly correlated with the differences in N_2O emissions between treatments with and without cover crops. Overall, in 38% of the data pairs, N_2O emissions were reduced with limited impacts on yields; in half of these pairs, yields were maintained or increased while in the other half they were reduced by only $\leq$10%. Knowledge gaps on mitigation of agricultural N_2O emissions could be addressed by applying an ecosystem-based, cross-scale perspective in conjunction with the N saturation conceptual framework to guide research priorities and experimental designs.

Keywords Agroecosystem · Cover crop · Ecologically-based nutrient management · Fertilizer use efficiency · Meta-analysis · Nitrous oxide

Electronic supplementary material The online version of this article (doi:10.1007/s10705-017-9836-z) contains supplementary material, which is available to authorized users.

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Introduction

The release of reactive nitrogen (N) into the biosphere by humans has increased by 120% since 1970 (Galloway et al. 2008). Industrial agriculture accounts for the greatest proportion of anthropogenic N forcing globally, largely due to the use of inorganic N fertilizers as the primary source of N (Vitousek et al. 1997; Galloway et al. 2008). Reactive N is lost through two main pathways; NO_3^- leaching and release of gaseous N forms, particularly N_2 and N_2O (Galloway et al. 2003). Nitrous oxide (N_2O) is a potent greenhouse gas that is 298 times more potent than CO_2 on a 100-year time scale, and it is a strong ozone-depletion substance (Forster et al. 2007). Agriculture accounts for ~60% of anthropogenic N_2O emissions, and agricultural soils are the dominant source (IPCC 2006). As a result, there is an urgent need to improve management of agricultural N to reduce N_2O emissions from agricultural soils.

Nitrogen saturation theory and agroecosystems

Optimizing N management to achieve yield goals while minimizing environmental losses has proven to be a considerable challenge. The loss of reactive N from agricultural systems is viewed primarily as a consequence of temporal asynchrony and spatial separation between applied nutrients and the crops (Stevenson and Baldwin 1969; Welch et al. 1971; Cassman et al. 2002). Efforts to mitigate N losses have focused on increasing the proportion of N fertilizer taken up by crops using "the 4Rs", a fertilizer management strategy that aims to increase crop uptake by optimizing application rate, chemical composition, timing and placement of fertilizers (c.f. Chen et al. 2011; Venterea et al. 2016). Using this strategy there has been some improvement in fertilizer use efficiency (FUE); that is, yields have generally increased per kg N^{-1} fertilizer added due to a variety of mechanisms (Fageria and Baligar 2005; David et al. 2010). Still, on average about half of the fertilizer N applied is lost to the environment (Galloway et al. 2003).

Recently, a broader approach to N management known as "ecologically based nutrient management" (ENM) has been proposed (Drinkwater and Snapp 2007). ENM is based on concepts developed to explain changes in forest N biogeochemistry resulting from chronic anthropogenic N deposition (Agren and Bosatta 1988; Aber et al. 1989). Ecosystems are considered to be N saturated when primary productivity is no longer limited by N, and N additions exceed the capacity of the ecosystem to cycle or store N internally (Agren and Bosatta 1988; Aber et al. 1989; Gundersen et al. 2006). The consequences of excess N in ecosystems include increased N mineralization and nitrification, NO_3^- leaching, and greenhouse gas fluxes as well as soil acidification and base cation depletion (Fenn et al. 1998). The N saturation conceptual model highlights the significance of C–N coupling in driving N retention/loss and has led to major advances in C and N, and more recently, P biogeochemistry (Fenn et al. 1998; Luo et al. 2004; Magnani et al. 2007; De Schrijver et al. 2008; Mulholland et al. 2008; Schlesinger 2009; Fenn et al. 2010; Vitousek et al. 2010; Crowley et al. 2012). Application of this conceptual framework could have similar ramifications for N management in agricultural systems (Drinkwater and Snapp 2007; Thorburn et al. 2011; Fisk et al. 2015; Attard et al. 2016; Tosti et al. 2016).

In addition to optimizing crop uptake of N fertilizer, ENM seeks to reduce N losses using practices such as reduced fallow periods, diversified crop rotations, increased reliance on biological N fixation and additions of C containing N inputs such as manure and legumes which favor C-N coupling and accrual of soil organic matter reserves (Drinkwater et al. 1998, 2008; Kallenbach and Grandy 2011; Bowles et al. 2015; McDaniel et al. 2014). Recoupling C and N can foster microbially-mediated processes that enhance internal N cycling pathways that favor N retention/N accrual (Fisk et al. 2015). Previous meta-analyses have shown that using ENM practices increased N retention and reduced NO_3^- leaching compared to practices that only target improved crop assimilation of added fertilizer (Tonitto et al. 2006; Gardner and Drinkwater 2009; Quemada et al. 2013) suggesting that ENM can be used in conjunction with the 4Rs to optimize N fertilizer management and reduce environmental N losses.

Agricultural practices and their impact on N_2O emissions

A number of meta-analyses have synthesized the growing N_2O literature, and these reviews have contributed substantially to a better understanding of the impact of management practices on N_2O emissions (Table 1). Most of these meta-analyses have characterized relationships between fertilizer N management and N_2O emissions with some assessments also examining yield-scale emissions. Others have limited the scope of their analysis to a particular region or

Table 1 Focus area and the dataset size of previous quantitative synthesis (2010–2016)

References	Study focus	# Studies	# Pair-wise observations if meta-analysis
Van Kessel et al. (2013)	Tillage	41	239
Akiyama et al. (2010)	Slow release fertilizer and urease/nitrification inhibitors	35	113
Chen et al. (2013)	Residue incorporation	30	219
Van Groenigen et al. (2010)	N rates/surplus	19	147
Kim et al. (2013)	Fertilizer rates	11	–
Shcherbak et al. (2014)	Fertilizer rates	78	–
Linquist et al. (2012)	Major cereal crops	57	328
Aguilera et al. (2013)	Fertilizer and water management, Mediterranean region	24	–
Basche et al. (2014)	Cover crops	26	106
Decock (2014)	Fertilizer management, rotation, tillage, the Midwestern US	48	–
Abalos et al. (2016)	Fertilizer management, North America	23	200

constrained their analysis to focus on the impact of a few specific practices on N_2O emissions. However, a comprehensive assessment of the full range of agricultural practices and their impact on N_2O emissions has not been conducted. To assess the full range of options currently available for N_2O mitigation, we conducted an extensive analysis of all management practices for which there were at least eight published papers meeting our criteria, including both 4Rs and ENM practices. We also quantified the tradeoff between N_2O mitigation and yield outcomes in grain cropping systems. We were particularly interested in determining whether or not the degree of N saturation resulting from distinct management practices was related to N_2O emissions. Based on our analyses, we considered how to improve empirical research on N_2O emissions from agroecosystems, identified knowledge gaps and recommended future research priorities.

Methodology

Building the dataset

An exhaustive literature search of studies investigating N_2O emissions from grain cropping systems was conducted with *ISI-Web of Science* for articles published before June 2014. Because the first search produced a limited numbers of papers for enhanced efficiency fertilizers, cover crops and diversified rotations, we conducted a second search focusing on these practices in December 2015 to increase the size of the database and enable meta-analysis of these practices. Studies on fertilizer rates were compiled by screening through a global database by Stehfest and Bouwman (2006) and further including recent studies after 2006. Only studies conducted in field conditions that were at least one growing season in duration were included. Our final database consisted of 129 studies and 597 pairwise comparisons (meta-analysis references and database are available as online Supplemental Information; S1, S2). We included cover crop studies that measured N_2O emissions from cover crop growth periods, cash crop growth periods or both, and examined these sub-groups separately. For studies comparing diversified rotations to simplified rotations, cumulative N_2O emissions from the entire rotation were used as one observation. For other studies testing treatments other than rotation effects (e.g. tillage, fertilizer practices), we treated each grain crop in the rotation as an observation. When experiments were repeated for multiple growing seasons/years, average cumulative emissions per growing season/year (in kg N_2O-N ha^{-1} $year^{-1}$) were used to avoid bias towards multi-year measurements. Soil texture, pH, climate, yield data were extracted from each study when available. Studies reported crop yields at varying moisture contents, so water content of grain yields was adjusted to the percent moisture commonly used for each grain: 14.5% moisture content for maize (*Zea mays*), 16.5% wheat for (*Triticum* spp), 18% for barley and 10% for canola. For studies that reported yield information, we calculated yield-scaled N_2O emissions (YSE), which was N_2O emissions divided by the crop yields.

Categorizing the management strategies

Management strategies were categorized into three broad groups and the control–treatment pairs within each group are described in Table 2. The first group consisted of practices that aim to increase FUE by increasing crop assimilation of fertilizer N based on the 4Rs concept. This category includes practices that manipulate fertilizer application rate, placement and timing, or use enhanced efficiency fertilizers. Beyond the focus on crop uptake, other processes governing N cycling and agroecosystem-scale N saturation are not addressed by these practices with the exception of chemical modifications of fertilizers such as inclusion of nitrification inhibitors. Our second grouping, ecologically-based nutrient management (ENM) aims to reduce N saturation in space and time by reducing N additions in conjunction with greater reliance on internal soil N cycling processes. Diversifying crop rotations and reducing bare fallows, particularly by adding cover crops or perennials, as well as expanding reliance on legume N sources are examples of practices that are compatible with this strategy. Lastly, several practices that impact N_2O emissions and other environmental N losses do not fall into either FUE or ENM. For example, replacing inorganic N fertilizer with manure can either recouple C and N cycling or exacerbate N saturation depending on the timing and rate of manure application (Edmeades 2003; Blesh and Drinkwater 2013). Likewise, reduced tillage can be used in conjunction with either FUE or ENM practices but does not fit into either management strategy in its own right. Studies evaluating the impact of these practices on N_2O emissions have been placed in the "Others" category (Table 2).

We found 203 pairs of observations assessing the impacts of different fertilizer rates. To better reflect the agronomic context, a subset of the studies on N fertilizer rates was created by identifying the recommended fertilizer rates used in each study and excluding pairs with no fertilizer application. N_2O emissions from fertilizer at recommended rates were defined as controls, with other application rates as treatments. The subset was categorized by higher or lower than recommended rates to examine the relationships between fertilizer rates and N_2O emissions in further detail. Recommended rates were determined using extension resources for specific crops by region or state unless otherwise specified by the authors.

Data analysis

The effect size for each control–treatment pair was estimated using the response ratio ($R = Xt/Xc$), where Xt is the mean N_2O emission from the treatment, and Xc is the mean N_2O emission from the control. To perform meta-analysis, the natural

Table 2 Descriptions of control–treatment pairs by category, number of studies and pairwise comparisons for each category

Category	Control	Treatment	# Studies	# Control–treatment pairs
Fertilizer use efficiency (FUE)	Recommended fertilizer rate	Higher fertilizer rate	21	61
	Recommended fertilizer rate	Reduced fertilizer rate	15	49
	Shallow fertilizer placement	Deep fertilizer placement	8	28
	Fall fertilizer application	Delayed fertilizer application[a]	8	34
	Urea	Polymer-coated Urea	23	60
Ecologically-based nutrient management (ENM)	Bare fallow	Cover crops	21	61
	Simplified rotation	Diversified rotation	12	31
Others	Inorganic fertilizer	Manure	23	91
	Conventional tillage	Reduced tillage	46	181

[a] Delayed fertilizer application: fertilizer application in spring or split application during growing season

logarithm of Response Ratio (LRR) was used to normalize data distribution (Johnson and Curtis 2001). We performed an unweighted meta-analysis because roughly more than half of the studies did not report a measure of variance. Bias-corrected 95% Confidence Intervals (95% CIs) were generated through a bootstrapping procedure in MetaWin 2.0 (5000 iterations). Bootstrapping CIs can be biased due to re-sampling from a small dataset (Bancroft et al. 2007; Montero-Castaño and Vila 2012) so for groups with <10 observation pairs, we used the more conservative 95% CIs instead of bias-corrected 95% CI. The 95% CIs were converted to the percent change of N_2O emissions from treatment groups compared to the controls for easier interpretation. A CI not overlapping with zero suggests a significant treatment effect and non-overlapping CIs indicate significant difference between groups.

Using the method described above, we conducted our meta-analysis on area-scaled and yield-scaled N_2O emissions. We further explored the effect sizes with categorical variables such as soil texture, fertilizer rates and manure forms and continuous variables such as manure pH and manure C:N ratios to examine how specific management regimes impact N_2O emissions.

To assess the tradeoff between yield and N_2O mitigation, Yield Ratio was calculated as the ratio of yield in a treatment to yield in the control. N_2O Ratio was calculated as the N_2O from a treatment over the control. For studies comparing diversified to simplified rotations, Yield Ratio was calculated based on yields of the same crops in the different rotation using yields from the least diverse rotation as the control.

Results

Overview of dataset

Control–treatment pairs targeting FUE practices (39%) and reduced tillage (30%) dominated the dataset, followed by studies comparing manure to inorganic N fertilizers (15%), while studies on two ENM strategies accounted for 15% of the pairs (Table 2). Although the data set represented a global coverage, there was geographic imbalance within the dataset, with most of the measurements in developed countries in North America and Europe. The geographic coverage of the data was as follows: USA (34%) and Canada (23%), Asia (19%), Europe (17%), South America (4.5%), and other regions (6.7%) with the vast majority (81%) being located in temperate climatic zones. Only 28% of the 129 studies were conducted in irrigated areas and the top five cash crops studied were maize (56%), wheat (18%), barley (14%), soybean (8.2%), and canola (4.2%).

Overall, adjusting fertilizer rates had the most significant impact on N_2O emissions among all management practices (Fig. 1). For all other management practices we did not detect significant differences in N_2O emissions compared to the controls.

Strategies aimed at FUE

Meta-analysis showed that on average, applying fertilizer at higher than the recommended rates increased N_2O emissions by 55% (bootstrapping 95% CI 38–73%), while applying fertilizer at lower than recommended rates decreased N_2O emissions by 33% (bootstrapping 95% CI −39 to −26%). Average N_2O emissions from the control group (recommended rates) were 2.6 kg N_2O-N ha^{-1} $year^{-1}$ (range 0.1–19.5 kg N_2O-N ha^{-1} $year^{-1}$) while average emissions from treatments higher or lower than recommended fertilizer rates were 3.1 kg N_2O-N ha^{-1} $year^{-1}$ (range 0.09–20.0 kg N_2O-N ha^{-1} $year^{-1}$) and 2.3 kg N_2O-N ha^{-1} $year^{-1}$ (range 0.17–19.6 kg N_2O-N ha^{-1} $year^{-1}$), respectively.

Manure compared to Inorganic fertilizer

When all studies comparing manures to N fertilizer were included in the analysis, regardless of differences in N additions, the percent change for N_2O emissions using manure instead of fertilizers averaged 12% with a range of −9 to 38% (bootstrapping 95% CI). The average N_2O emission from the manure treatment group was 3.8 kg N_2O-N ha^{-1} $year^{-1}$ compared to 3.4 kg N_2O-N ha^{-1} $year^{-1}$ for the inorganic fertilizer treatment group. However, in 8 out of 23 studies, N additions from manure were greater compared to those from N fertilizer. Removal of these pairs resulted in average N_2O emissions of 4.8 kg N_2O-N ha^{-1} $year^{-1}$ from inorganic fertilizers and 4.2 kg N_2O-N ha^{-1} $year^{-1}$ from manures. The average percent change for N_2O emissions using manure instead of fertilizers shifted from 11 to −3% (bootstrapping 95% CI −13 to

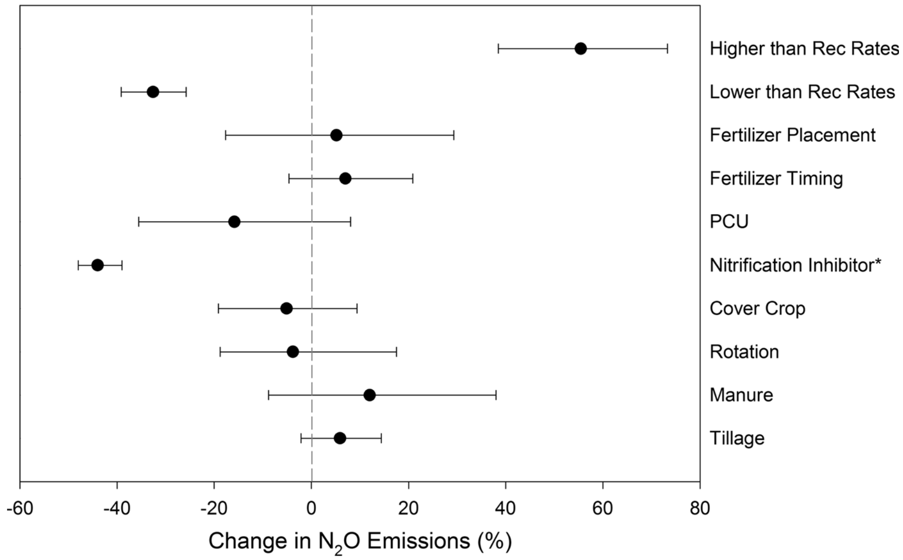

Fig. 1 Effect of management practices on area-scaled N_2O emissions reported as percent change from the control. Mean values and 95% confidence intervals of the back-transformed response ratios are shown. The result for nitrification inhibitors was from Qiao et al. (2015) and was shown for comparison

8%) indicating that the level N inputs was a significant driver for the manure effect on N_2O in the full dataset. This hypothesis was confirmed by a positive correlation between the LRR for N_2O emissions and the difference in total N inputs ($y = 0.0081x - 0.033$, $R^2 = 0.23$, $p < 0.0001$, Fig. 2a). Furthermore, the slightly negative intercept of the regression line resulting from the wide spread of data points around the origin suggested that other factors, such as manure properties and environmental factors, contribute to the negative drift of the intercept. For example, total ammoniacal nitrogen contents, the sum of NH_3 and ammonium (NH_4^+) in manure, can differ significantly depending on manure sources and soil textures (Sommer and Hutchings 2001). We therefore further examined this variability. Using this subset of pairs where total N inputs from manure and inorganic fertilizer were the same, neither manure type (slurry or solid) nor manure pH influenced N_2O emissions. However, we found a significantly negative correlation between the LRR for N_2O emissions and soil clay content (Fig. 2b). In coarser soils, manure slightly increased N_2O emissions compared to inorganic fertilizer, while in finer soils, manure tended to have similar or lower N_2O emission compared to fertilizer. We also found a negative relationship between LRR and manure C:N ratios ($y = 3.6 - 0.13x$, $R^2 = 0.2$, $p = 0.04$, data not shown). It is important to note that 11 out of the 23 studies in the *Manure* group did not report manure C:N ratios, and 14 studies did not report manure pH values, and six studies did not provide soil texture information.

Ecological nutrient management strategies

The 21 studies in the *Cover Crop* category were highly variable in terms of the focus of N_2O measurements. Six studies (19 observations) measured N_2O during only the cover crop growth period. An additional nine studies (37 observations) measured N_2O emissions during only grain crop growth, after cover crops had been killed or incorporated. The remaining six studies measured N_2O emissions during both cover crop and grain crop growth, but only three of these studies reported the two crop phases separately. Using the data available for each crop phase, we found that N_2O emissions from cover crop growth periods were 58% lower (bootstrapping 95% CI −81 to −27%) compared to bare fallows (Fig. 3). When we excluded the six observations from legume cover crops we found a slightly greater average reduction of 66% with a similar level of variation (bootstrapping 95% CI ranging from −87 to −29%) suggesting that living cover was the primary driver, regardless of the N acquisition strategy. However, the limited data from legume cover crop growing periods may have hindered our ability to distinguish the impact of legumes and non-legumes during the cover crop phase.

All studies reporting N_2O emissions during the cash crop growing season compared "controls" receiving N

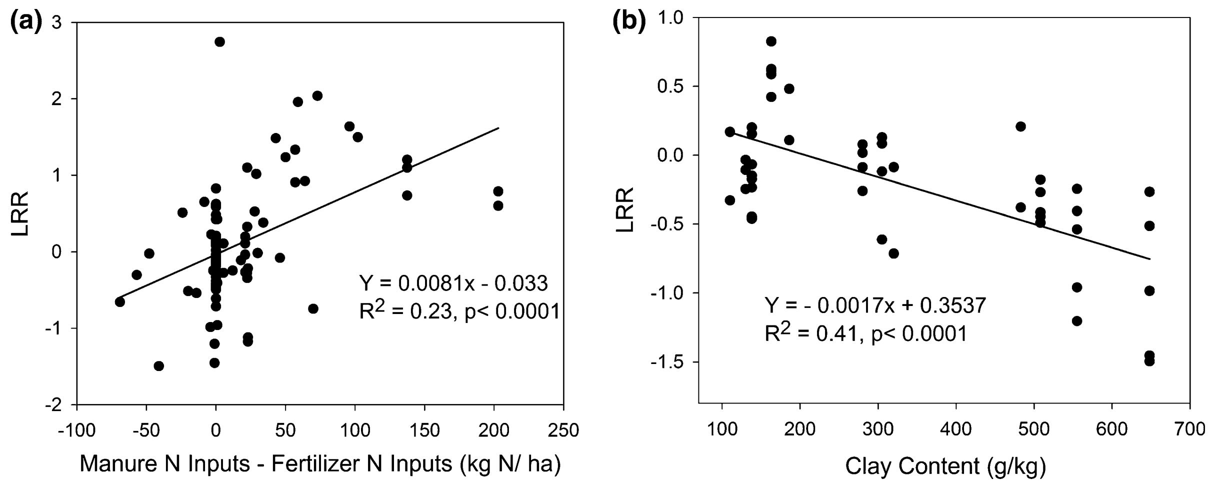

Fig. 2 **a** Relationships between the natural logarithm of the response ratio (LRR) for N_2O emissions from manure (treatment) versus fertilizer (control) and **a** difference of total N inputs from manure and fertilizer (n = 92). **b** Relationships between the natural logarithm of the response ratio (LRR) for N_2O emissions from manure (treatment) versus fertilizer (control) and the soil clay content (g kg^{-1}). The total N inputs from manure and fertilizer were the same (n = 47)

fertilizer to treatments receiving N fertilizer plus cover crop biomass. As a result, the cover crop treatments received an average of 89 kg N ha^{-1} more N compared to the controls with total C and N additions varying greatly, depending on cover crop biomass and species. On the low end, grass cover crops producing less than 1.5 Mg dry weight of biomass added only 9–31 kg N ha^{-1} while highly productive legume cover crops added 156–279 kg ha^{-1} of additional N. Only two studies reduced fertilizer N rates in the cover cropped treatments to reflect N added from legumes (Robertson et al. 2000; Alluvione et al. 2010). The C:N ratios of the cover crop biomass also varied greatly, ranging from 10 to 26 for legumes and grass cover crops, respectively.

Based on the 12 studies reporting N_2O emissions in cover cropping systems during the cash crop growth periods in conjunction with cover crop N content, we found a significant positive correlation between the LRR for N_2O emissions and the extra N inputs from both legume and non-legume cover crops ($y = 0.0023x$, $R^2 = 0.08$, $p = 0.003$, Fig. 4a). Removing an extreme value where extra N inputs = 242 kg N ha^{-1} and LRR = 1.7 increased the proportion of the variation in N_2O emissions was attributed to the extra N inputs ($y = 0.0013x$, $R^2 = 0.15$, $p = 0.04$). N_2O emissions tended to be negatively correlated with cover crop C:N ratios but the relationship was not significant ($y = -0.099x + 2.9$, $R^2 = 0.09$, $p = 0.15$, Fig. 4b).

We could not analyze the full cover cropping dataset for the independent effects of cover crop type (legume versus non-legume) or tillage (incorporation vs no-till) because only two observations represented

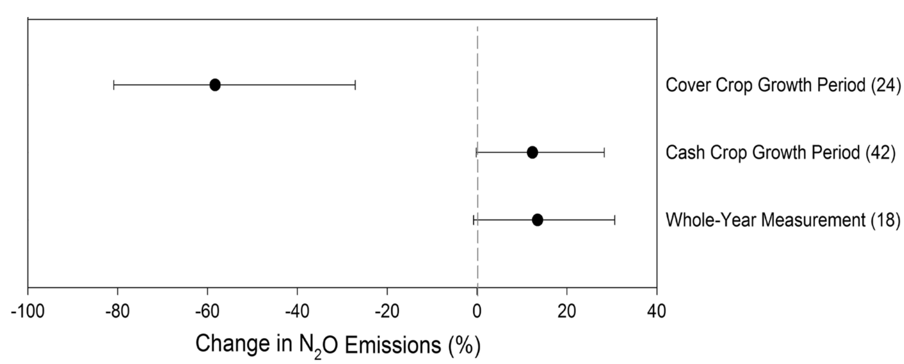

Fig. 3 Effect of cover crops on area-scaled N_2O emissions depending on different measurement periods. Mean values and 95% confidence intervals of the back-transformed response ratios are shown

instances where legume cover crops were not incorporated. Thus, two conditions, which we expected to increase N_2O emissions (N-rich biomass and incorporation of shoots), were confounded. We analyzed sub-categories with sufficient data points and did not detect significant effects of either cover crop type or tillage on N_2O emissions (Fig. 4c). The small number of observations combined with the variable additions of fertilizer N to cover crop treatments probably contributes to the large variation in N_2O emissions observed within these subsets.

We found no detectable effect of diversifying rotations on N_2O emissions, possibly because the "diverse" rotations usually consisted of only two alternating crops. For example, in the *Diversified Rotations* group, eight out of the twelve studies compared monocultures such as continuous maize or wheat to rotations that included one additional crop such as maize–soybean or maize–dry bean, or lupin–wheat. Therefore, most of the studies compared continuous monocultures to relatively simple rotations that did not reduce bare fallow periods. In the three studies that reduced bare fallow periods in conjunction with reduced fertilizer N inputs diversified rotations had reduced or similar N_2O emissions in (Jacinthe and Dick 1997; Jantalia et al. 2008; Benoit et al. 2015).

Impacts on N_2O emissions and associated yield consequences

Seventy-three out of the 129 studies reported yields and enabled us to compare yield-scale emissions (YSE) for 341 data pairs (57% of the full dataset). The overall pattern of YSE was similar to our results for

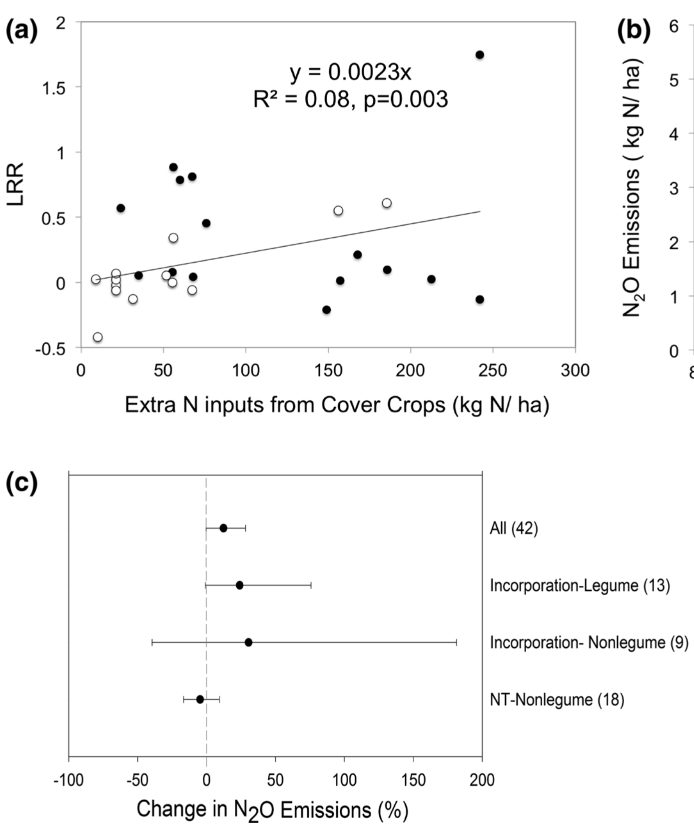

Fig. 4 **a** Relationship between the natural logarithm of the response ratio (LRR) for N_2O emissions from cover crops (treatment) versus bare fallows (control) and the extra N inputs from cover crops ($n = 29$). *Solid dots* incorporation, *open dots* no till. **b** Relationship between N_2O emissions and cover crop C:N ratios ($n = 27$). **c** Effect of cover crop types and tillage on N_2O emissions reported as percent change from the control (bare fallows). Mean values and 95% confidence intervals of the back-transformed response ratios are shown. N_2O emissions were based on measurements during cash crop growing periods in all three figures

area-scaled emissions; increasing fertilizer application rates above the recommended N rates significantly increased YSE while lower than recommended rates reduced YSE (0.45 compared to 0.25 kg N_2O-N Mg^{-1} grain, respectively; with min 0.01 and max 2.64 kg N_2O-N Mg^{-1} for both groups). Yield-scaled emissions were not significantly different among maize and winter grains (crop averages ranged from 0.24 to 0.30 kg N_2O-N Mg^{-1} for maize, barley and wheat) while soybean and canola had comparatively greater YSE (0.90 and 0.99 kg N_2O-N Mg^{-1}; data reported in Table S1 and Table S2).

Plotting N_2O Ratio (N_2O emission from treatment over control) against Yield Ratio (the yield of treatment over the yield of control) for each control–treatment pair showed that most treatments tended to have a greater impact on N_2O emissions compared to yields (Fig. 5). Only fifteen observations from four studies had Yield Ratios larger than 1.5. In contrast, sixty observations from 29 studies had N_2O Ratio larger than 1.5. Among 341 data pairs, 43% had increased N_2O emissions with half of these data pairs falling into the "lose–lose" quadrant, which increased N_2O and decreased yields. Only 17% of data pairs achieved the "win–win" situation that mitigated N_2O with yield benefits while 38% of the pairs fall into the bottom-left quadrant, which decreased N_2O emissions but had some yield penalty. Nearly half (44%) of these data pairs in bottom left quadrant had a $\leq$10% yield reduction, suggesting that N_2O mitigation could be achieved with minor yield loss.

Across the three categories, the ENM strategies had nearly symmetrical impacts on N_2O emissions and yields, with the vast majority of data pairs falling between 0.6 and 1.4 for both axes. In contrast, strategies in *FUE* and *Others* tended to have much greater variation in N_2O emissions with instances where N_2O emissions were 2–3 fold greater while yields showed only small gains, and in some cases were reduced. There was a significant exponential relationship between N_2O Ratio and Yield Ratio for studies that altered fertilizer rates ($R^2 = 0.21$, $p < 0.001$), Fig. 5a).

Fourteen out of 15 data pairs in the *Fertilizer Placement* group had N_2O ratios higher than one, ranging from 1.2 to 2.4, of which half resulted in yield decrease. In the *Tillage* group, using reduced tillage compared to conventional tillage resulted in moderate decrease in yields for 71% of the pairs, among which nearly half had increased N_2O emissions in conjunction with reduced yields. In the *Manure* group, replacing fertilizer N with manure decreased N_2O emissions in 29 out of the 51 data pairs, however, 20 out of the 29 pairs also decreased yields with the majority showing greater than 10% yield reduction. Lastly, in the *Polymer-Coated Urea* group, 39 of the 58 pairs resulted in decreased N_2O with the corresponding Yield Ratios equally split between increased or decreased yields.

Discussion

Fertilizer management

We found that N fertilizer rate had the most significant impact on N_2O emissions. This is in line with previous study reporting that N_2O emissions were mainly controlled by fertilizer rates (Bouwman et al. 2002). The consistency of this relationship between the quantity of N applied and N_2O emissions is congruent with studies linking N rates to the size of soil inorganic N pools, nitrate leaching and total N losses (Boy-Roura et al. 2016; Rasmussen et al. 2015; Shaddox et al. 2016). Furthermore, this demonstrates the central role of reactive N in driving loss pathways as predicted by the N saturation hypothesis. We also found an exponential relationship between N_2O Ratio and Yield Ratio suggesting that N_2O emissions increased exponentially when fertilizer application exceeded plant uptake (McSwiney and Robertson 2005; Van Groenigen et al. 2010; Linquist et al. 2012). This is in line with the findings of an extensive global synthesis by Shcherbak et al. (2014) that N_2O emissions in response to increasing N inputs were exponential rather than linear.

Altering the timing and depth of fertilizer application or slowing down the release of inorganic N by using polymer-coated urea instead of urea did not consistently reduce average annual N_2O emissions. While the specific mechanisms differ among practices, the underlying premise for all of these practices is that improving the synchrony/proximity of N fertilizer with plant uptake/plant roots will enable crops to take up more fertilizer N with corresponding reductions in environmental N losses. Using polymer-coated urea is likely to have the greatest impact on N losses and N_2O emissions in the short term, immediately after

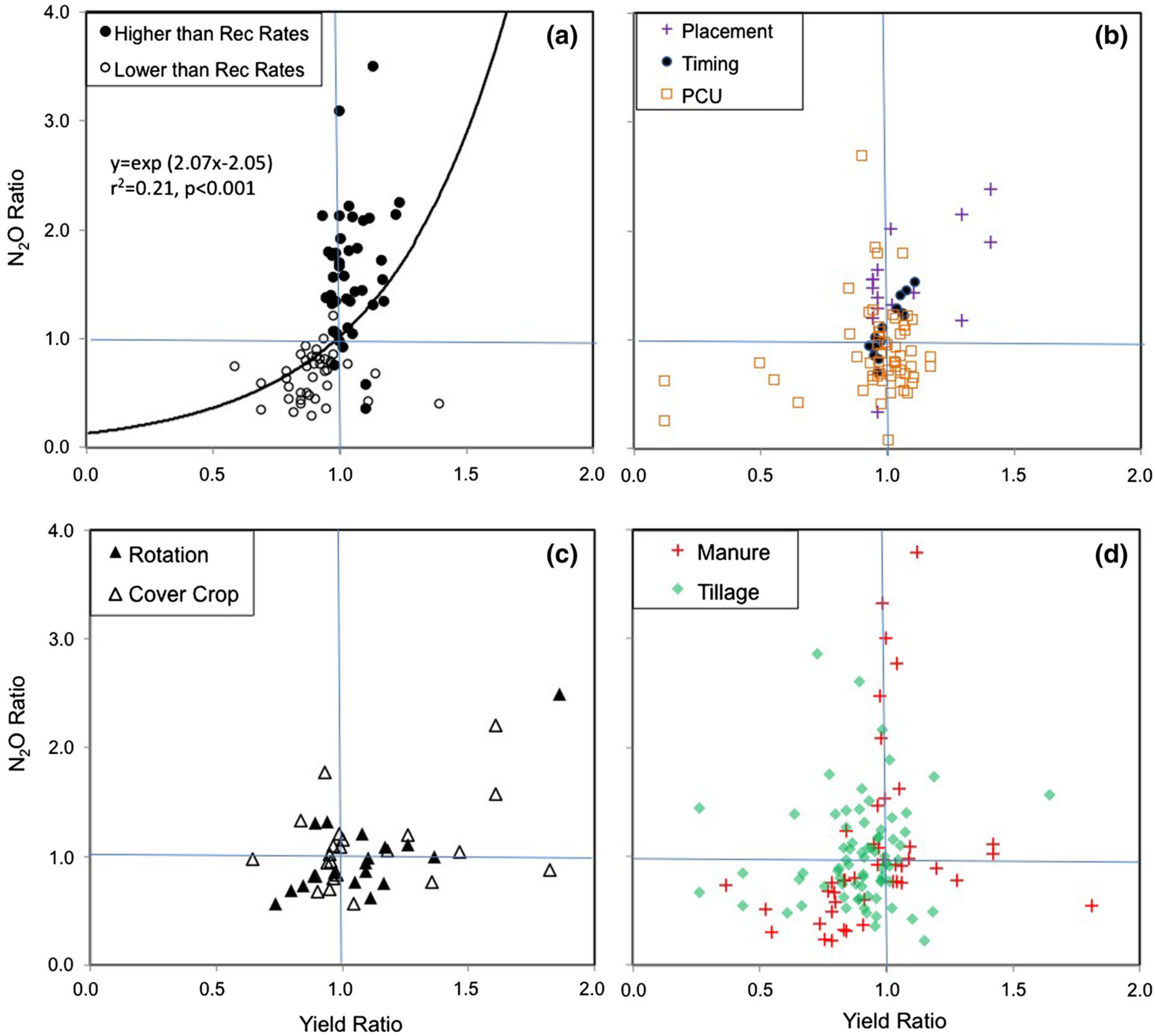

Fig. 5 The tradeoff between yield and N_2O mitigation. The Yield Ratio (X-axis) is calculated as the ratio of treatment yields to control yields and the N_2O ratio (Y-axis) is treatment N_2O emissions from over those from the controls. To increase the clarity of the graphs, some data points with extreme values were not shown. Seven data points in the *Tillage* group reporting a yield ratio larger than 2.0 and N_2O rations of 0.56–1.8 are not shown. All seven observations were from Plaza-Bonilla et al. (2014). Three data points in *Manure* group (from van Groenigen et al. 2004), one data point in *Tillage* group (from Abdalla et al. 2010) and one data point in *Polymer-Coated Urea* group (from Ji et al. 2012) reported N_2O Ratio higher than 5.0 and Yield Ratios ranging from 0.93 to 1.05 were not plotted

application, but its effectiveness over a longer time frame is subject to weather patterns (Hatfield and Venterea 2014). As a result, it appears that the timing of N losses is altered, but retention over the entire growing season or into the following year is not improved (Venterea et al. 2011). We were also unable to detect a consistent pattern in N_2O emissions resulting from placement techniques, probably due to the interaction of fertilizer placement with other factors such as fertilizer type, soil texture, tillage type, etc. (Eagle et al. 2012).

Meta-analysis of ^{15}N tracer studies indicates that some FUE practices do increase the proportion of N fertilizer taken up by the crops with corresponding reductions in the proportion of N lost, however the impact on N losses tends to be less than the increase in crop acquisition (Gardner and Drinkwater 2009). For example, improving temporal synchrony through various practices increased crop acquisition of N fertilizer by 13–42% but these same practices increased total N fertilizer recovery by only 2–21% (Gardner and Drinkwater 2009). The inconsistent conclusions between studies using ^{15}N tracers to study N losses as a whole versus those that measure only N_2O emissions highlight the fact that N_2O emissions, *per se* are not indicative of total N losses, since N can also be lost through leaching of NO_3^- and as N_2 during denitrification. Furthermore, the proportion of N released as N_2O rather than N_2 during denitrification can vary tremendously. For example, in some cases treatments with similar N_2O emissions have been shown to have very different rates of gaseous N losses due to differences in the partitioning of N released as N_2O versus N_2 (Cavigelli and Robertson 2000; Kramer et al. 2006).

Taken together these results suggest that practices targeting FUE in terms of crop uptake do not necessarily decrease N_2O emissions even if crop FUE is improved. However, one limitation of these FUE studies is that they all use a factorial design, and only a single aspect of N fertilizer management was modified. While this is a necessary starting point, other studies of FUE using 4Rs strategy find that integrating practices that improve synchrony of N availability and crop assimilation enables reductions in N rates while maintaining yields (Meng et al. 2016; Sela et al. 2016). In the N_2O studies available for this meta-analysis, when N fertilizer rate was tested, none of the other 4Rs were implemented (i.e. timing and placement are not adjusted to improve crop assimilation of fertilizer N). Furthermore, in studies of other 4Rs practices, such as deep placement or split applications, the rate of N was not reduced in the treatments where were expected to improve FUE. This points to a significant knowledge gap; the potential for the 4Rs strategy to contribute to even greater reductions in N rates, which in turn will certainly reduce N_2O emissions, has not been sufficiently investigated.

Manure management

Differences in N_2O emissions from fertilizer and manure in our dataset were positively correlated with differences in total N inputs between the two sources, suggesting that the effects from different N sources *per se* and total N rates should be examined separately. When we analyzed the pairs where N inputs were the same for manure and N fertilizer, the effect ratio shifted lower indicating that N_2O emissions from manure were similar to fertilizers or even lower. This suggests that using the same default emission factor (1%) for manure and fertilizer N in IPCC Tier 1 guideline for international/regional calculations is acceptable. However, the large variation around the fitted line in Fig. 2a highlights the importance of considering manure composition and local soil properties when developing country- or site-specific emission factors (IPCC Tier 2 & 3).

Many experiments comparing the effect of manure versus N fertilizer on N_2O emissions apply greater amounts of N to the manure treatments. This confounds the impact of N source versus the quantity applied and leads to the misperception that manure *per se* increases N_2O emissions (Perala et al. 2006; Sistani et al. 2011; Velthof and Mosquera 2011). In a regional study of the Midwestern US Cornbelt, Decock (2014) synthesized nine studies comparing emissions from manure with emissions from fertilizer. In this case, Decock found that manure led to higher N_2O emissions compared to fertilizer, concluding this could be due to "manure-induced changes in readily available C, soil structure and/or microbial communities" although 22 of the 73 observations had higher total N inputs from manure compared to fertilizer.

The application of greater amounts of total N in manure treatments used in some experiments probably reflects farmer practice because farmers often apply manure N at rates much higher than they would apply fertilizer N to achieve similar total plant-available N. Thus, these studies should be interpreted differently than those using a factorial design. They provide useful information about how typical farmer manure practices impact N_2O emissions compared to fertilizers but they don't provide the information we would need to make recommendations for better practices since we don't know if the greater N_2O is an inherent risk with manure use or if it is due to the larger N applications.

Our analysis underscores the importance of avoiding excessive applications of manure, and highlights the differing effect of soil texture on N_2O emissions following additions of inorganic fertilizer and manure. We found that as clay content decreased, using manures instead of inorganic fertilizers tended to increase N_2O emissions. This could be due to the lower SOC contents usually found in coarse-textured soils. The addition of C from manure application could relieve C limitation and N_2O emissions could therefore be stimulated (Chantigny et al. 2010). In the end, accurate predictions of N_2O emissions from manure will require consideration of manure characteristics as well as climatic conditions and soil properties. The heterogeneity of manures presents a particular challenge given the huge variability in C: N ratios, as well as the large differences in inorganic N, water and labile versus recalcitrant compounds, all of which will then interact with native soil organic matter and environmental conditions to drive N cycling. For example, Aguilera et al. (2013) reported that N_2O emission per N applied was higher for liquid manure than solid manure in Mediterranean soils where soil organic C and N contents were very low and N_2O production was promoted by higher NH_4^+ in slurry. The negative relationship we found between LRR and manure C:N ratios is congruent with our understanding of decomposition dynamics. Manures with larger C:N ratios tended to generate less N_2O compared to fertilizers due to increased immobilization and reductions in standing inorganic N pools. Thus, manure application rates could be adjusted to reflect C:N ratios so that manures with greater C contents are applied at higher rates.

Nitrous oxide emissions resulting from the land application of manure are not the only concern in manure management. Ammonium loss and other greenhouse gas emissions can occur during the processing and storage stages. Manure management plans should be based on impacts on total greenhouse gas emissions along the whole management chain because measures that reduce one GHG can influence emissions of other GHGs at different stages of manure management. In addition, the tradeoff between ammonium loss and N_2O emissions should be considered. For example, NH_3 mitigation measures for manure land application, i.e., slurry injection or direct incorporation of manure can result in increased emissions of N_2O and total GHG emissions across all stages of manure management (Hou et al. 2015). To optimize manure management and minimize environmental impacts will require full life cycle analyses of manure in order to avoid a piecemeal approach that may ultimately reduce GHG emissions at one stage only to increase emissions at other points in the process.

Ecologically-based nutrient management practices

Overall, only a limited number of field studies have measured the impact of cover cropping on N_2O emissions. To fully quantify the impact of cover cropping on N_2O emissions, it is important to collect measurements during both the fallow and the cash crop growing periods of the rotation cycle and to distinguish between these two phases. Studies that focused on the cash crop growing season dominated our dataset while studies comparing N_2O emissions from the growth periods of legume cover crops versus bare fallows, such as Li et al. (2015) were especially lacking. Growing non-legume cover crops during the traditional fallow periods significantly reduced N_2O emissions, probably because cover crops actively scavenged soil N and led to decreased N_2O by reducing soil NO_3 pools (Thorup-Kristensen et al. 2003). This suggests that including cover crops to reduce bare fallows could make significant contributions toward reducing N_2O emissions, as long as they are managed appropriately during the cash crop growing season. We found no significant effect of cover crops on N_2O emissions during the cash crop growing season.

While the small number of studies investigating the impact of cover cropping and other ecological nutrient management strategies is a major issue limiting our analysis, this limitation is amplified by the fact that in many studies rotation, use of cover crops and the amount of N added were confounded. Specifically, in most of these studies, the ENM treatments received greater N inputs compared to the controls. Most investigators did not reduce N fertilizer applications to account for the value of N released from the cover crop biomass, a standard recommendation in the extension literature. Nitrogen additions from cover crops can be substantial in the case of leguminous biomass and adding fertilizer N in conjunction with N-rich legume biomass usually results in significant N over-application (c.f. Komatsuzaki et al. 2008). The positive correlation between LRR and extra N inputs from cover crops indicates that, as with fertilizers and manure, the amount of N inputs is an important driver of N_2O emissions. Furthermore, as has proven to be the case in the manure dataset, assessing the impacts of cover crops on N_2O emissions when cover crop treatments are receiving more total N confounds the effect of the quantity of N inputs with the effect of using cover crops as an N source. Therefore, our finding that N_2O emissions during the cash crop growth phase were not significantly different following cover crops compared to N fertilizer must be viewed with caution because the cover cropped treatments received greater total N inputs compared to the fertilizer controls in all but one study (Robertson et al. 2000). In fact, the two studies that reduced fertilizer inputs based on N from green manures reported similar or reduced N_2O emissions with green manures (Robertson et al. 2000; Alluvione et al. 2010).

In contrast with our findings, Basche et al. (2014) reported that legume cover crops significantly increased N_2O emissions compared to non-legume cover crops during the cash crop growth phase. Similar to our analysis, Basche et al. (2014) included field studies measuring N_2O during the grain crop growing periods or all year round, but they also included studies we excluded, such as field experiments and growth chamber incubations that measured N_2O emissions for short periods (less than 60 days) following cover crop incorporation/killing. Half of the observations they reported showing much greater N_2O emissions from legumes compared to non-legumes are from studies comparing cover crop biomass to controls with zero N additions (e.g. Baggs et al. 2000; Millar et al. 2004; Basche et al. 2014; Fig. 2). Their conclusions reflect the fact that when cover crop treatments are compared to a control receiving zero N, N_2O emissions will be largely driven by the amount of N and C:N ratio of the biomass and will most certainly be greater compared to a zero N control. This would also be the case if N fertilizer treatments were compared to zero N controls. To better inform N fertilizer management in conjunction with cover crops, we need to understand how fertilizer additions interact with cover crop biomass. To do this, a broader range of treatments that include different N fertilizer rates ± cover crops will need to be included in experimental designs (c.f. Stivers and Shennan 1991).

For example, an experimental design that compares the impact of cover crops alone to a zero N control as well as a treatment of cover crop + reduced N fertilizer can shed light on the interactions between N fertilizer and cover crops when they are used together. In one study N_2O emissions were measured under maize receiving N inputs from alfalfa with ±N and controls with ±N fertilizer (Drury et al. 2014). Comparing the 0 N control to the plowed-down alfalfa alone resulted in a large response ratio (R = 5.78, N_2O emissions 0.51 and 2.95 kg N_2O-N ha^{-1}, from 0 N and legume treatment, respectively). In contrast, comparing the alfalfa with the fertilizer N control resulted in a response ratio of 0.40 (N_2O from N fertilizer at 129 kg ha^{-1} = 7.36 kg N_2O-N ha^{-1} versus 2.95 kg N_2O-N ha^{-1} receiving 242 kg N ha^{-1} from alfalfa). When the +N control and +N alfalfa treatments receiving the same amount of N fertilizers were compared, the response ratio was reduced to 0.88, but N_2O emissions from both treatments were much greater (7.36 and 6.46 kg N_2O-N ha^{-1}, respectively). The greatest differences in N_2O emissions were observed between the zero N control versus the treatments receiving either fertilizer N only or fertilizer N + legume biomass (N_2O from 0 N = 0.51 compared to 7.36 and 6.46 kg N ha^{-1}, respectively) suggesting that among these scenarios the N fertilizer was the N source with the greatest tendency to increase N_2O emissions. While results from a single study are interesting, our main point in highlighting these results is to demonstrate the importance of using a more comprehensive experimental design to elucidate interactions between N sources and distinguish between the impact of application rate and composition of added N.

Even in the face of greater N additions to cover cropped treatments, we found a trend that C:N ratios correlated inversely with the N_2O emissions. Previous field and laboratory studies have reported such relationships under similar environmental conditions (Huang et al. 2004; Gomes et al. 2009). The fact that this relationship emerged despite environmental variation further highlights the importance of C abundance in regulating N_2O emissions and supports the idea that C and N should be managed in conjunction with one another in order to improve N use efficiency and N retention (Drinkwater and Snapp 2007; Fisk et al. 2015).

In the *Diversified Rotation* category, when fertilizer rates for maize were not reduced based on the residual N from the preceding legume crops, increased N_2O emissions occurred during the maize phase of the rotation after legume crops (Mosier et al. 2006; Drury et al. 2008; Halvorson et al. 2008; Halvorson et al. 2010). In other studies where fertilizer rates for the non-legume cash crops were reduced there was no increase in N_2O emissions in grains following legumes and the cumulative emissions in the diversified rotations were either lower or not significantly different compared to the monocultures (Adviento-Borbe et al. 2007; Barton et al. 2013).

Tradeoff between yield and N_2O mitigation

Our analysis suggests that farmers could achieve significant N_2O mitigation with only minor yield losses, by implementing well-designed N management strategies. It is worth noting that yield loss does not necessarily translate into economic losses. Matching crop demands with economic returns rather than applying N for maximum agronomic yields could avoid substantial N_2O emission and achieve yield–environment co-benefits (McSwiney and Robertson 2005; Robertson and Vitousek 2009; Hoben et al. 2011; Linquist et al. 2012). Furthermore, studies of FUE suggest that comprehensive implementation of the 4Rs strategy could enable reductions in N fertilizer rates without incurring yield reductions (Meng et al. 2016; Sela et al. 2016). Clearly, there is great potential in achieving N_2O and yield "win–win" outcomes that have yet to be explored.

About half of the studies in our dataset did not report yield information. Although there were increasing numbers of papers reporting both area-scaled and yield-scaled emissions, we still saw a segregation of "agronomic" literature that focused on yield improvement, and "environmental" literature that focused greenhouse gas emissions. Maintaining crop production and improving N management to minimize environmental consequences should become the dual-targets of research on N_2O emissions from agricultural systems. The multifunctionality of agriculture should be recognized and future research should aim towards reconciling agricultural productivity and environmental integrity (Robertson and Swinton 2005).

Improving empirical research on N_2O emissions from agriculture

Linking management practices to N_2O emissions is particularly challenging due to multiple, interacting proximate and distal factors that drive nitrification/denitrification and ultimately determine N_2O emission rates (Robertson 1989; Chapin et al. 2011). In addition to issues related to the problem of confounding variability in N source with N quantity commonly found in studies of alternative N sources, other aspects of the dataset limited our ability to explain the variability in N_2O emissions within and across management practices. Better understanding of how to successfully mitigate N_2O emissions from agricultural systems will require studies of N_2O fluxes in agroecosystems to account for a broad range of biotic and abiotic factors, beginning with ecosystem state factors. Ecosystem state factors such as soil parent material, climate and topography play a role in shaping agroecosystems (i.e. crops that are grown, irrigation, and nutrient management regimes) and also interact with management practices to effect N_2O emissions. Nevertheless, many studies did not report basic soil information and very few considered interactions between the management practices and ecosystem state factors. For instance, while many studies recognized the importance of landscape characteristics on N_2O emissions (Corre et al. 1996; Castellano et al. 2010; Vilain et al. 2010; Gu et al. 2011; Li et al. 2012; Schelde et al. 2011) only few examined the interaction between landscape characteristics and management strategies (Sehy et al. 2003; Izaurralde et al. 2004; Negassa et al. 2015). Sehy et al. (2003) found that adjusting fertilizer rates for different landscape positions was effective at reducing N_2O.

In addition to the limitations resulting from the focus on single alterations in 4Rs fertilizer management practices discussed above, interactions among N management, N sources and other practices, which indirectly influence N cycling have rarely been investigated. For example, Petersen et al. (2011) found a significant interaction between tillage and the use of cover crops in a manure-amended luvisol. Using cover crops together with conventional tillage generated much higher N_2O emissions compared to reduced tillage, both with or without cover crops. The authors suggested that conventional tillage created better contact between cover crop residues, slurry and soil, resulting in rapid decomposition and anaerobic microsites, which stimulated N_2O emissions. Kallenbach et al. (2010) found the effect of cover crops differed depending on the type of irrigation and the irrigation by cover crop interaction varied between growing season and non-growing season phases of the rotation. Studies designed to investigate these kinds of interactions need to be expanded in order to fully understand which combinations of management practices maintain yields while reducing surplus N and N_2O emissions.

Temporal dynamics of N cycling and N_2O emissions also need more attention. Eighty out of the 134 studies in our dataset reported N_2O emissions only for grain crop growing seasons and did not measure N_2O during intervals between cash crops. Thus, a large proportion of studies missed N_2O emissions during winter or spring-thaw periods which can account for up to 70% of the annual N_2O budget (Nyborg et al. 1997; Wagner-Riddle and Thurtell 1998; Teepe et al. 2001; Dörsch et al. 2004). Future studies must capture measurements during these crucial periods and report N_2O emissions for these different phases in the cropping cycling as well as in aggregate for the annual budget. Another issue that has received less attention is the impact of management-induced changes occurring over differing time frames. Van Kessel et al. (2013) found that no-till or reduced tillage reduced N_2O emissions compared to conventional tillage, but this only occurred after >10 years of implementation. Fisk et al. (2015) found that different sources of C affect N cycling and retention at differing time-scales. The low quality, high C:N residues applied over a period of ten years impacted long term N retention and increased soil organic N reservoirs while the highly labile C additions simulating root exudates impacted fast N cycling processes such as the competition for NH_4^+ between nitrification and immobilization. Both of these changes in N cycling contributed to reduced N losses via different mechanisms occurring at different timescales. Future experiments should be designed to distinguish between the effects of management practices in the immediate growing season as well as over longer timeframes in order to design Best Management Practices that adapt to management-induced changes in the soil environment.

Given the large number of interacting factors that influence N_2O emissions, it is not surprising that developing biogeochemical models that can accurately predict N_2O emissions is extremely challenging (Butterbach-Bahl et al. 2013). While it is evident from the previous discussions that no single study can investigate the full range of drivers, the potential for each study to contribute to our ability to develop useful process models and Best Management Practices will be greatly enhanced if experimental designs are improved to reflect our current understanding of the abiotic and biotic factors driving N_2O in conjunction with more comprehensive reporting of ecosystem state factors, management and edaphic conditions. Here we provide a checklist for information that could greatly enhance the value of studies on N cycling and N_2O for model development and cross-site analyses (Table 3). In addition to the listed information, reporting on some well-developed indicators could greatly enhance our ability to compare environmental factors across studies. For example, Nitrate Intensity, the summation of daily soil nitrate concentration at 0–15 cm depth over the same period of N_2O measurement is a case in point, proposed by Burton et al. (2008). This indicator has proven useful and could be reported as standard quantitative indicator for N availability. The same approach could be used for other variables that are typically measured in concert with N_2O measurements including DOC, VWC, relative gas diffusivity and WFPS (Ball et al. 2014).

From microbes to comprehensive global warming potential accounting

Finally, two additional issues are crucial for developing comprehensive plans for mitigating N_2O emissions in agriculture. First, the role of soil microbial community composition, especially with respect to relevant functional groups that carry out nitrification/denitrification is an important biotic factor driving N_2O emissions which, until recently has been impossible to characterize. There has been a recent push to establish correlation relationships between the abundance of functional genes, corresponding process rates such as nitrification/denitrification and resulting N_2O fluxes with mixed outcomes (Bier et al. 2015; Rocca et al. 2015). DNA-based measurements reflect long-term management history and therefore can be used as indicators or predictors of potential process rates (Morales et al. 2010; Petersen et al. 2012). However, using gene abundance to predict N_2O emissions in the field remains elusive, probably because in situ process rates reflect gene abundance as well as gene expression and other downstream regulatory steps which are influenced by changes in edaphic factors (Philippot and Hallin 2005). With more state-of-art technologies becoming available in the future, we will soon have a better understanding of the mechanisms controlling gene abundance, transcription and translation and enzyme activities. This knowledge can be used to determine the spatial and temporal scales at which including information on microbial communities can be used in predicting

Table 3 Management and environmental factors that should be reported in agricultural N management and N_2O emission studies

Environmental information	Agronomical information
Climatic information	Management history
Soil order/series	Crop rotation (species and cultivar)
Soil texture and slope	Grain yield
Soil pH	Properties of N additions: fertilizer/manure/cover crop biomass, (N and C content, inorganic N, moisture, etc.)
Soil temperature/moisture	N application timing and method
Soil NO_3, NH_4	Other management practices held constant across treatments (tillage, irrigation, etc.)
Total C and N	
Other soil N and C pools such as DON, DOC	

process rates and N_2O emissions (Levy-Booth et al. 2014).

Second, we know that soil N_2O emissions are only part of the GHG footprint resulting from any agricultural system. A full accounting of upstream GHG emissions, such as Life Cycle Analysis, can account for N_2O resulting from manufacturing processes in the case of N fertilizer or from handling of materials such as manures. Agricultural management systems differ greatly in terms of GHG emissions generated from the inputs, as well as those occurring in the field. For example, legume based grain systems tend to have reduced GHG emissions from inputs because biologically fixed N replaces Haber–Bosch N, which is an energy intensive product. A recent study by Sainju (2016) synthesized the net Global Warming Potential (GWP) of tillage, fertilizer management and crop rotations. The results showed that no-till systems greatly reduced the net GWP compared to conventional tillage, and perennial cropping systems had lower GWP compared to annual cropping systems. The three experiments included in the synthesis suggested reduced GWP when legumes were added into small-grain rotations, however, limited data made it difficult to reach conclusions about the impact of diversified rotations or cover crops and continuous maize and maize-soybean rotations dominated the dataset. If these upstream findings are combined with the results from our analysis of field-level emissions, many practices that show no significant effect in the field may actually result in a net reduction of N_2O emissions because of reductions in upstream emissions. Relying only on field losses to guide policy decisions will not produce outcomes that will lead to net reductions of N_2O or other greenhouse gases.

Conclusion

We were able to reach several key conclusions from our analyses. We join other authors in calling for careful management of N fertilizer with continued efforts to reduce N application rates since N_2O emissions are extremely sensitive to N fertilizer rate. We specifically recommend that studies of reduced N rates in conjunction with integrated 4Rs strategies be undertaken. Furthermore, all other things being equal, greater N additions lead to increased N_2O emissions, regardless of the N source (i.e. N fertilizer, green manure or animal manures). For example, increased N_2O emissions from manures compared to N fertilizers can be attributed to greater N additions in manure treatments rather than the use of manures per se. However, manure and N fertilizer interact with soil texture differently suggesting that fine-tuning N management to reflect ecosystems state factors is a necessary next step in N_2O mitigation. While only small portion of past studies on management practices reported both yield improvement and N_2O mitigation, there is a great potential for achieving environmental-economic co-benefits through improved N management provided these two outcomes are studied together. ENM practices generally had N_2O emissions during the cash crop growing season that were not significantly different from conventional fertilizer-based practices; however this outcome is based on a small number of studies and N was frequently over-applied in the ENM systems suggesting it is possible that with strategic reductions in N rates N_2O emissions could be reduced. We found strong evidence that replacing bare fallows with covers or cash crops reduces N_2O, and this is in keeping with findings that

NO_3^- leaching is also reduced under these circumstances. Further study of ENM practices for N_2O mitigation and overall reduction in agricultural N losses is warranted. Taken together, our integrated analysis suggests that there is great potential for applying N saturation theory to systematically test hypotheses and develop greater understanding of the interactions among management practices and between environmental factors and management regimes.

The complexity of N_2O production processes requires that cross-scale and interdisciplinary studies are conducted in order to fully understand how to best reduce these emissions from food production systems. At the field scale, agronomic information and N_2O measurements could be used together to design mitigation strategies that reduce carbon footprints and maximize economic benefits. At the landscape scale, understanding the interaction of landscape characteristics and management practices would facilitate the design of Best Management Practices that account for landscape variability. Information on the soil microbial community could improve the prediction of N_2O emissions at the field scale and across land uses.

Acknowledgements We thank Christina Tonitto and Jennifer Blesh for feedback on the data analysis and comments on an earlier version of this manuscript. We are grateful for two anonymous reviewers who provided insightful comments during the review process. Funding for this work was provided by United States Department of Agriculture-National Institute of Food and Agriculture (2012-68002-20081).

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