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Meta-analysis on the potential for increasing nitrogen losses from intensifying tropical agriculture

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Abstract

Fertilized temperate croplands export large amounts of reactive nitrogen (N), which degrades water and air quality and contributes to climate change. Fertilizer use is poised to increase in the tropics, where widespread food insecurity persists and increased agricultural productivity will be needed, but much less is known about the potential consequences of increased tropical N fertilizer application. We conducted a meta-analysis of tropical field studies of nitrate leaching, nitrous oxide emissions, nitric oxide emissions, and ammonia volatilization totaling more than 1,000 observations. We found that the relationship between N inputs and losses differed little between temperate and tropical croplands, although total nitric oxide losses were higher in the tropics. Among the potential drivers we studied, the N input rate controlled all N losses, but soil texture and water inputs also controlled hydrological N losses. Irrigated systems had significantly higher losses of ammonia, and pasture agroecosystems had higher nitric oxide losses. Tripling of fertilizer N inputs to tropical croplands from 50 to 150 kg N ha^{-1} $year^{-1}$ would have substantial environmental implications and would lead to increases in nitrate leaching (+30%), nitrous oxide emissions (+30%), nitric oxide (+66%) emissions, and ammonia volatilization (+74%), bringing tropical agricultural nitrate, nitrous oxide, and ammonia losses in line with temperate losses and raising nitric oxide losses above them.

KEYWORDS
ammonia volatilization, meta-analysis, nitrate leaching, nitric oxide, nitrous oxide, reactive nitrogen, tropical agriculture

1 | INTRODUCTION

The world's growing population will require increased grain production. Increases in global grain demand from 60% to 110% (Alexandratos & Bruinsma, 2012; Tilman, Balzer, Hill, & Befort, 2011) from 2005 levels or 25%–70% from 2014 levels (Hunter, Smith, Schipanski, Atwood, & Mortensen, 2017) are projected by 2050. Much of this increase is expected to come from tropical regions. A potential way to meet these demands is to increase productivity on existing agricultural lands ("intensification"), which can minimize negative externalities from land clearing such as habitat loss, biodiversity loss, and greenhouse gas emissions (Hunter et al., 2017). Intensification is a particularly attractive option in tropical croplands, many of which are underfertilized (Conant, Berdanier, & Grace, 2013; Liu et al., 2010; Mueller et al., 2014; Vitousek et al., 2009). In addition to helping meet global demand, intensification of tropical croplands is urgently needed to meet regional and local food security demands (Hazell & Wood, 2008). Intensification has already begun in some tropical regions, such as the Brazilian Cerrado (Spera, Galford, Coe, Macedo, & Mustard, 2016), where higher nitrogen (N) fertilizer use has been occurring since the 1990s.

Evidence from well-studied temperate agroecosystems shows that, although intensification increases production, it also degrades the environment and threatens human health. Roughly half of N inputs in temperate croplands reach crop biomass in a given crop cycle (Cassman, Dobermann, & Walters, 2002; Conant et al., 2013), and much of the excess (or asynchronous) N inputs are lost from agroecosystems. In order to combat this inefficiency, N fertilizer application has increased and is a primary driver of the doubling of global reactive N inputs to the biosphere over natural inputs (Fowler et al., 2013). Specifically, 22% (in wheat) and 15% (in maize) of fertilizer N are lost as nitrate (NO_3^-) in water (Zhou & Butterbach-Bahl, 2014), and 1% of N fertilizer is lost as the gas nitrous oxide (N_2O; De Klein et al., 2006). However, recent studies found that N_2O responses to N inputs are nonlinear (Hoben, Gehl, Millar, Grace, & Robertson, 2011; Shcherbak, Millar, & Robertson, 2014), including in tropical agroecosystems (Hickman, Tully, Groffman, Diru, & Palm, 2015), implying that linear loss models such as IPCC emission factors overestimate losses at low N rates and underestimate them at high N rates. Approximately 0.5% of applied N is lost as the gas nitric oxide (NO; Stehfest & Bouwman, 2006; Veldkamp & Keller, 1997), and 14% of applied N is lost via ammonia volatilization (Bouwman, Boumans, & Batjes, 2002). These four losses have important environmental consequences: NO_3^- degrades water quality, N_2O contributes to climate change, and destroys stratospheric ozone, and NO and NH_3 contribute to regional air pollution and N deposition (Ciais et al., 2013; Galloway et al., 2003; Tilman, Cassman, Matson, Naylor, & Polasky, 2002).

Most of what we know about the inefficiency of N fertilizer use comes from temperate North America, Europe, and Asia (Liu et al., 2010). Although there have been a number of recent meta-analyses on N losses from agroecosystems, they include very few observations from tropical agriculture (Abdalla et al., 2019; Liu et al., 2017; Shcherbak et al., 2014; Zhou & Butterbach-Bahl, 2014). For example, Shcherbak et al., 2014 included only five studies from the tropics. Although process-based models exist for predicting N losses based on climate data such as the DeNitrification DeComposition model (Gilhespy et al., 2014), temperate and tropical cropping systems are driven by factors other than climate such as soil and crop types. From a mechanistic perspective, N loss rates likely depend on factors such as climate, soil, crop type, and the details of fertilization. These factors vary across temperate and tropical regions, so we might expect different N loss rates in the tropics. However, a review of N_2O emissions in the tropics and subtropics found that mean emissions were 1.2% of applied N (Albanito et al., 2017), only slightly higher than in temperate agroecosystems. There have been studies from individual fields across the tropics, but as of yet these studies have not been aggregated in a way that evaluates all four major losses of inorganic N, compares them to temperate N losses, or identifies environmental or management drivers. Therefore, the extent to which increasing tropical N inputs will contribute to reactive N pollution is not well constrained.

Here, we conducted a systematic review and meta-analysis to answer: (a) Does the relationship of N losses relative to N inputs differ between tropical and temperate regions? (b) How do NO_3^-, N_2O, NO, and NH_3 losses from tropical agricultural systems respond to N inputs? and (c) What environmental and management factors control the magnitude of N losses in tropical agroecosystems?

2 | MATERIALS AND METHODS

2.1 | Literature review and data collection

We used a systematic search to capture a wide range of studies across diverse environmental and management conditions and to minimize bias; the publication cut-off date was November 9, 2018. We used Web of Science and Elsevier's Scopus academic search databases because of their broad coverage and ability to handle complex search strings. The search string we used was tested and constructed to yield the most relevant and numerous results until reaching a large number of duplicates between the two search engines (indicating a comprehensive coverage of the relevant literature). The full search terms and dates run (Supporting Information) took the general structure: "(nitrous oxide OR N_2O) and (agricultur* OR soil OR crop) and (fertilizer OR compost OR input OR nitrogen) and (tropic*)," as well as a comprehensive list of tropical countries and specific tropical states or provinces of large countries (i.e., Hawaii, U.S. or Yunnan, China). We used Boolean "OR" operators to be inclusive as possible within subject searches and used "AND" to connect subjects; wildcard characters (e.g., asterisk "*") represented any group of characters, including no character, and question mark "?" represented any single character to allow for combinations of words such as "agriculture" and "agricultural" or different spellings such as "fertiliser" versus "fertilizer."

We removed duplicate results and reviewed more than 350 studies of NO_3^- leaching, 950 of N_2O emissions, 400 of NO emissions, and 1,200 of NH_3 volatilization. Next, we screened results by inclusion criteria: That the study: (a) occurred the tropics (23.4°N–23.4°S), (b) was measured under field conditions at a the plot-scale (we excluded lab-based or modeling calculations and watershed-scale studies for NO_3^- since they were much larger scale and much less common than the vast majority of leaching studies), (c) specified N input rate and type (inorganic or organic); (d) reported losses in kg N ha^{-1} study $period^{-1}$, (e) calculated cumulative seasonal N_2O, NO, or NH_3 fluxes from at least three individual flux measurements (Liu et al., 2017; applies only to studies including trace gas measurements), (f) used direct flux measurement methods (i.e., indirect methods for NO_3^- such as two-time measurements of vertical soil NO_3^- concentration differences through soil profile were excluded), and (g) was written in English. First, we screened titles and abstracts and eliminated papers if any

TABLE 1 Tropical observations and papers by type of N loss reported

	NO_3^-	N_2O	NO	NH_3
Papers meeting search criteria	23	93	11	21
Observations	193	597	49	193
Regional observations				
South and Central America	12.4% (*n* = 24)	31.8% (*n* = 190)	65.3% (*n* = 32)	59.6% (*n* = 115)
Asia and Oceania	4.7% (*n* = 9)	53.1% (*n* = 317)	4.1% (*n* = 2)	24.9% (*n* = 48)
Sub-Saharan Africa	78.2% (*n* = 151)	12.1% (*n* = 72)	30.6% (*n* = 15)	12.4% (*n* = 24)
Other	4.7% (*n* = 9)	3.0% (*n* = 18)	0% (*n* = 0)	3.1% (*n* = 6)
Oxisol or ultisol soil types[a]	22.8% (*n* = 44)	36.5% (*n* = 218)	34.7% (*n* = 17)	38.3% (*n* = 74)

[a]Percent of observations where Oxisol or Ultisol soils were reported (excluding observations where soil type was not reported).

of the inclusion criteria were not met; then we screened the full texts of the remaining papers and extracted data from those that fit. In an effort to increase the sample size for the NO_3^- and NO databases, we also searched the references cited within papers that met our inclusion criteria for relevant papers. We searched within any meta-analyses or review papers from our search results to find additional field studies to add to the NO_3^- leaching (including one unpublished review by co-author K. Tully) and NO databases. We also included a co-author's unpublished tropical NO_3^- leaching data from Tanzania (Tully, data available upon request). From the initial set plus the additional studies, a total of 23, 93, 11, and 21 papers fit the inclusion criteria for NO_3^- leaching, N_2O, NO, and NH_3, respectively (Table 1).

We extracted data from all relevant papers and entered them into a database; we digitized figures for papers which did not report the data we needed in text or tables using the software Plot Digitizer (Huwaldt, 2015). We recorded individual measurements instead of treatment means when possible and recorded standard errors (when provided) for each response variable, kg $NO_3^-/N_2O/NO/NH_3$-N ha^{-1} study $period^{-1}$. In addition, we recorded data on site factors (location, soil type, soil texture, mean annual precipitation, and soil pH) and management information (crop type, fertilizer type, summed fertilizer, and biologically fixed N input rate (when reported), fertilizer application frequency, irrigation, and whether organic inputs or N-fixing crops were used). When the total number of N_2O, NO, or NH_3 measurements was not reported, we calculated estimates for the number of measurements either based on the written description of sampling frequency or by counting the points on flux graphs. We considered including several other site variables, including total soil N, soil extractable NO_3^- and ammonium concentrations, soil carbon, soil porosity, and other data, but they were reported too infrequently to be included in the analysis. For NO_3^- leaching, we used only the deepest measurement of leaching if multiple depths were reported (observations ranged from 40 to 200 cm depth, with a median of 100 cm), and we combined precipitation and irrigation data as total water input (mm) whereas we used mean annual precipitation data in the N_2O, NO, and NH_3 datasets. Almost all of the soil pH values were reported as soil pH in water; however, two of the N_2O and one of the NO_3^- observations were measured in calcium chloride, and we used a conversion factor of pH in calcium chloride + 0.8 to approximate the pH in water (Sanchez, 2019). When the precise details were not reported, we approximated latitude and longitude coordinates by searching for study location, soil texture by estimating percentages from qualitative descriptions such as "sandy clay loam", and study duration by multiplying the number of months in the study by 30 days. If not reported quantitatively or qualitatively, we extracted percent sand soil texture from ISRIC's Soil Grids (0.05 m; Hengl et al., 2017) and estimated mean annual precipitation (mm/year) based on a long-term mean from 1950 to 2000 (Fekete, Vörösmarty, & Grabs, 2002). Crops were recorded as described in the publications and grouped into several types: cereal (primarily wheat and maize, barley, and one instance of upland rice), rice (flooded), legume, pasture, fallow, tree crop, and other. We split flooded rice into two categories. There were sufficient data to separate continuously flooded from non-continuously flooded (i.e., alternate wetting and drying) rice irrigation for N_2O, and all of the NH_3 observations of rice were continuously flooded. There were too few data to estimate differences between flooding management for NO_3^-, so we did not separate them in the model specified by Equation (3). "Fallow" in some parts of tropics implies a vegetative cover which differs from the bare field fallow we intend for this category; the combination of both fallow types in our data may explain the wide variation in tropical fallow NO_3^- leaching losses. We used data from temperate sites in previous meta-analyses that also met our inclusion criteria on NO_3^- leaching, NO and N_2O emissions, and NH_3 volatilization (Abdalla et al., 2019; Liu et al., 2017; Pan, Lam, Mosier, Luo, & Chen, 2016; Stehfest & Bouwman, 2006; Zhou & Butterbach-Bahl, 2014) to examine differences between temperate and tropical regions.

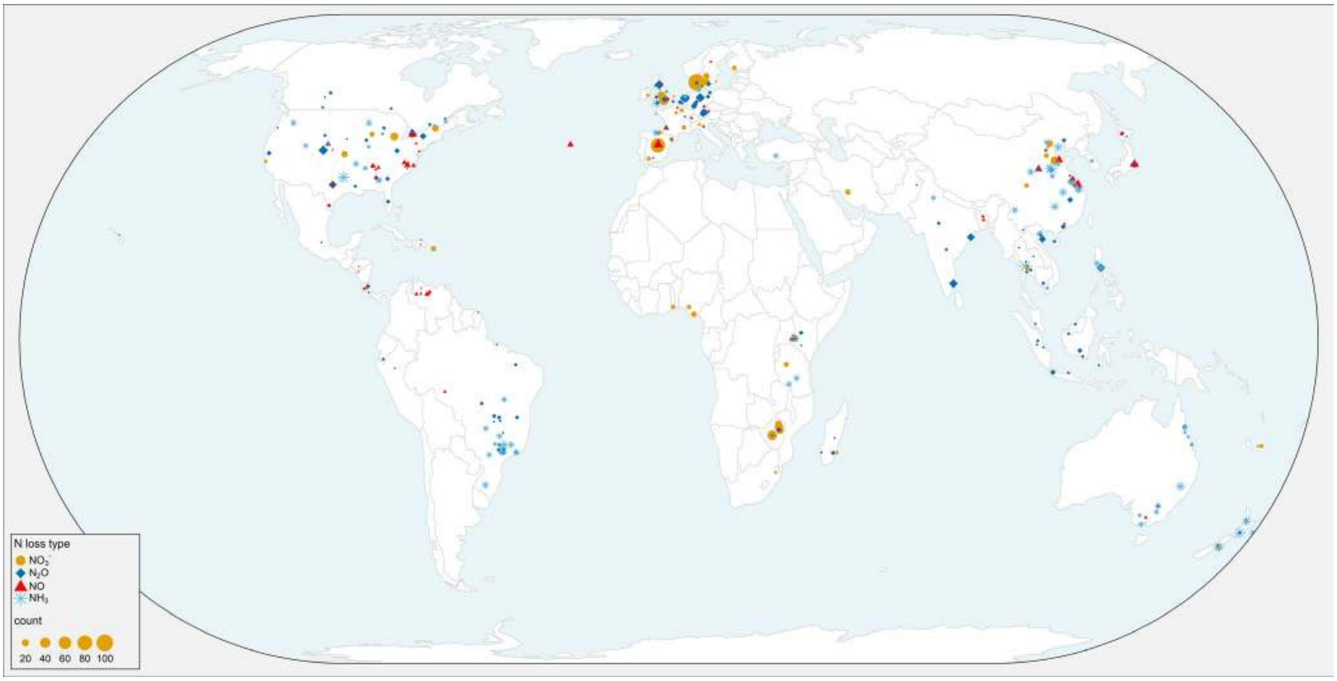

FIGURE 1 Map of the tropical sites collected in this study and the temperate reference study sites gathered in previous reviews to which we compared our tropical dataset. Each symbol indicates one site. The shape of the symbol indicates the type of loss measured and the size indicates the number of observations from that site (n = 62 sites for NO_3^-, n = 217 sites for N_2O, n = 69 sites for NO, and n = 87 sites for NH_3)

2.2 | Characteristics of the tropical dataset

The 148 tropical studies in our meta-analysis came from all major regions—South and Central America, sub-Saharan Africa, and Asia/Oceania—but were not evenly distributed (Figure 1, Table 1). Many NO_3^- leaching observations came from sub-Saharan Africa, whereas many N_2O observations came from Asia/Oceania and many of NO observations came from South and Central America (Table 1). Studies of NO_3^- (58%) and NO (71%) but not N_2O (22%) or NH_3 (35%) were focused on cereal crops (Table S1). There were high proportions of "other" tropical crop types (e.g., sugarcane) in the NO_3^-, N_2O, and NH_3 studies, of "flooded rice" (continuously and non-continuously flooded), and pasture in the NO studies (Table S1). We investigated the quality of the tropical data we collected (the selection criteria for details). The median numbers of replicates within a study were 9 for NO_3^-, 4 for N_2O, 8 for NO, and 4 for NH_3. The median number of measurements throughout the season to calculate the cumulative N losses were 22 for NO_3^-, 25 for N_2O, 38 for NO, and 11 for NH_3. The median standard errors of the mean of N losses (kg N/ha) were 1.7 for NO_3^-, 0.2 for N_2O, 1.5 for NO, and 1.9 for NH_3.

2.3 | Statistical analysis

We analyzed data and created plots in the software R (version 3.5.0; R Core Team, 2018). The data for these analyses are available in supporting datasets S1–4, and the code is available for download at https://github.com/ahuddell/N_loss_metaanalysis. We used the *ggplot2* (Wickham & Chang, 2018) package for some plots and used the *tmap* (Tennekes, 2018) package for Figure 1. We standardized all continuous covariates for the statistical analysis by subtracting the mean and dividing by two standard deviations so that the estimated coefficients were unitless and directly comparable to untransformed binary predictors (Gelman, 2008). We checked all potential covariates for collinearity with variance inflation factors (VIFs) from the *car* package in R (Fox, Weisberg, & Price, 2018), and those with VIFs > 3 were removed (Zuur, Ieno, & Elphick, 2010). The response variables were over-dispersed and non-normally distributed, so we first replaced negative and zero values with one half of the positive minimum value observed and then $\log_{10}$-transformed each response variable to normalize each response variable.

We fit linear mixed effect models with restricted maximum likelihood and a Gaussian error distribution for each question using the lmer function in the *lme4* package for R (Bates, Maechler, Bolker, & Walker, 2018). We chose this hierarchical modeling approach to deal with the non-independence of repeated, grouped observations from the same site (we recorded data for different treatments or time periods from the same site as individual observations; Pinheiro & Bates, 2000). We evaluated the data for assumptions of linear mixed effect models such as homogeneity of variance, normality of residuals, linearity in each of the covariates, and normality and significance of the random effects. To evaluate whether N loss responses to N inputs differ between tropical and temperate agricultural systems, we fit linear mixed effect models for each response variable (Equation 1 below):

$$\log(\text{N loss}) = \beta_0 + \gamma_i + \beta_1 \times Nin + \beta_2 \times d + \beta_3 \times t + \beta_4 \times Nin \times t + \beta_5 \times sdep + \varepsilon, \quad (1)$$

where $\log(\text{N loss})$ is $\log_{10}$-transformed N loss (originally in kg N ha^{-1} $season^{-1}$ but now unitless after log transformation (Matta, Massa, Gubskaya, & Knoll, 2011)) for NO_3^-, N_2O, NO, or NH_3; β_0 is the fixed-effect intercept; γ_i is the random variation in the intercept for each site *i*, using site latitude as a unique identifier; β_1 is the coefficient of *Nin*, the sum of all inorganic and organic N inputs (originally in kg N ha^{-1} $year^{-1}$, but unitless because of the standardization $Nin = (Nin_{\text{original}} - \text{mean}(Nin_{\text{original}})) / 2 \times sd(Nin_{\text{original}})$, as described above); β_2 is the coefficient for *d*, study duration (originally in days, but standardized in the same way as *Nin*); β_3 is the coefficient for *t*, tropical or temperate (tropical = 1, temperate = 0); β_4 is the coefficient for the interaction of N inputs with tropical versus temperate (β_1 is the temperate response to N inputs and β_4 is the difference between the tropical and temperate responses to N inputs, so $\beta_1 + \beta_4$ is the tropical response to N inputs); β_5 is the coefficient for sdep, the sample depth of the lysimeter (cm) which was only included for NO_3^- leaching; and ε is unexplained residual variation. We calculated *p*-values on linear mixed-effects models with the *lmerTest* package (Kuznetsova, Brockhoff, & Christensen, 2017). We used population prediction intervals (Bolker, 2008) to calculate confidence limits using the mvrnorm function in the *MASS* package in R (Venables & Ripley, 2002) for Figure 2. We backtransformed model outputs from log to linear space and unstandardized coefficients for figures and interpretation, but the values reported in Table 2 and Table S2 are for transformed and standardized values. As shown in Equation (2) below, the unfertilized baseline values in Table 2 are calculated at the mean of γ_i (the "average" site), the mean of *d* (the average study duration), the mean of *sdep* (the average lysimeter sample depth), the mean of ε (the average random variation), and an *Nin* value of 0 (to get the expectation for an unfertilized plot). All of these mean values of standardized covariates are 0, so the unfertilized baseline values in Table 2 are given by:

$$\text{N loss} = 10^{\left(\beta_0 + \beta_3 \times t - (\beta_1 + \beta_4 \times t) \times \frac{\text{mean}(Nin_{\text{original}})}{2 \times sd(Nin_{\text{original}})}\right)}. \quad (2)$$

To explore the degree to which environmental and management factors such as soil texture, soil pH, climate, crop type, irrigation, and management practices explain N losses, we selected environmental and management variables that were commonly reported in the literature and for which a priori biogeochemical or physical links to each response variable exist (Anderson & Burnham, 2002). For example, we tested for an effect when organic N was incorporated into the cropping system via biological N fixation or organic N amendments based on a previous meta-analysis which found that coupling N and carbon inputs to temperate grain crops most effectively increased N retention (Gardner & Drinkwater, 2009). We avoided model selection because our goal was to understand mechanisms explaining N losses rather than to optimize N loss predictions. As we did for the data described for Equation (1), we replaced negative and zero values with one half of the positive minimum value observed, then $\log_{10}$-transformed each response variable. For this question, we also scaled each N loss by the study duration (days) to best capture the variation driven by differences in study length. The model we used to address environmental and management controls is (Equation 3 below)

$$\log(\text{N loss}_d) = \beta_0 + \gamma_i + \beta_1 \times Nin + \beta_2 \times st + \beta_3 \times p + \beta_4 \times i + \beta_5 \times o + \beta_6 \times sa + \beta_7 \times \text{pH} + \beta_{8j} \times c_j + \beta_9 \times sdep + \varepsilon, \quad (3)$$

where $\log(\text{N loss}_d)$ is $\log_{10}$-transformed N loss (originally in g N ha^{-1} day^{-1} but now unitless after log transformation; Matta et al.,

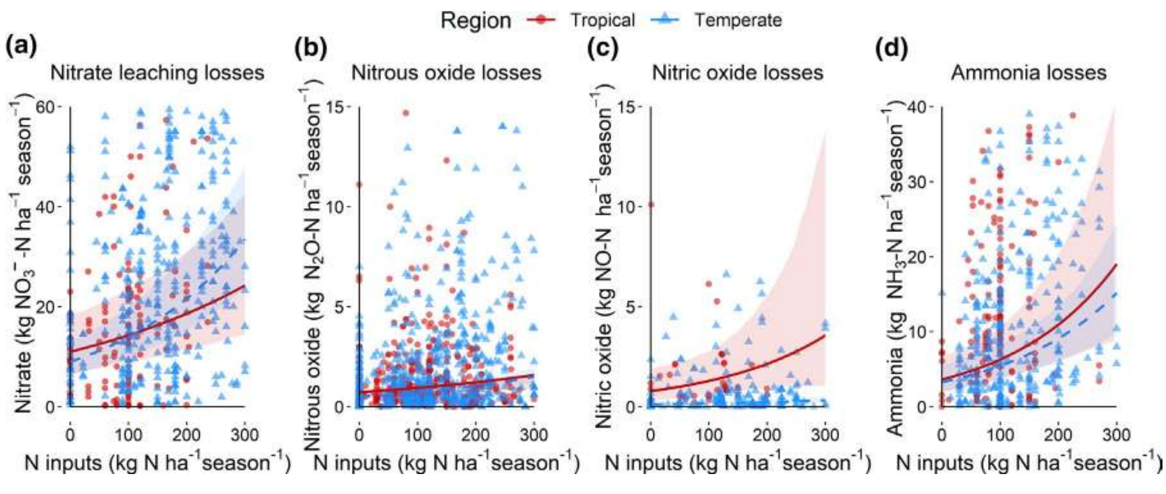

FIGURE 2 Tropical versus temperate N losses as functions of N inputs. Each point (red = tropical, blue = temperate) is an observation. Modeled means (curves) and 95% confidence intervals (shaded regions) are shown for (a) NO_3^- (*n* = 493), (b) N_2O (*n* = 1,136), (c) NO (*n* = 256), and (d) NH_3 (*n* = 565) losses as functions of N inputs. The curves and confidence intervals are taken from the statistical results from Equation (1) and are plotted at the mean study duration and sample depth (for NO_3^-). These plots have restricted axes scaled down from the full dataset to make it easier to see the majority of the data. Consequently, datapoints with large N inputs and outputs are not shown here; the full datasets are shown in Figure S1. "Season" is defined as the full study period for each observation, which typically spans a full growing season or year

TABLE 2 Coefficient estimates for the fixed effects for the linear mixed effects model described in Equation (1) to test for differences between the temperate and tropical datasets

	NO_3^{-g}		N_2O^g		NO^g		NH_3^g	
Fixed effects	Estimate (*SE*)	*t*-value	Estimate (*SE*)	*t*-value	Estimate (*SE*)	*t*-value	Estimate (*SE*)	*t*-value
β_0 (intercept)	**1.25 (0.08)***	16.34	−0.06 (0.05)	−1.06	**−0.76 (0.09)***	**−8.17**	**0.83 (0.09)***	**9.4**
β_1 (N input)[a]	**0.53 (0.04)***	12.08	**0.54 (0.04)***	13.52	**0.42 (0.08)***	**5.27**	**0.66 (0.13)***	**5.22**
β_2 (duration)[a]	**0.27 (0.04)***	6.49	**0.44 (0.04)***	9.84	**0.86 (0.11)***	**8.12**	−0.20 (0.17)	−0.10
β_3 (tropical)[b]	−0.03 (0.13)	−0.25	−0.03 (0.08)	−0.31	**0.95 (0.20)***	**4.72**	0.07 (0.15)	0.47
β_4 (tropical × N input[a])[c]	**−0.21 (0.10)***	−2.09	−0.15 (0.08)	−1.89	0.18 (0.33)	0.56	0.05 (0.22)	0.24
β_5 (sample depth)[d]	0.01 (0.11)	0.11	—	—	—	—	—	—
Conditional r^2 (marginal r^2)[e]	0.62 (0.20)	—	0.66 (0.19)	—	0.68 (0.35)	—	0.43 (0.11)	—
Unfertilized tropical baseline[f]	11.0		0.7		0.8		3.6	
Unfertilized temperate baseline[f]	9.0		0.5		0.1		3.3	

Note: Significant coefficient estimates are indicated by **bold** and $p < .001$***, $p < .05$*.

[a]Driver variables were standardized (by subtracting mean and dividing by two standard deviations) for analysis.

[b]Coefficient estimate for the change in the loss rate at mean N inputs from temperate (where β_3 is multiplied by 0) to tropical sites (where β_3 is multiplied by 1).

[c]Coefficient estimate for how loss rates change with N inputs in tropical sites (tropical = 1) compared to temperate sites (tropical = 0), that is, the tropical x N input interaction.

[d]Coefficient estimate for soil water collection sample depth (cm) for nitrate only.

[e]Marginal r^2 is the variance explained by fixed factors; conditional r^2 is the variance explained by fixed and random factors together.

[f]See the methods Equation (2) for the calculation of unfertilized baseline values, which are given here in kg N ha^{-1} $season^{-1}$.

[g]The response variables are unitless because they were log-transformed, but were in units of kg N ha^{-1} $season^{-1}$ before transformation.

2011) for NO_3^-, N_2O, NO, or NH_3; β_0 is the fixed-effect intercept (the coefficient for the cereal crop type at the mean values of N inputs, soil texture, precipitation inputs, and sample depth; with no organic or N fixation inputs, and with no split application); γ_i is the random variation in the intercept for each site *i*, using site latitude as a unique identifier; β_1 is the coefficient of *Nin*, N inputs (originally in kg N ha^{-1} $year^{-1}$ but standardized as described above); β_2 is the coefficient for *st*, soil texture (% sand near the surface, standardized; there were not enough data to include this covariate in the NO model); β_3 is the coefficient for *p*, actual precipitation + irrigation inputs (originally in mm/study duration, but standardized) for NO_3^- leaching or mean annual precipitation (originally in mm/year, but standardized) for NO, N_2O, and NH_3 models; β_4 is the coefficient for *i*, irrigation (1 = irrigated, 0 = no irrigation)—except for NO_3^- since actual irrigation inputs are captured by β_3 and NO where irrigation was dropped due to a VIF > 3; β_5 is the coefficient for *o*, organic inputs or N fixing crop presence (1 = some organic inputs or an N-fixing crop, 0 = none); β_6 is the coefficient for *sa*, split N application (1 = N application frequency > 1, 0 = single N application); β_7 is the coefficient for pH (soil pH); β_{8j} is the coefficient for c_j, crop type group *j* within each dataset as categorical dummy variables; β_9 is the coefficient for sdep, the sample depth of the lysimeter (cm), standardized, which was only included for NO_3^- leaching; and ε is unexplained residual variation.

The coefficients for the fixed effects in all models can be interpreted as the expected proportional change in each N loss, comparing units that differ by one standard deviation of the predictors with the other predictors held constant at their average values (Gelman, 2008). The relative importance of explanatory variables can be directly compared in units of standard deviations (Gelman & Hill, 2007); however, it is important to note that the relative importance is only defined in terms of the variation from these observational data. Variance explained was assessed using marginal and conditional r^2 values (Nakagawa & Schielzeth, 2013) calculated with the r.squaredGLMM function in the *MuMIn* package (Barton, 2018).

3 | RESULTS

3.1 | N losses in tropical and temperate regions

Unfertilized baseline NO_3^- losses in the tropics were slightly but not significantly higher than baselines losses in temperate areas (Table 2, Figure 2). However, tropical NO_3^- leaching responses to N inputs were slightly, but significantly, lower than temperate NO_3^-

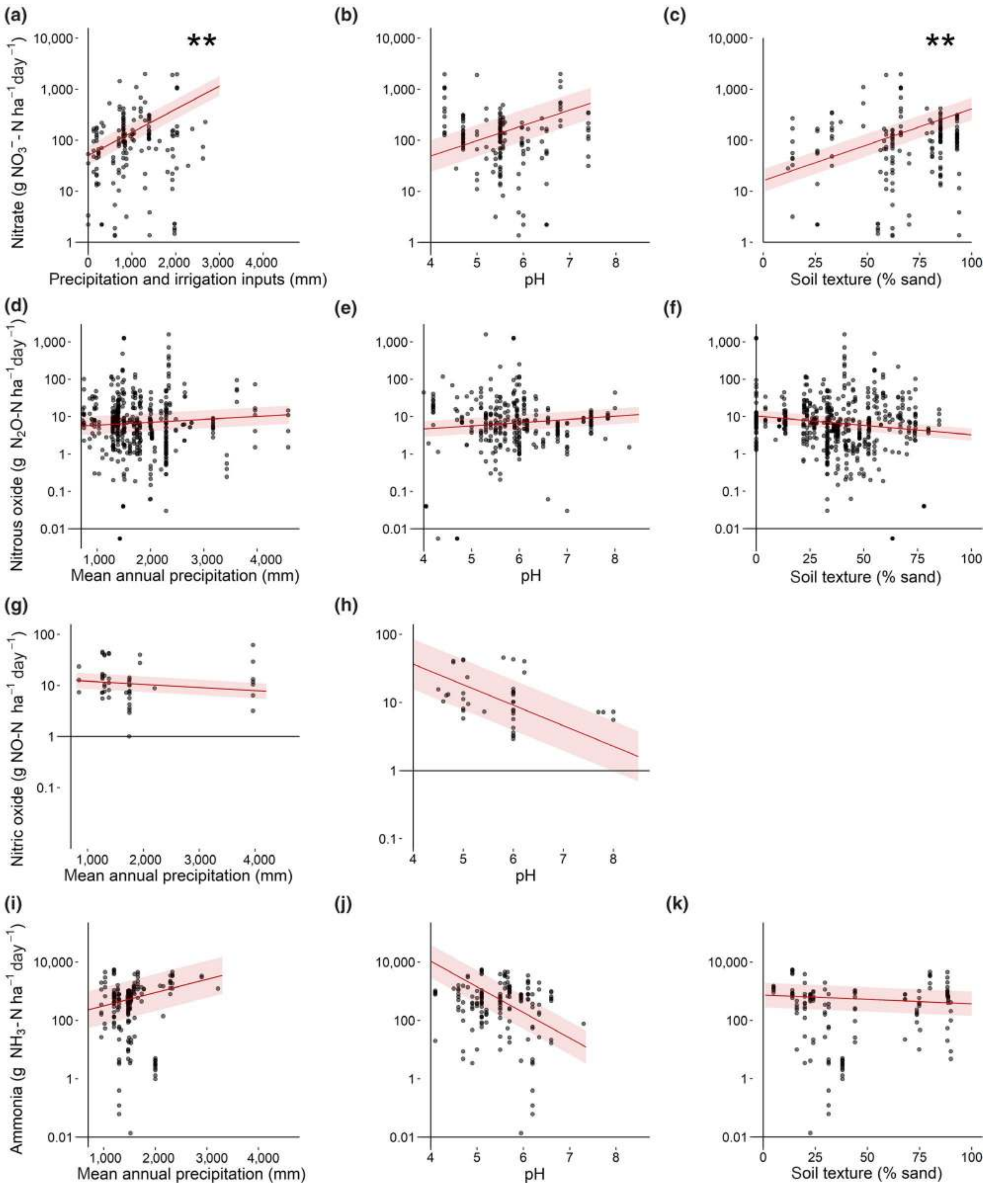

FIGURE 3 Relationships between environmental drivers and N losses from tropical croplands. Specifically, N losses are plotted against hydrological inputs (precipitation and irrigation for nitrate, mean annual precipitation for the others), soil pH, and soil texture (% sand). The fits are results from Equation (3), plotted at the mean of all other continuous variables and a value of zero for binary and categorical variables. The shaded regions are standard errors. Panels a (n = 170), d (n = 503), g (n = 49), and i (n = 193) show losses of NO_3^-, N_2O, NO, and NH_3, respectively, against hydrological inputs. Panels b (n = 166), e (n = 329), h (n = 40), and j (n = 167) show losses of NO_3^-, N_2O, NO, and NH_3, respectively, against soil pH. Panels c (n = 169), f (n = 593), and k (n = 142) show NO_3^-, N_2O, and NH_3 losses against sandy soil texture. All response variables are plotted on logarithmic scales with untransformed values. Significant relationships are indicated by $p < .01$**

leaching responses to N inputs over the fertilization range studied (Figure 2a, β_4 in Table 2). The combination of a higher baseline and a lower slope, along with the high variance in the data, meant that NO_3^- leaching losses in temperate versus tropical agricultural systems were statistically indistinguishable across a wide range of N inputs (see overlapping confidence intervals in Figure 2a). For N_2O (Figure 2b) and NH_3 emissions (Figure 2d), the temperate and tropical unfertilized baseline emissions (Table 2) and responses to N inputs

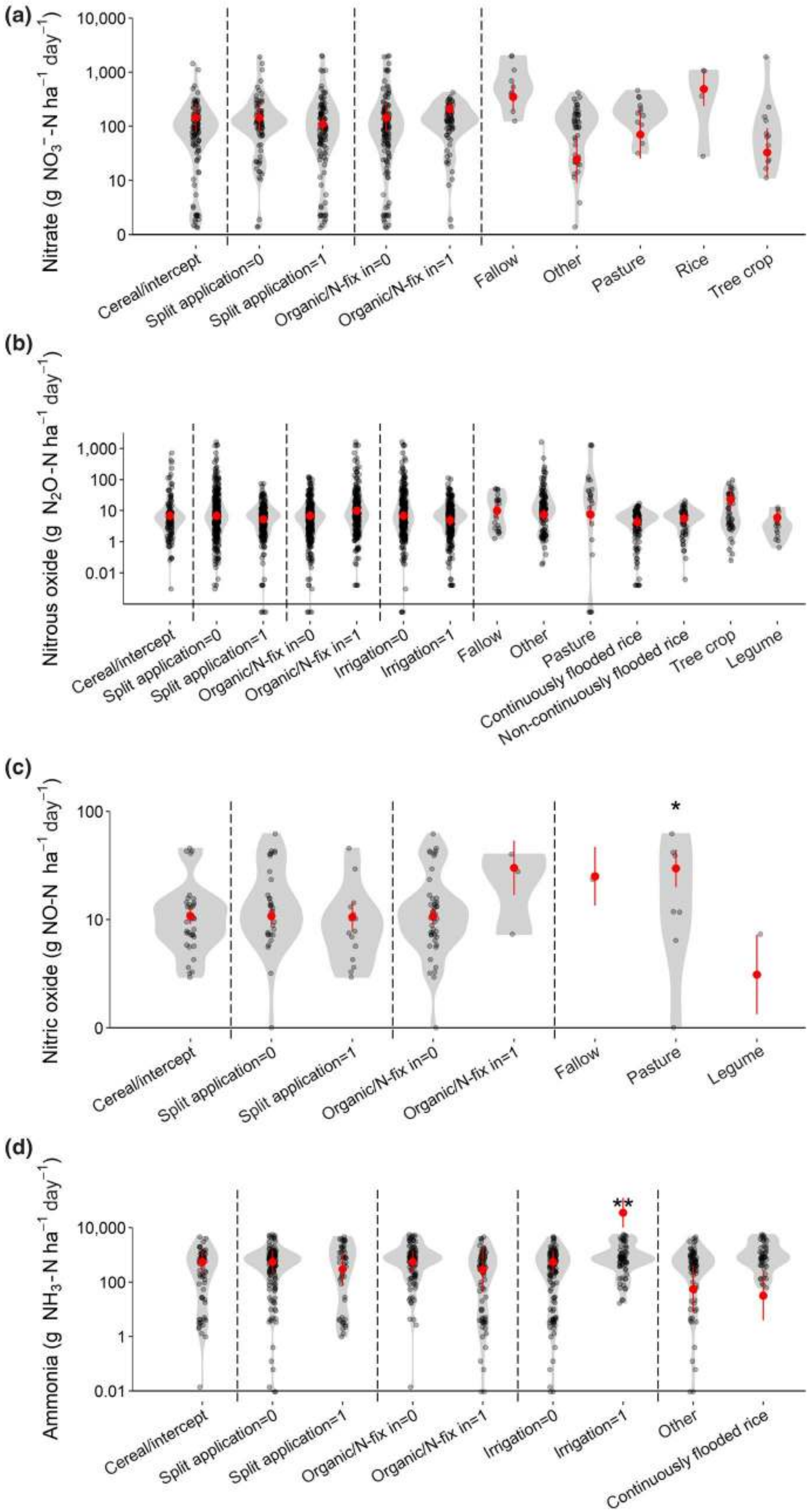

FIGURE 4 Relationships between management drivers and N losses from tropical croplands. (a) NO_3^- (n = 193), (b) N_2O (n = 597), (c) NO (n = 49), and (d) NH_3 (n = 193). N loss data (black) and coefficient estimates and standard errors from Equation (3) (red) are plotted. Each dataset is categorized by whether the N application was split, by whether there were organic or N-fixation inputs, and by crop type. The black dots indicate individual observations. For NO_3^- (a) and N_2O (b), there are enough overlapping data that, even with horizontal jittering, it is difficult to determine the relative data density. The horizontal thickness of the gray background indicates the frequency of observations at each level on the y-axis. Specifically, the left-hand fit for each grouping is the model's "baseline" case: A cereal crop without split application or organic or N fixation inputs, for the average water inputs and the average % sand. The other fits in each grouping are the offsets from the baseline case for each management driver. Because the fits are plotted for a single offset from the baseline case, they do not necessarily line up with the data when there is covariation between drivers. For example, the fit between irrigation and NH_3 is above the bulk of the data because those sites comprise mostly other and continuously flooded rice crop types, both of which have a large negative effect size. Significant relationships other than the intercepts are indicated by $p < .01$**, $p < .05$* (Table S2). All response variables are plotted on logarithmic scales with untransformed values

(β_4 in Table 2) were similar. For NO emissions, unfertilized baseline emissions were greater in the tropics (Table 2), but NO emissions increased with N inputs at a similar proportional rate in tropical and temperate zones (β_4 in Table 2).

Leaching losses of NO_3^- and NH_3 volatilization were generally an order of magnitude larger than trace gas losses in tropical croplands (Figure 2). All forms of N loss increased with N inputs (Figure 2, Table 2), though N inputs alone explained only a small amount of the variation in N losses (Figure 2). The fixed effects in our first model—N fertilization rate, study duration, location (tropical or temperate), and sample depth for NO_3^-—explained 20% of NO_3^-, 19% of N_2O, 35% of NO, and 11% of NH_3 loss variation (Table 2). The random effect of study location along with these fixed effects explained 62% of NO_3^-, 66% of N_2O, 68% of NO, and 43% of NH_3 loss variation (Table 2).

3.2 | Environmental and management controls on tropical N losses

Different environmental and management factors controlled gaseous compared with hydrological N losses from tropical agriculture. For NO_3^- leaching, soil texture (the percentage of sand; $p < .01$) was the strongest predictor, followed by a combination of precipitation and irrigation ($p < .01$) and N inputs ($p < .05$; Figure 3; Table S2). N input rate was the only significant predictor for N_2O losses ($p < .001$) and was a significant predictor for NO ($p < .01$) and NH_3 ($p < .05$; Table S2). There were some large effect sizes for different crop types, though only the pasture studies for NO were statistically significant (Table S2; Figure 4). Irrigation was a significant predictor of NH_3 losses with a large effect size ($p < .01$; Table S2; Figure 4). None of the other environmental or management factors helped to explain a significant amount of the variation in N losses (Figures 3 and 4; Table S2). Overall, the models including environmental and management factors explained 36% of NO_3^-, 13% of N_2O, 42% of NO, and 31% of NH_3 losses (marginal r^2; Table S2). The conditional r^2 values (0.78 for NO_3^-, 0.70 for N_2O, 0.64 for NO, and 0.59 for NH_3; Table S2) indicated that site differences that were unaccounted for by our fixed effects explained a large fraction of the variance in N losses.

4 | DISCUSSION

4.1 | N losses have similar variation and responses to N inputs across regions

N loss responses to N inputs were similar in both tropical and temperate regions over a common range of N inputs (Figure 2; Table 2). However, overall NO losses were higher in tropical regions due to higher unfertilized baseline losses (Table 2). Although there was substantial variation in each type of N loss for a given level of N input (widespread of data in Figure 2), it is illuminating to study how the mean N losses increased across a range of N inputs. Current and expected future fertilization rates across different regions of the tropics vary widely. For example, most sub-Sahara African countries fall short of the 50 kg fertilizer ha^{-1} $year^{-1}$ goal set in the Abuja declaration of 2006 (Wanzala, 2011), whereas our data demonstrate that some parts of the tropics use much more. For the purposes of presentation, we considered the effects of a tripling of N inputs from 50 to 150 kg N ha^{-1} $year^{-1}$. Based on the data we analyzed, a tripling of N inputs from 50 to 150 kg N ha^{-1} $year^{-1}$ in tropical croplands led to a 30% increase in mean NO_3^- leaching losses (12.52–16.30 kg NO_3^--N ha^{-1} $year^{-1}$; Figure 2a), a 30% increase in mean N_2O losses (0.82–1.07 kg N_2O-N ha^{-1} $year^{-1}$; Figure 2b), a 66% increase in NO losses (1.00–1.66 kg NO-N ha^{-1} $year^{-1}$; Figure 2c), and a 74% increase in NH_3 losses (4.76 to 8.29 kg NH_3-N ha^{-1} $year^{-1}$; Figure 2d).

The large variation in N losses across sites experiencing similar N inputs was striking (Figure 2). There was especially large uncertainty in the tropical NO losses (Figure 2c) due to low sample size, but similar variation between tropical and temperate estimates in the other N losses. The uncertainty in the N losses and N inputs relationship led us to investigate other drivers of N losses. Much of this variation might stem from differences in soil type, precipitation, and other environmental factors. For example, tropical soil orders such as Oxisols and some Ultisols, which make up approximately 30% of our data (Table 1), can have net anion exchange capacity that decreases NO_3^- leaching (Wong, Hughes, & Rowell, 1990). A study of intensive crop agriculture on Brazilian Oxisol soils found that despite inputs of >200 kg N ha^{-1} $year^{-1}$ from N-fixation on soybean fields (Figueira, Davidson, Nagy, Riskin, & Martinelli, 2016), NO_3^- export in soybean watersheds was small (0.4 kg NO_3^--N ha^{-1} $year^{-1}$) (Riskin et al., 2017). Other studies on Oxisols in tropical East Africa and Brazil observed N_2O emission factors (N_2O emissions as fractions of N inputs) that were an order of magnitude lower than IPCC estimates (−0.11 to 0.26% compared with 1%; Hickman, Palm, Mutuo, Melillo, & Tang, 2014; Jankowski et al., 2018). By comparison, mean NO_3^- losses from temperate sites across various crop systems and soil types that likely lack net anion exchange capacity were two orders of magnitude greater (44 kg N ha^{-1} study $duration^{-1}$) than the Brazil soybean measurements (Zhou & Butterbach-Bahl, 2014). Ammonia losses not only tend to increase with temperature but also vary widely depending on soil or floodwater pH, soil cation exchange capacity, soil moisture, and wind speed (Bouwman et al., 2002; Freney, Simpson, & Denmead, 1983). Other environmental factors like climate and soil texture can affect proximate controls on N_2O and NO fluxes such as soil N availability, soil water content, oxygen availability, and temperature (Davidson, Keller, Erickson, Verchot, & Veldkamp, 2000; Pilegaard, 2013), potentially leading to differences in gaseous fluxes between temperate and tropical regions. For example, the positive exponential response of NO emissions to soil temperature and different soil moisture dynamics (Davidson & Kingerlee, 1997; Stehfest & Bouwman, 2006) might help explain the larger baseline NO emissions in tropical croplands (Figure 2c).

4.2 | Environmental and agricultural management controls on N losses

Environmental site characteristics were important in controlling hydrological but not gaseous N losses. Soil texture (% sand) and a combination of precipitation and irrigation were important controls on NO_3^- leaching (Figure 3; Table S2). Together, soil texture and water inputs influence drainage, an important control on leaching (Di & Cameron, 2002). Irrigation also had a big effect on NH_3 losses; most of the observations with irrigation were from continuously flooded rice, which had above average NH_3 losses. Manure is another important source of NH_3 losses globally, but probably because our search terms focused on fertilizer, manure did not appear in our dataset. Surprisingly, we found a weak negative relationship between soil pH and NH_3 losses (Figure 3j); perhaps the range of soil pH measured in these studies was too narrow to see the expected positive effect. The complex dynamics of soil texture and soil moisture on gaseous N losses are likely very important controls on N_2O and NO losses at shorter timescales (Davidson et al., 2000), but we did not observe clear patterns between gaseous N losses and these site environmental characteristics at this scale.

We were surprised that our meta-analysis did not detect more significant effects of management practices such as crop type, use of organic fertilizer, or split N application, given that other studies have found these management practices to be important controls on N losses. There are some interactions between these management practices and N inputs that our models did not capture. A meta-analyses of ^{15}N tracer studies in temperate grain systems found that agricultural practices that couple carbon and N inputs to the soil via organic amendments and diversified crop rotations significantly increased N retention compared to reducing N input rates (Gardner & Drinkwater, 2009). Reduced and delayed N application in Mexican wheat agroecosystems reduced NO and N_2O emissions while maintaining yield (Matson, Naylor, & Ortiz-Monasterio, 1998). Our samples sizes were likely insufficient to detect differences for NO, given the high variability, but sample size was likely not an issue for NO_3^-, N_2O, or NH_3 (Table S3; Table 1). Other environmental factors such as soil N mineralization, which were not reported frequently enough for us to analyze, may be driving much of the unexplained variation in N losses.

4.3 | Opportunities for future improvement

Several issues inherent to meta-analysis (Gurevitch & Hedges, 1999) potentially influenced our interpretations. First, there was disproportionate representation of certain crop types or regions, such as the high proportion of NO_3^- leaching observations from sub-Saharan Africa, or few NO_3^- leaching or NO observations from flooded rice (Table S1). Second, there were relatively low samples sizes for some analyses, for example, only 11 papers for NO, though those papers covered 23 different site-years. Third, standardization and replication of N loss measurements would help constrain our estimates. As in previous meta-analyses (Abdalla et al., 2019; Zhou & Butterbach-Bahl, 2014), methods to quantify NO_3^- leaching varied and relied on different tools. Although we excluded very different methods such as estimating NO_3^- losses from changes in soil NO_3^- availability at different points of the season, there was still substantial variation in sampling techniques across studies in our analyses. Soil water was sometimes extracted by tension, or collected from free-draining lysimeters or drainage pipes, and there was a wide range of different water balance models used for estimating drainage. It is also worth noting that we lacked evidence on the degree to which NO_3^- leached below the rooting zone, as measured by the studies in our analysis, was leached out of the watershed. Fourth, because there is less infrastructure for agricultural experimentation in the tropics, there is less long-term, continuous monitoring, and studies are sometimes conducted in locations with unknown history of recent cultivation. Because of potential rapid N transformations such as a flush of N mineralization from tillage and wetting from the Birch effect (Birch, 1958), newly cultivated sites might exhibit different responses than the previously cultivated sites that are better studied. Overall, predictions of N losses could be improved with more experimentation of N losses in understudied tropical sites with well-controlled and replicated designs, multiple levels of fertilization, and measurements of yield, environmental, and management factors.

4.4 | Implications of our findings

Proponents of sustainable intensification typically suggest reducing yield gaps by N fertilization (Foley et al., 2011; Godfray et al., 2010; Mueller et al., 2012) of nutrient-limited tropical croplands (Conant et al., 2013; Liu et al., 2010; Mueller et al., 2014; Vitousek et al., 2009). Increases in fertilization are greatly needed in nutrient-limited agricultural systems to improve food security. Fertilization increases are especially needed in regions such as sub-Saharan Africa where, as of 2008, 25 countries used less than 20 kg N ha^{-1} $year^{-1}$ and only five countries approached or exceeded the Abuja target of 50 kg fertilizer ha^{-1} $year^{-1}$ (Wanzala, 2011). Increases of N inputs in areas with low current N inputs to reach the levels of the Abuja target will likely improve food security without significant negative consequences of N losses. For example, one study estimated that an increase of N inputs across sub-Saharan Africa to 150 kg N/ha would not significantly impact air quality (via ozone production from higher NO emissions; Huang, Hickman, & Wu, 2018). However, despite the large uncertainties in the relationship between N losses and N inputs, our results suggest that raising N inputs from 50 to 150 kg N ha^{-1} $year^{-1}$ for a broad range of tropical croplands will have important environmental consequences. Increased N fertilization is likely to increase NO_3^- leaching and potentially cause local and regional water pollution comparable to that caused by similar fertilization rates in temperate regions (Carpenter et al., 1998; Galloway et al., 2003; Liu et al., 2010). Higher N fertilization rates would also increase emissions of N_2O that contribute to global radiative forcing and NH_3 which causes air and

N pollution. NO losses, which cause air pollution, will likely be even greater in tropical than temperate croplands (Figure 2c).

The large variability in N losses indicates that there may be room for limiting additional reactive N losses from intensification by identifying other key drivers of N losses and focusing intensification on croplands where N losses are lower. Our finding that NO_3^- losses were higher in sandier and wetter cropping systems indicated that environmental differences between sites are important. Individual cases such as the low N_2O emissions from African and Oxisols (Hickman et al., 2014; Jankowski et al., 2018) and low NO_3^- leaching from some Oxisols (Riskin et al., 2017) demonstrate that environmental differences between different tropical regions and cropping systems affect N losses. Because of higher but more uncertain estimates in the tropics, we need further research on the potential consequences and rates of NO emissions from intensified tropical croplands. Data limitations at this large scale prevented us from analyzing additional environmental factors, such as soil nutrient status, which may be a key control on N losses. We need more systematic data collection on environmental and management factors in important cropping systems and regions and a better understanding of their effect on tropical N losses to improve N loss predictions. Combining better N loss predictions with N fertilization and yield data would also improve our understanding of the trade-offs between N fertilizer-induced yield increases and unwanted reactive N losses, helping us to prioritize specific areas for intensification.

Although estimates of the fate of N applied to tropical croplands can be improved, our findings have clear implications for current management. Given that tropical agroecosystems as a whole exhibit similar or higher N losses than their temperate counterparts, regional fertilization strategies in tropical agroecosystems should assume that N fertilization will have similar, or in the case of NO, greater negative environmental consequences than in temperate areas.

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AUTHORS CONTRIBUTIONS

A.H., G.G., D.M., K.T., C.P., C.N., and J.H designed the study. A.H., K.T., and C.C. collected the data. A.H. conducted the data analyses. A.H., G.G., D.M., C.N., and C.P. interpreted data. A.H., G.G., D.M., K.T., C.P., C.N., and J.H. wrote the paper.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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