

Review

The global potential for mitigating nitrous oxide emissions from croplands

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SUMMARY

Agricultural activities contribute almost half of the total anthropogenic nitrous oxide (N₂O) emissions, but proper assessment of mitigation measures is hampered by large uncertainties during the quantification of cropland N₂O emissions and mitigation potentials. This review summarizes the up-to-date datasets and approaches to provide spatially explicit and crop-specific assessment of the global mitigation potentials. Here, we show that global cropland N₂O emissions have quadrupled to 1.2 Tg N₂O-N year⁻¹ over 1961–2020. The mitigation potential is 0.7 Tg N₂O-N without compromising the crop production, with 86% from optimizing nitrogen fertilization, three-quarters (78%) from maize (22%), vegetables, and fruits (16%), other crops (15%), wheat (13%), and rice (12%), and over 80% from South Asia, China, the European Union, other American countries, the United States, and Southeast Asia. More accurate estimation of cropland N₂O mitigation potentials requires extending the N₂O observation network, improving modeling capacity, quantifying the feasibility of mitigation measures, and seeking additional mitigation measures.

INTRODUCTION

Nitrous oxide (N₂O) is a long-lived stratospheric ozone-depleting substance and greenhouse gas, which has a 100-year global warming potential 273 times higher than that of carbon dioxide.¹ The concentration of atmospheric N₂O has increased by more than 20% from 270 parts per billion (ppb) in 1750 to 331 ppb in 2018.^{2,3} Cropland is the largest contributor of anthropogenic N₂O emissions, accounting for approximately one-third of total anthropogenic N₂O emissions.² To sustain an increasing global population and the demand for food, N₂O emissions are projected to increase by 35%–60% between 2005 and 2030, largely driven by excessive use of synthetic nitrogen (N) fertilizers and manures to croplands.^{4–6} Reducing cropland N₂O emissions while maintaining crop production is thus conducive to achieving low levels of climate warming and preventing stratospheric ozone depletion. It is prerequisite to have a comprehensive understanding of cropland N₂O production mechanisms and an accurate assessment of cropland N₂O emissions.^{7,8}

Cropland N₂O emissions is a net result of N₂O production, reduction, transformation, and diffusion through the soil layers to the atmosphere,⁹ with each process controlled by various

abiotic and biotic factors. Microbial metabolic pathways account for approximately 70% of global N₂O emissions, including microbial nitrification and denitrification.¹⁰ Key drivers of N₂O emissions influencing these processes include soil properties, climate conditions, agricultural management practices, and microbial communities.^{10–13} A fair amount of research has explored such key drivers of each specific process under various specific conditions primarily based on field experiments or laboratory incubations. However, the relative importance of each process to N₂O production under different environmental conditions remains largely unknown, which is a barrier for accurate estimation of cropland N₂O emissions.

Significant efforts have been made to quantify cropland N₂O emissions from the field to regional and global scales, albeit large uncertainties still exist.^{14,15} Uncertainties from direct measurements lie in a deficit of coverage for the developing countries, limited sampling frequency, replication, and lack of detailed records of site information (e.g., local microscale biophysical characteristics and management history).^{7,16} Large discrepancies also exist among cropland N₂O emission estimates derived from different approaches (e.g., statistical upscaling models, process-based models, and atmospheric inversion



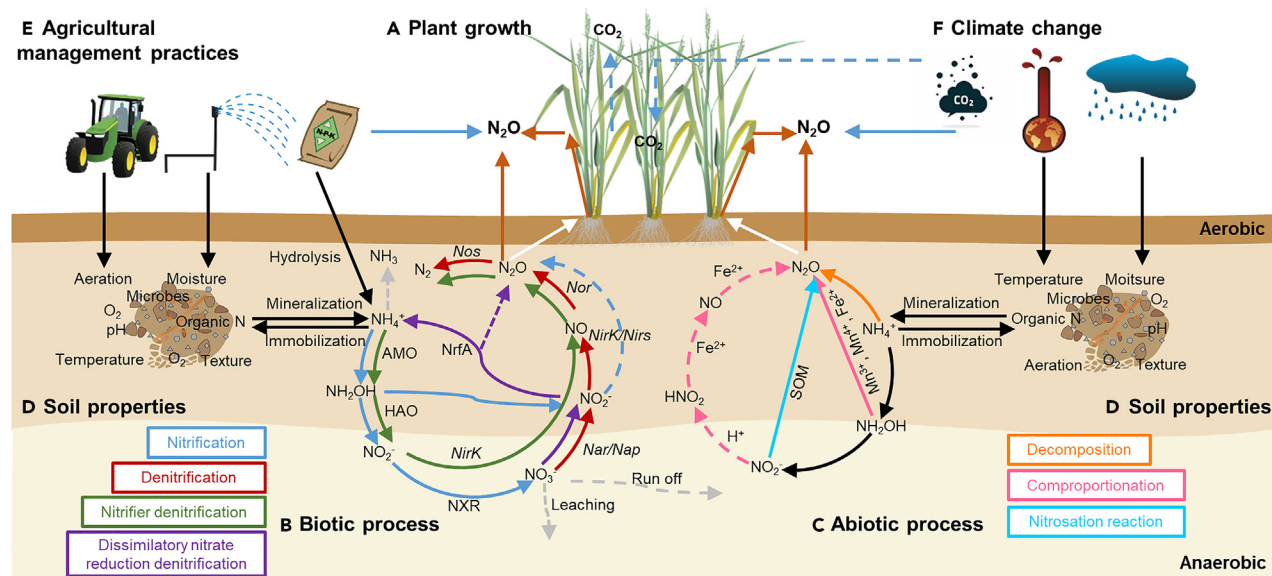


Figure 1. Conceptual framework for processes and key drivers regulating N_2O production from cropland

(A) Plant growth through photosynthesis, productivity, biomass allocation, and litter and root exudates impact on soil properties and pathways for N_2O production.

(B) Main biotic processes through microbial pathways for N_2O production, reduction, transformation, and diffusion from cropland.

(C) Main abiotic processes through chemical pathways for N_2O production and diffusion from cropland.

(D) Soil properties control on biotic and abiotic pathways for N_2O production.

(E) Climate change impacts on cropland N_2O production through plant growth, soil properties, and microbial and chemical pathways.

(F) Agricultural management practices impact cropland N_2O production through plant growth, soil properties, and microbial and chemical pathways.

models^{4,17,18}). These discrepancies are primarily attributable to high spatiotemporal variability and the complex mechanisms of N_2O production and consumption controlled by multiple biotic and abiotic factors, while it is under debate whether observed variation does not imply uncertainty but merely reflects the multitude of situations occurring in practice in large sets of measurements.¹⁹ Knowledge gaps may, however, also derive from combined impacts from multiple drivers across different cropping systems and predominant underlying mechanisms.¹⁴ Taken together, these limitations pose challenges to the accurate quantification of cropland N_2O emissions and, thereby, the assessment of mitigation potentials.

Besides the efforts to improve cropland N_2O emission estimates, effective mitigation measures and their potentials have also been increasingly investigated, based on field experiments and regional models.^{5,20–23} A series of knowledge-based N management practices have been proposed to mitigate cropland N_2O emissions, including the optimization of fertilization (i.e., rate, type, timing, placement), and the adoption of more efficient irrigation technologies.^{5,23–25} A possible shift in human diets aiming to decrease meat consumption could also play a role, since it would change feed production and associated emissions.^{26,27} However, most previous studies have focused on the individual mitigation options, specific crops, or local regions. Such assessments of mitigation potential have shown large uncertainties in the magnitude and even direction, depending on crop types, environmental factors, and management-related properties.⁵

As shown above, a comprehensive understanding of the achievable mitigation potentials from each of the aforementioned options, and their combinations, across global croplands

and its spatial pattern is currently lacking. To address this issue, we collated existing emissions estimates and derived spatially explicit and crop-specific N_2O emission maps for the past decades from recent scientific literature on this topic. We then re-estimated results based on a newly developed linear mixed-effects (LME) model and a set of high-resolution management datasets (e.g., fertilization rate, types, timing, placement).⁷ We also used this approach to assess achievable mitigation potentials from optimizing N fertilization, improving irrigation practices, shifting human diets, and their combinations by integrating mitigation potentials of these measures derived from meta-analyses, N fertilization within planetary boundaries, food security, and planetary health diet into N_2O estimates. We addressed three questions. (1) How do key biotic and abiotic factors regulate N_2O production in croplands? (2) What are the spatiotemporal patterns of crop-specific N_2O emissions? (3) How much and where could mitigation be achieved while maintaining crop yields? This review advances our understanding of global cropland N_2O emissions and mitigation opportunities and provides references for future research and policy priorities for cropland N_2O mitigation.

MECHANISM AND DRIVERS OF CROPLAND N_2O EMISSIONS

Cropland N_2O is produced from a range of abiotic and biotic processes (Figure 1). Microbial metabolic pathways account for approximately 70% of global N_2O emissions,²⁸ which mainly include nitrification, denitrification, nitrifier denitrification, dissimilatory nitrate reduction to ammonium (DNRA),

nitrification-coupled denitrification, complete ammonia oxidation (Comammox), and fungal denitrification.^{10,29} Nitrification involves ammonia-oxidizing bacteria (AOB)- and ammonia-oxidizing archaea-mediated ammonia oxidation (conversion of NH_3 to NO_2^-) and nitrite oxidation bacteria-mediated nitrite oxidation (NO_2^- to NO_3^-).^{30,31} The main products of nitrification process are NO_2^- and NO_3^- , accompanied by smaller amounts of N_2O and NO . Denitrification involves several stepwise reduction reactions (the reduction of NO_3^- to NO_2^- , NO , N_2O , and N_2) of enzymatic pathways,³² mediated by several species of denitrifiers (e.g., *Paracoccus denitrificans*, *Pseudomonas* sp.,^{33,34} and *Thiobacillus denitrificans*³⁵), dissimilatory nitrate reductase (*Nar* and *Nap*), dissimilatory nitrite reductase (*Nir*), nitric oxide reductase (*Nor*), and nitrous oxide reductase (*Nos*), with the *narG* and *napA*, *nirK/S*, *norB*, and *nosZ* genes encoding correspondingly.^{36–38} Nitrifier denitrification is the process whereby ammonia is first oxidized to NO_2^- , which is further reduced to NO and N_2O by AOB (specific nitrite reductase *NIR*) as the electron acceptor for denitrification under varying degrees of microaerophily.^{39–41} DNRA is a pathway of internal N cycling, referring to the process of NO_3^- reduction to NO_2^- and NH_4^+ mediated by cytochrome nitrite reductase (*NrfA*), accompanied by the transient accumulation of NO_2^- and production of N_2O under obligate anaerobic or facultative anaerobic conditions.^{42–44}

Apart from the aforementioned pathways, the remaining microbial processes and the abiotic process of chemodenitrification, decomposition, and nitrification reaction also play important roles in cropland N_2O emissions.^{45–47} It is essential to quantify the relative contribution of different processes to the production of N_2O , whether for the accurate simulations or abatement of N_2O emissions. However, with the development of molecular and stable isotope methods,^{48,49} accounting for the relative contributions is still challenging, since multiple pathways for N_2O production and reduction are simultaneously activated in different microenvironments even in the same soil, as discussed above.¹¹

Cropland N_2O emissions are a net result of N_2O production, reduction, transformation, and diffusion through the soil layers to the atmosphere.⁹ Factors impacting N_2O emissions through regulating the aforementioned processes are generally categorized into four groups (Figure 1): soil properties, climatic variables, agricultural management practices, and biological properties.^{10–13} Edaphic factors mainly refer to soil physicochemical conditions that control N_2O emissions through mediating N and carbon substrates, metabolic energy sources, aeration conditions, and enzymatic activity, including soil-available N and carbon, moisture, oxygen, texture, pH, and soil temperature.^{10,11,13,50–62} Climatic variables (including the change of precipitation and temperature, elevated atmospheric CO_2 concentrations, and the increase in atmospheric ozone concentration and atmospheric N deposition) can affect cropland N_2O emissions not only directly by changing soil moisture and temperature regimes but also indirectly via crop and soil interactions.^{63–72} Agricultural management practices, such as fertilization, irrigation and drainage, and tillage, play an important role in cropland N_2O emissions through shaping spatiotemporal variability in soil processes.^{25,73–85} Cropland N_2O emissions mainly occur through nitrification and denitrification driven by microorganisms. Therefore, the population abundance, structure, and

activity of related microorganisms in soil have important effects on the emission of N_2O .^{86,87}

The processes for cropland N_2O production are generally complex and diverse, being affected by various drivers (Figure 1). As a consequence, quantitative conclusions about the main process for cropland N_2O emissions and corresponding contribution under combined influences of different climate, soil, and agricultural managements are challenging. The relationships between N_2O emissions from cropland and each environmental factor are usually investigated by laboratory experiments and simple correlation analysis without considering the influence of other factors. Besides, the effects of climate change, soil properties, and agricultural management on soil N_2O emissions may interact with each other; for example, the effect of climate change and soil properties on soil N_2O emissions may be obscured by the agricultural management. The incomplete understanding of the processes and mechanisms of N_2O production leads to large uncertainties in field observations and modeling. Therefore, for the purpose of accurate modeling and effective abatement, a multi-gradient and multi-factor network of field experiments is urgently needed to clarify the mechanisms of climate, soil, and agricultural management practices on N_2O emissions from cropland.

CURRENT N_2O EMISSIONS BY CROP AND REGION

A key starting point for assessing the mitigation potential and prioritizing mitigation measures from cropland is understanding the baseline levels of cropland N_2O emissions for which measurements of N_2O fluxes across diverse soil-crop-climate systems are fundamental. Through combining up-to-date N_2O -emission observation datasets from online data repositories and peer-reviewed meta-analyses, this review formed an extensive global observation dataset from 515 sites spanning 37 countries covering the period from 1978 to 2018 (Figure S1 and Table S1). Although most observations still focused on main grain crops (e.g., wheat, maize, rice) from large-emission regions (e.g., China, United States [US], Europe), increasing attention on cash crops (e.g., orchard, vegetables, tea) and from previously under-represented regions (e.g., Oceania, Sub-Saharan Africa) were noted.

To estimate cropland N_2O emissions at regional or global scales, considerable efforts have been taken in recent decades and have formed two broad types of approach, namely, top-down and bottom-up approaches.² Top-down approaches refer to estimating N_2O emissions using atmospheric inversion models by integrating measured N_2O concentrations from tall tower or aircraft flask sampling as constraints and often use prior information of bottom-up estimated emissions. Bottom-up approaches include emission inventories based on Intergovernmental Panel on Climate Change (IPCC) tier 1 or tier 2, spatial extrapolation of field flux measurements by statistical models, and processes-based modeling constrained by ground-based observations (IPCC tier 3) (Table S2).¹⁴

To obtain comprehensive insights on total quantities and temporal-spatial patterns of current cropland N_2O emissions, this review synthesized previous estimates and additional conducted estimates using the model of Shcherbak et al.⁷⁶ and crop-specific emission factor (EF) models of Cui et al.⁷ (experimental

Table 1. Summary of approaches for estimating N₂O emissions

Approach		Methodology	EF	N input (types)	N input (crop-specific)	Emissions (Tg N ₂ O-N)	Sources
Bottom-up	Inventory	FAO ⁹⁴	constant	S, M, C	crop uniform	1.14 ± 0.36 (1961–2020)	F
		EDGAR ⁹⁵	constant	S, M, C	crop-specific; 25 crops	1.25 ± 0.29 (1961–2020)	F, B
		GAINS ⁹⁶	constant	S, M, C	crop-uniform	1.53 ± 0.13 (1961–2014)	F
	Statistical models	L-N-RR ⁹⁰	dynamic (Nrate)	S, M	crop-specific; 171 crops	0.77 (2000)	F, B
		NL-N-RR ⁹⁰	dynamic (Nrate)	S, M	crop-specific; 171 crops	0.66 (0.56–0.78) (2000)	F, B
		REML ¹⁵	dynamic (C, S, M)	S, M	crop-specific; 3 crops	2.692 (2000)	F, B
		SRNM ^{18,88}	dynamic (C, S, M)	S, M, C	paddy rice; upland crops	0.82 ± 0.34 (1961–2014)	F
		Random forest ⁹⁷	dynamic (C, S, M)	S, M	crop-specific; 26 crops	1.39 (2014)	F
		Based on Shcherbak et al. ⁷⁶	dynamic (Nrate)	S, M, C	crop-specific; 26 crops	0.91 ± 0.36 (1961–2020)	F
		LME ⁷	dynamic (C, S, M)	S, M, C	crop-specific; 26 crops	0.74 ± 0.26 (1961–2020)	F
	Process-based models	NMIP ¹⁴	dynamic (C, S, M)	S, M, D	crop-uniform	3.3 ± 1.1 (2007–2016)	F, B, I
		DLEM ⁴	dynamic (C, S, M)	S, D, F	crop-specific; 10 crops	2.0 ± 0.1 (2000–2014)	F, B, I
		DNDC ^{98,99}	dynamic (C, S, M)	S, M, C, D, F	crop-specific (corn and wheat)	3.6 (2.96–4.35) (1996–2013)	F, B, I
		DAYCENT	dynamic (C, S, M)	S, M, C	crop-specific (corn, wheat, soybean)	1.77 (1991–2000)	F, B,
Top-down		Davidson ⁹²		S, M	crop-uniform	2.2 (1.5–2.4) (2005)	F, B, I, O
		Thompson et al. ⁹¹		S, M, D, F	crop-uniform	2.3% ± 0.6% (EF, 2013)	F, B, I, O

For emission factors (EF): S, soil; C, climate; M, management; Nrate, only N application rate was considered.

For N input type: S, synthetic; M, manure; C, crop residue; D, deposition; F, N fixation.

For sources: F, fertilizer-induced (corresponding to N inputs including S, M, C, and F); B, background emissions; I, indirect; O, other anthropogenic sources.

procedures, Table 1, and Figure 2). Global cropland N₂O emissions by these approaches ranged from 0.7 to 3.6 Tg N year⁻¹ (Table 1).^{14,15,18,88–90} The variations in these estimates were mainly attributed to differences in terms of emission sources (fertilizer-induced, background, indirect, and other anthropogenic sources), modeling structure (linear or non-linear, fixed or random intercept and slope), influencing factors considered (climatic-, soil-, and management-related factors), and N input (types and crop-specific disaggregation) (Table 1). For example, estimates from top-down approaches are higher than those from bottom-up by a factor of 2 for the US Corn Belt, mainly because top-down approaches additionally account for other direct emissions (e.g., from fossil fuels, industry, biomass burning, waste, and waste water) and indirect emissions (e.g., from atmospheric deposition, leaching to aquatic systems where emissions occur, human sewage, and unmanaged soils).^{17,91,92} The difficulty for

top-down approaches to distinguish emission sources results in bottom-up approaches being more usually used for estimating cropland N₂O emissions. Spatial discrepancies between different bottom-up approaches were reported in India, south-eastern US, Brazil, Europe, and South and East Asia, owing to differences in EFs or N inputs used.^{4,18,93}

This review primarily focuses on fertilized-induced (including synthetic fertilizer, manure, and crop residue) cropland N₂O emissions because fertilizer application is the primary target of mitigation measures; therefore, emissions from fertilized pastures or soil mineralization processes (i.e., background) were removed for comparison (Figure 2). Global fertilizer-induced cropland N₂O emissions showed an overall increasing rate of 15–22 Gg N₂O-N year⁻² (1.6%–2.6%) over the past six decades, while the increase leveled off from 27 Gg N year⁻² before 1984 to 18 Gg N year⁻² after 1984 (Figure 2). Estimates

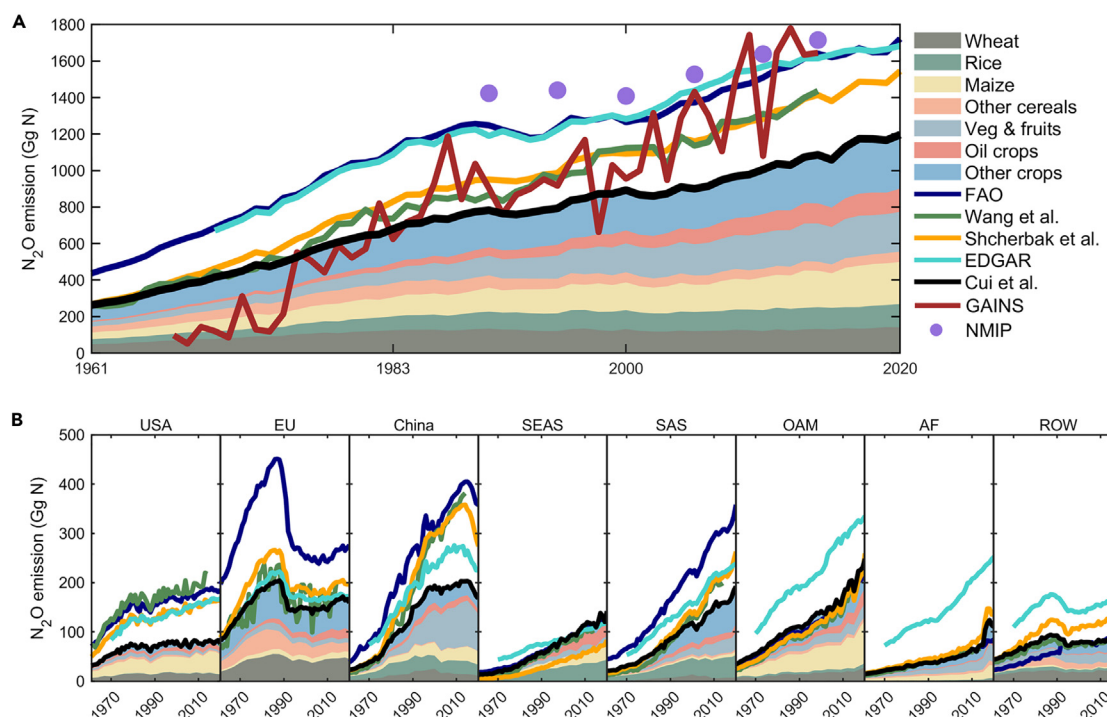


Figure 2. Global and regional estimates of fertilizer-induced N₂O emissions from cropland from 1961 to 2020

(A) Global fertilizer-induced N₂O emissions from cropland.

(B) Regional fertilizer-induced N₂O emissions from cropland. Examples of estimations of N₂O emissions are shown here. We normalized the FAOSTAT³⁴ and GAINS⁹⁶ by removing the contribution from synthetic fertilizers applied to pasture, the EDGAR⁹⁵ version 6.0 by excluding the contributions from synthetic fertilizers applied to pasture and soil mineralization, and the NMIP¹⁴ by excluding the contribution from “background” emissions. We also estimated emissions from 1961 to 2020 using models from Cui et al.⁷ and Shcherbak et al.⁷⁶ The study regions are USA, European Union (EU), China, Southeast Asia (SEAS), South Asia (SAS), other America (OAM), Africa (AF), and rest of world (ROW).

of process-based models (e.g., N₂O Model Intercomparison Project [NMIP]) showed larger interannual variation compared to that of the other bottom-up approaches due to accounting for influence of climate variability (Figure 2).⁴ The temporal evolution of cropland N₂O emissions differed among regions. Cropland N₂O emissions have become relatively stable in the US since the 1980s, while they began to decrease in Europe.¹⁴ The growth in cropland N₂O emissions in China began to slow down around the 2000s,^{22,89} while maintaining rapid growth in South and Southeast Asia.² As a result, emission hotspots shifted from Europe (111.3 Gg N₂O-N year⁻¹) and US (43.5) in the 1960s to China (198.6 Gg N₂O-N year⁻¹), South Asia (171.3) and Southeast Asia (127.6) in the 2010s. This result was highlighted by both statistical models and process-based modeling.^{4,18} The temporal dynamics of cropland N₂O emissions were primarily driven by changes in N fertilizer input rates but also for diverse reasons associated with agronomic management, socioeconomic development, and policy enforcement (Figure 2).^{100–108} Process models also highlighted the contributions from manure use and atmospheric N deposition, especially for East Asia and India.^{98,109}

From a crop-specific perspective, previous estimates were conducted in a fixed year (e.g., 2000 from Gerber et al.⁹⁰), for limited crops (e.g., maize and wheat from Tesfaye et al.⁹³) or regions (e.g., China from Ma et al.¹¹⁰ and US from Lu et al.¹¹¹). In other words, there is no reconciled framework of the global

N₂O emission dynamics by crop. Therefore, this review specifically estimated N₂O emissions for 26 crops from 1961 to 2020. This was done by combining the crop-specific EF models of Cui et al.⁷ and the newly crop-specific N fertilization dataset at 5-arcminute spatial resolution from 1961 to 2020¹¹² (experimental procedures and Figure 2). This EF model was selected to account for impacts from various climatic, edaphic, and management-related factors while avoiding assumptions associated with the representation of complex N₂O production processes as in process-based models.

Overall, different estimates all highlighted the largest contributions of wheat and maize (12%–22% and 17%–19%),^{7,90} primarily from East and South Asia and Europe.⁹³ Vegetables and fruits currently account for only 9% of global cropland N₂O emissions but are the largest contributor of the increased rate of global cropland N₂O emissions over 1961–2020 (21%), especially in developing regions (e.g., China). Crop-specific contributions varied among regions. The overall increase of cropland N₂O emission in the US, Central America, Brazil, and South Africa are mainly contributed by the emissions from maize (62%, 48%, 48%, and 43%, respectively), while the emissions in China, Central Asia, and the Middle East are mainly contributed by vegetables and fruits (50%, 38%, and 36%, respectively). The dominant contribution of corn-soybean cropping systems for increasing N₂O emissions in the US was also confirmed by high-resolution estimates using Dynamic Land Ecosystem

Model (DLEM) process models.¹¹³ In addition, recent less rapidly increasing cropland N₂O emissions in China were contributed by wheat, rice, legumes, and other crops with reduced N fertilizer input.²² The decrease in N₂O emissions in Europe since the 1990s was primarily contributed by wheat and barley as a result of reduced N input. The insights regarding spatial hotspots and crop-specific contributions of N₂O emissions are essential for prioritizing mitigation measures.

MITIGATION POTENTIALS

N₂O emissions from croplands could be mitigated by improving management practices during crop production, such as optimizing N fertilization and improving irrigation practices or adjusting the consumption of food products. In the sections below, we review the mitigation strategies and provide updated estimates for the mitigation potentials of N₂O emissions by crop types and regions.

Potentials from optimizing N fertilization

Soil-available N, as a substrate for N₂O production, mainly comes from fertilizer input.¹² However, improper N fertilization may cause environmental and food security issues.^{114,115} It has been proved that a substantial reduction of N₂O emissions can be achieved through the optimization of N fertilization, such as the “4R nutrient stewardship” (i.e., right application rate, right fertilizer type, right application placement, and right application time).^{7,22,93,116} These measures mainly aim to provide better synchronization between crop N demand and N supply and, therefore, simultaneously increase N-use efficiency and decrease N₂O emission or other N losses (e.g., NH₃, N runoff, and NO₃[−] leaching).²³ Moreover, these measures would decrease N₂O emissions and N₂O EFs through several inter-related biogeochemical mechanisms, including changing soil properties, microbial activities, physical adsorption and diffusion, and physiological and ecological characteristics of plants (Figure 1). Mitigation effects from these measures have been investigated by a number of experimental studies and meta-analyses in which substantial variations were noticed in magnitude and directions depending on multiple factors (e.g., environmental conditions and crop types).^{23–25,116–122}

Fertilizer N application rate is not only an important predictor of cropland N₂O emissions but also an unambiguous proxy for N₂O reductions.¹²³ Generally, the enabling potentials to optimize N inputs can be indirectly expressed by N surplus (defined as the sum of N inputs [fertilizer, manure, biologically fixed N, and N deposition] minus N outputs).^{114,124} N surplus value is particularly useful for researchers, policymakers, and international organizations, since it could be used for capping the N inputs to the realistic benchmarks or a proposed planetary boundary.¹²⁵ Previous studies have shown that the global N surplus (total N input minus N uptake by crops) would need to be reduced to 40–52 Tg N year^{−1} in order to remain within a safe boundary without breaching the bounds for acceptable air and water quality (Table S3).^{7,8,114,126,127} However, reducing N inputs to lower the N surplus within a proposed planetary boundary is one-sided, and crop yield should be maintained at the same time.⁷ Based on the regional N boundaries for arable land to close yield gaps in regions where environmental thresholds for eutrophica-

tion of terrestrial and aquatic ecosystems and nitrate in groundwater are not exceeded, the newly updated global cropland N surplus was estimated at 102.0 Tg N year^{−1} in 2020 and would need to decrease to 46.5 Tg N year^{−1} (Figure 3A). Reduction of N surplus would be mainly contributed by rice, wheat, maize, and vegetables and fruits (with a reduction of 11.6, 10.4, 9.0, and 8.9 Tg N year^{−1}, respectively), accounting for 71% of total N surplus (Figure 3A), which is 10% lower than that when only a planetary boundary was considered.¹¹⁴

Mitigation potentials of global cropland N₂O emissions from the reduction of excess N inputs varied largely with different bottom-up approach estimates, ranging from 30% to 51%.^{7,93,127} For example, the mitigation potential of N₂O emissions from global cropland was estimated to be 423 Gg N year^{−1} for 17 main crops using the IPCC tier-1 approach with a reduction of 51 Tg N year^{−1} excess N input,¹²⁷ while only a 304 Gg N year^{−1} reduction in N₂O emissions was reported using the non-linear response of EF to N input approach with a reduction of 34 Tg N year^{−1}.⁷ It is worth mentioning that for the same reduction of N input at different baseline applications, constant EFs overestimated the reduction of N₂O where N was applied at low N application rate, while it underestimated the reduction of N₂O where N was overapplied.⁷⁶ Considering the non-linear relationship between EF and N input, as well as the goals of remaining within the N threshold of the planetary boundary while not compromising yields in the estimate mentioned above,⁸ global mitigation potential of N₂O emissions from reducing fertilizer N inputs was re-estimated as 409.7 Gg N year^{−1}, equivalent to a 33.4% reduction in total global N₂O emissions (Figure 3B). Fruits and vegetables contributed 89.0 Gg N year^{−1} reduction and accounted for about 21.7% of the overall mitigation potentials, followed by maize (72.5 Gg N year^{−1}, 17.7%), rice (55.6 Gg N year^{−1}, 13.6%), and wheat (46.3 Gg N year^{−1}, 11.3%) (Figure 3B), an order different from the reduction of N input (ΔN_{surplus}) to the respective crops. Approximate mitigation potentials from maize (i.e., 70 Gg N year^{−1}) and higher mitigation from wheat (i.e., 85 Gg N year^{−1}) were reported by Tesfaye et al.,⁹³ equivalent to 36% and 35% reduction of total global N₂O production of maize and wheat, due to a higher method of N excess estimation (75% reduction of N excess) and different EF models (aggregated EF models including CCAFS-MOT, IPCC tier 1, IPCC tier 2, and Tropical N₂O).⁹³

Spatially, regardless of the approaches for modeling EFs or assessing N surplus, mitigation priorities generally are in the regions where crops produce high emissions or where N is applied in excess, such as China, India, and Southeast Asia, accounting for more than half of global N reduction (Figure 3A).^{7,8,114} In the updated assessment of N₂O emissions, South Asia, China, and Southeast Asia had the largest mitigation potentials, totaling 229.8 Gg N year^{−1} and accounting for 56.9% of total N₂O emissions (Figure 3B). Similarly, due to the large harvested area and relatively higher excess N, major maize-growing countries such as China, US, India, and Brazil and major wheat-growing countries such as China, India, US, and Russia are not only the regions of highest N₂O emission but also mitigation priorities.⁹³ However, the hotspots of N₂O mitigation do not overlap completely with the hotspots of excess N application, because the mitigation potentials may have a Pareto optimum between high-N-surplus and high-EF regions.⁷ For example, the priority

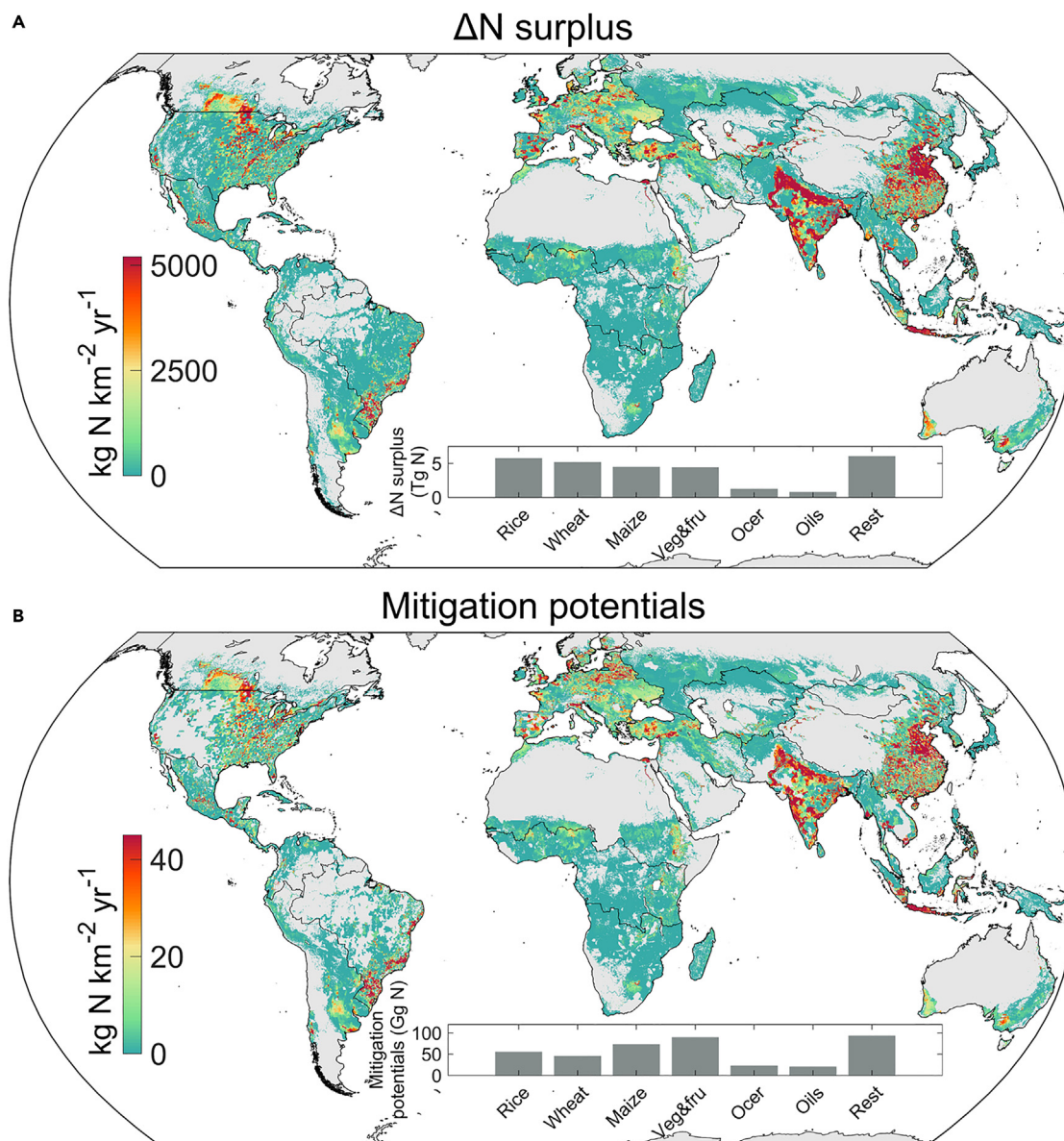


Figure 3. Reduction of N surplus and mitigation potentials of N₂O emissions for optimizing N fertilization

(A) Spatial pattern of reduction of N surplus (ΔN surplus [all crops]).

(B) Spatial pattern of mitigation potentials (all crops).

The bars show ΔN surplus and mitigation potentials for each crop.

areas for mitigation might be either underestimated in low-N-input regions (such as parts of eastern Europe) or overestimated in high-N-input regions (such as parts of the North China Plain).⁷ Therefore, apart from N reduction, the local environmental conditions incorporated into crop-specific EFs are critical for accurate estimates of N₂O mitigation.

Fertilizer types (here widely referring to synthetic chemical fertilizers, organic amendments, controlled-release fertilizers, and fertilizer inhibitors, as well as the combination of synthetic and organic fertilizers) can affect the cropland N₂O emissions due to difference in content of NH₄⁺, NO₃⁻, and organic carbon. In addition to the enhanced efficiency fertilizers (EEFs), the reduction effect of fertilizer types on N₂O emissions was

insignificant or inconsistent among current studies, depending on the soil, climate, cropping system, and agricultural management.^{23,116,117,128–131} EEFs, including controlled-release N fertilizers (CRFs), nitrification inhibitors (NIs), urease inhibitors (UIs), and double inhibitors (DIs), have been proven as the most efficient approach for reducing the conventional N application rate, while overall N₂O emissions were reduced by up to 24%–42%, as shown in previous meta-analyses (Figure 4A).^{23,116–121,132} Meanwhile, the use of EEFs could significantly enhance the crop yield regardless of crop type, although for DIs the change in wheat yield is not significant (Figure S2). The use of EEFs can enhance crop uptake and reduce the potential of nutrient losses to the environment

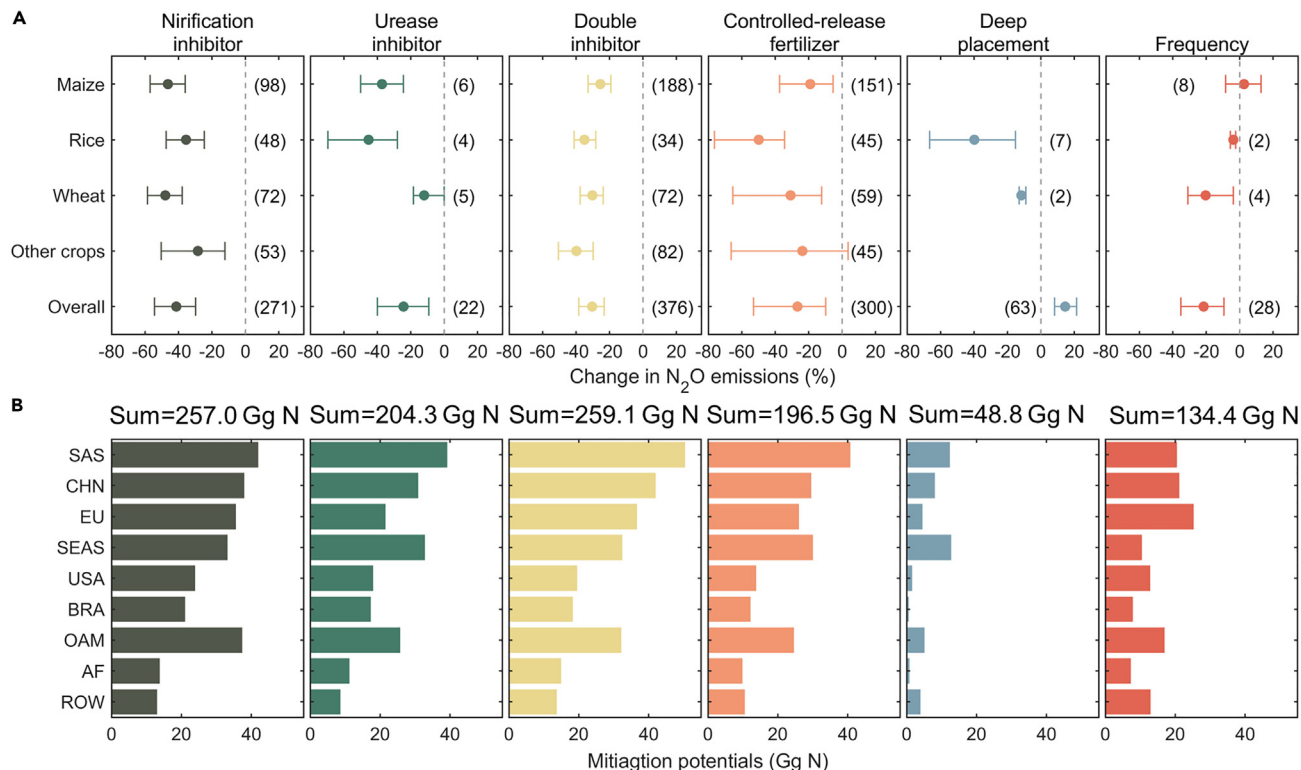


Figure 4. Mitigation potentials of additional management practices and technologies

(A) Aggregated effects of management practices and technologies on N_2O emissions from field and meta-analysis studies.^{23,116–122} Error bars represent 95% confidence intervals (CIs).

(B) Regional mitigation potentials while applying these technologies on croplands based on current N application rates and social-economic level. The potential implementation rates for different technologies were based on Gu et al.¹¹⁶ (see [experimental procedures](#)). The regions are South Asia (SAS), China (CHN), European Union (EU), Southeast Asia (SEAS), USA, Brazil (BRA), other America (OAM), Africa (AF), and rest of world (ROW).

mainly through delaying N transformation processes or slowing the release of N.¹³³ However, the efficacy of EEFs varies greatly with EEF types, cropping systems, biophysical conditions, and management conditions.

For example, the efficacy of EEFs on N_2O reduction varied among types, CRFs (19%) being less effective than NIs (38%) and DIs (30%).¹¹⁸ The efficacy of NIs also varied by NI formulation, i.e., 3,4-dimethylpyrazole phosphate (DMPP) is assumed to be more effective in inhibiting nitrification than dicyandiamide and nitrapyrin due to the similar mobility of DMPP with NH_4^+ in soil.^{118,134} Compared to paddy rice, the efficacies in upland crops were more complicated and generally less effective.²⁴ Upland crops are generally subject to variable biophysical conditions (i.e., soil temperature and soil moisture), which affect the N-release patterns from EEFs and therefore preclude synchronicity of N release and uptake.²⁴ Despite the overall lower efficacy, the effects of CRFs (−49.8%) on rice are larger than those of inhibitors (−35.0% to −45.2%) and less so for wheat and maize (Figure 4A). The soil biophysical conditions also control the effectiveness of EEFs; for example, DMPP is only effective in alkaline soils for the reduction of N_2O through inhibiting the growth of AOB.¹²⁵ Based on the adoption rate of EEFs related to N application rate proposed by Gu et al.,¹¹⁶ the mitigation potentials of N_2O emissions with the implementation of EEFs by crop and region were identified. The mitigation potential of

EEFs ranged from 197 to 259 Gg N year^{−1}, in descending order by DIs, NIs, UIs, and CRFs (Figure 4B). The reduction priorities of EEFs were mainly in South Asia, China, Southeast Asia, other America (OAM), and the European Union (EU), accounting for about three-quarters of the total mitigation potentials (Figure 4B).

Timing and placement of N fertilization are also important for achieving a significant containment of N_2O emissions without decreasing potential yield.⁵ For fertilization measures, the effect of deep fertilization on crop yield was better than that of increasing fertilization frequency, and paddy rice was better than upland crops (Figure S2). Appropriate timing of N fertilization ensured synchronization between N supply and crop demand, while the effect of fertilization timing on N_2O emissions was usually substituted with N fertilization frequency. The effects of optimizing N application methods and frequency (e.g., deep placement and increasing splitting frequency of N fertilizer) on N_2O emissions and N_2O EFs are contradictory among different studies (e.g., significant reduction, non-significant reduction, or increase) and vary across crop types (e.g., rice, wheat, and maize).^{23,116,122} For example, the effect of deep placement on N_2O mitigation is more significant for rice (−39.8% [95% confidence interval −64.5% to −13.0%]) than that for wheat (−11.7% [−14.4% to −10.3%]). By contrast, increasing splitting frequency is more effective for wheat (−20.4% [−36.9% to −9.8%]) compared to rice (−3.8% [−5.2% to −2.0%])

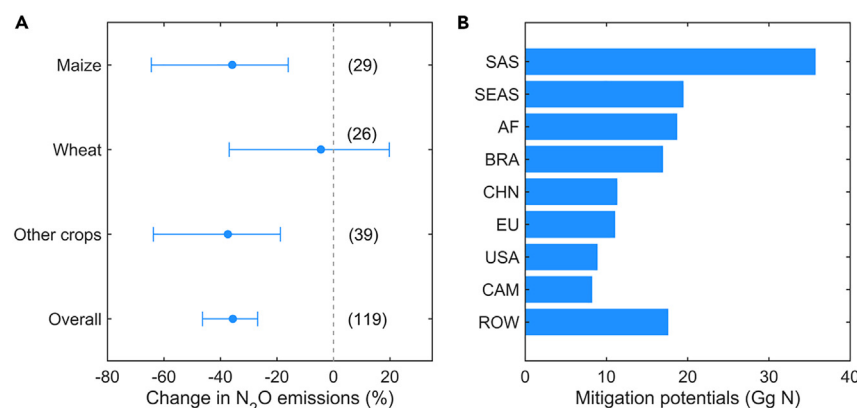


Figure 5. Mitigation potentials of improved irrigation practices

(A) Effects of improved irrigation on N₂O emissions.^{25,116,121} Error bars represent 95% confidence intervals (CIs).

(B) Regional mitigation potentials while applying improved irrigation practices based on irrigated area. The potential implementation rates for irrigation were based on Gu et al.¹¹⁶ (see [experimental procedures](#)). The regions are South Asia (SAS), Southeast Asia (SEAS), Brazil (BRA), China (CHN), European Union (EU), North Africa (NAF), USA, Central America (CAM), Equatorial Africa (EQUAF), and rest of world (ROW).

(Figure 4A). However, there are risks of increasing N₂O emissions through increasing splitting frequency of fertilizer N application for maize (2.6% [−7.5%–13.6%]) (Figure 4A). The substantial uncertainties are in close association with amount and types of fertilizer application.^{5,132} Deep placement of fertilizer, as compared to surface spreading or shallow placement, would decrease concentrations of NH₄⁺ and NO₃[−] in the surface soil (upland) or flooded water (paddy rice) and form a thicker soil layer before N₂O diffusion into air, which potentially facilitates more N₂O being reduced to N₂. Moreover, both deep placement and increasing splitting frequency could increase crop N uptake and partially reduce excessive inorganic N as substrate for microbial communities, thereby reducing losses in the form of N₂O.^{116,132} Following the implementation rate of “4R nutrient stewardship” as described by Gu et al.,¹¹⁶ the mitigation potential of N fertilization methods and frequency were 48.8 Gg N year^{−1} and 134.4 Gg N year^{−1}, mainly from EU and Asia, respectively (Figure 4B).

Potentials from improving irrigation practices

Several improved irrigation practices have been proposed that may influence soil processes, including alternative wetting and drying (AWD) for rice and drip irrigation and sprinkler irrigation mainly for upland crops. Although AWD shows potential to reduce methane emissions, frequent alternate aerobic and anaerobic conditions under AWD could prompt N₂O emissions.¹³⁵ Across the other practices, drip irrigation is the most promising practice to deliver N₂O mitigation benefits.^{25,136} Drip irrigation refers to delivering a low volume of water to the rooting zone of crops, resulting in different spatiotemporal distribution of water moisture compared to sprinkler, furrow, or flooding irrigation.^{5,25} This also has consequences for N₂O emissions. Due to more NH₄⁺ being concentrated in wet areas and NO₃[−] in dry areas, lower soil activity of N₂O-producing microbes, and better synchronicity between water delivery and crop N needs, drip irrigation favors N₂O reduction to N₂ and prevents the production of N₂O emissions.^{33,137–139}

Overall, improved irrigation (such as drip irrigation) is also an effective strategy for significantly reducing N₂O emissions by 25%–44% and enhancing crop yield by 7%–16% compared to furrow and sprinkler irrigation or flooding irrigation, depending on different crop types (Figures 5A and S3).^{25,116,121} The global drip-irrigated agricultural area in the year 2020 was 14.4 million

ha, which is widely spread in arid and sub-arid climate zones, accounting for 5% of the total irrigated area.¹⁴⁰ Therefore,

significant mitigation potentials might be achieved through the expanded adoption of drip irrigation. The magnitude and spatial pattern of mitigation potentials on N₂O emissions under drip irrigation was estimated using the adoption rate of improved irrigation provided by Gu et al.,¹¹⁶ which is closely related to the rate of N uptake. The mitigation potentials of improved irrigation (excluding rain-fed areas) were 147.7 Gg N year^{−1}, where hot-spots for mitigation are cropland in high-priority areas requiring drip irrigation, mainly in South Asia, Southeast Asia, Brazil, and China (Figure 5B). Since generally fewer N fertilizers are used in the drip-irrigation systems, the combined mitigation potential is possibly even larger than the current figures suggest.

Potentials from shifting human diets

Greenhouse gas (GHG) mitigation through sustainable dietary change has attracted growing research interest, especially since the 2010s, and has been proposed as an indispensable demand-side option to meet climate change targets.^{16,27,141,142} Sustainable dietary changes typically refer to a shift in human diet from animal-sourced and processed foods toward plant-sourced foods. Such a shift could deliver double dividends to mitigate GHG emissions from both direct and indirect (i.e., less land for crop cultivation of livestock feed) pathways domestically and from large exporters.¹⁴³ Direct mitigation of livestock-related N₂O emissions from shifting diet is beyond the scope of this review, although livestock production was reported as a major source of anthropogenic emissions.^{144–146}

The N₂O mitigation potential from dietary change is usually assessed together with that of CO₂ and CH₄ (combined as total GHG emissions or carbon footprints) and calculated as emission differences between baseline and diverse diet scenarios^{27,141,147} mainly embedded in the reduced harvested areas together with reduced fertilizer inputs.¹⁴⁷ However, noticeable differences of N₂O mitigation potentials from dietary change are found in related studies, mainly associated with modeling approaches to calculate N₂O emissions and design diet scenarios. Diet scenarios are generally classified into three types, i.e., reduced animal products (e.g., less meat or dairy, no ruminant or red meat, no dairy, meatless Monday, Mediterranean diet), modified vegetarian (e.g., vegetarian [lacto and/or ovo], vegan, pescatarian), and dietary recommendations (e.g., modeled based on recommendations from Food and Agriculture Organization [FAO], World Health Organization [WHO], national governments, or

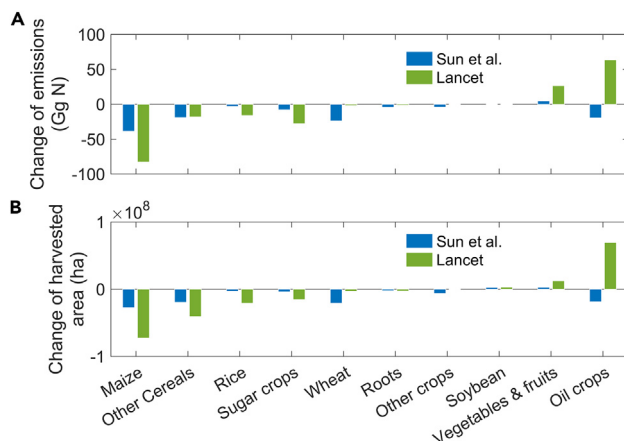


Figure 6. Change of harvested area and N₂O emissions of each crop group toward targeting the EAT-Lancet diet

Changes in global cropland N₂O emissions (A) and harvested area (B) due to shifting human diets. Blue bars indicate the spatially explicit change of harvested area provided by Sun et al.²⁶ and corresponding changes of cropland N₂O emissions in high-income nations; green bars indicate the global average change of harvested area by the EAT-Lancet Commission¹⁵² and corresponding changes of global cropland N₂O emissions.

expert organizations [e.g., American Heart Association]).¹⁴⁸ The former two scenarios primarily target environmental sustainability, forming simple scenarios of reduction in per-capita meat consumption or substitution of grains or low-N-footprint meat (e.g., pork, poultry, or fish) for high-N-footprint meat (e.g., beef), without considering dietary nutritional balance.^{149,150} With improved knowledge of the link between human health and environmental sustainability, “dietary recommendations” are well established at both national¹⁵¹ and international levels (e.g., WHO) proposed by the EAT-Lancet Commission as the “planetary health diet.”¹⁵²

The EAT-Lancet Commission¹⁵² and Sun et al.²⁶ assessed the potential of the EAT-Lancet diet for changing crop production and sparing cropland, respectively. Assuming that crop yield is constant, the relative changes in food production by the EAT-Lancet diet could be converted into changes in harvested areas. Such changes are provided as globally average results by the EAT-Lancet Commission, which were further spatially segmented by Sun et al.²⁶ Notably, the impacts of diet shifting on N₂O emission from pasture and other land was not considered for this review. Combining the two sets of projected changes in crop-specific harvested areas and the updated LME-estimated spatial-explicit EFs, the overall mitigation potentials were approximately 115 and 57 Gg N₂O-N year⁻¹ from Sun et al.²⁶ and the EAT-Lancet Commission, respectively (Figure 6A). Both results indicated maize (an important source of direct feed for livestock or as a component of forage for pigs, poultry, and cattle²²) as the largest contributors of reduction, accounting for 33% and 144% (mainly due to the increased intake of oil crops offsetting the mitigation by maize) of total mitigation potentials from dietary changes. However, the secondary contributors differed between the results derived from Sun et al.²⁶ and the EAT-Lancet Commission¹⁵² (wheat and sugar crops, respectively), possibly reflecting differences in crop-specific demand to achieve the EAT-Lancet diet (in other words, har-

vested area) between high-income and medium- or low-income countries (Figure 6B) as well as the emissions by crop (Figure 2). In addition, these mitigation potentials are difficult to directly compare with those of other related studies, which assessed mitigation potentials from all GHG emissions. Indirectly taking advantage of their results was also complicated, since more information is required about feed-to-food conversion efficiency of animal products and complex trade relationships between food producers and consumers. Notwithstanding, this assessment highlights mitigation hotspots by crops and regions and provides implications for mitigation priority.

Potentials from measure combinations

Although N₂O emissions from cropland are inevitable, a comprehensive assessment of combined measures on N₂O reduction is imperative for evaluating overall mitigation potential and identifying the priorities of measures implemented by region and by crop.

Different assessments have reported mitigation potentials as 22%–60% of total N₂O emissions, with the variations ascribed to the assessed emission systems (i.e., cropland, agricultural sources, and anthropogenic sources) and types, efficacy, and adoption rate of mitigation measures.^{96,147,153,154} For example, based on the assessment in this review, mitigation potentials over cropland N₂O emissions from optimization of N fertilization rate, irrigation, and diets were approximately 751.9 Gg N year⁻¹, accounting for 61.2% of direct emissions (Figure 7A). However, only 34% mitigation of cropland N₂O emissions was achieved from mitigation management practices reported by Gu et al.,¹¹⁶ as these measures were implemented to decrease losses of all N pollutants, which may increase the risk of N₂O emissions.¹¹⁶ In contrast, relatively higher mitigation potentials (accounting for 35%–60% of total N₂O emissions) from agricultural sources could be achieved from integrated measures of improved efficiency of crop and animal production and manure use, reducing food loss and waste and changing diets.¹⁵⁴ Approximately 22%–26% of N₂O emissions from global anthropogenic sources could be mitigated through improved efficiency measures, diet change, food waste, and loss reduction, with the largest mitigation potentials from agricultural soils.^{96,147,153} Generally, optimizing N fertilization was the most effective pathway for N₂O mitigation in these studies; however, this does not mean that other agronomic practices and demand-side adjustment are negligible.

Globally, the distribution of N₂O mitigation potentials was uneven (Figure 7B); South Asia, China, EU, OAM, US, and South-east Asia altogether contributed 81.1% of the total N₂O mitigation potentials (Figure 7C). The substantial regional differences were further identified regarding the most effective pathway for N₂O mitigation. In general, optimizing N fertilization contributed the largest proportion (70%–92%) of N₂O reduction for the aforementioned regions with the highest N₂O mitigation potentials; for the EU and US, changing human diets also significantly mitigated N₂O emissions (11.6% and 19.3%, respectively) (Figure 7C). Maize production contributed 22.3% of the overall mitigation potentials, followed by vegetables and fruits (16.2%), other crops (14.5%), wheat (13.0%), and rice (12.9%) (Figure 7D). Optimizing N fertilization is the greatest mitigation strategy among crops, while shifting diet and irrigation are also

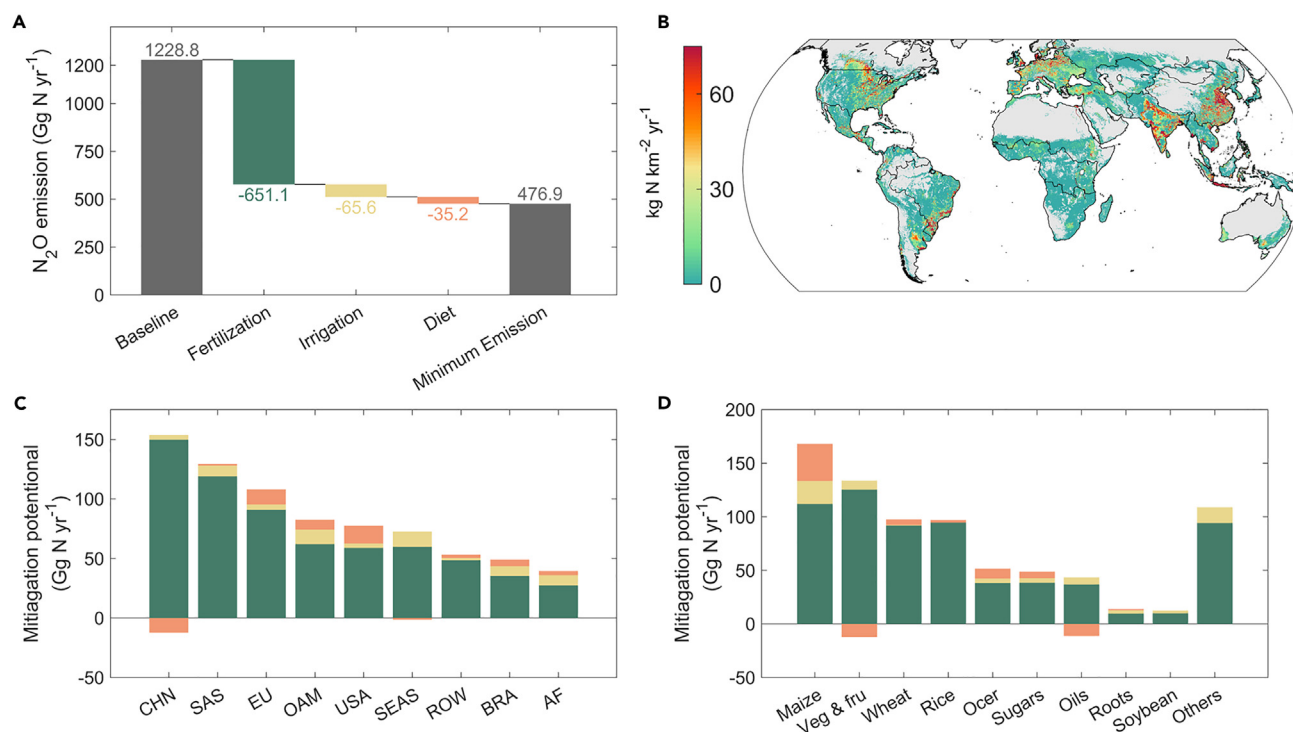


Figure 7. Mitigation potentials from combinations of all measures

(A) Waterfall plot shows current fertilizer-induced emissions (gray) and minimum emissions while optimizing N fertilization (Fertilization), applying improved irrigation practices (Irrigation) and shifting diet structure (Diet) in 2020.

(B) Spatial pattern of mitigation potentials.

(C) Regional mitigation potentials. The regions are South Asia (SAS), China (CHN), European Union (EU), Southeast Asia (SEAS), USA, Brazil (BRA), other America (OAM), Africa (AF), and rest of world (ROW).

(D) Mitigation potentials for each crop group (wheat, rice, maize, other cereals [OCE], vegetables and fruits [Veg & fru], oil crops [Oils], roots, sugar crops [Sugars], soybean, and other crops [Others]).

efficient in mitigation for maize and other crops. Since a diminishing marginal effect was detected when combining all the practices, practices should be prioritized with the highest effect size on N_2O reduction and regions or crops with the highest N_2O emissions or mitigation potentials. Note that the mitigation potentials were assessed in the ideal situation and not in actual production,²¹ which is highly dependent on the implementation of measures as well as the local soil and climate conditions. Nonetheless, the mitigation potentials reported in this review may be conservative due to the difficulties and uncertainties of integrating the multiple mitigation pathways (such as biochar,¹⁵⁵ food waste and loss,¹⁵⁴ and microbial consumption of N_2O with fungivorous mites¹⁵⁶).

RESEARCH AND POLICY PRIORITIES

Research priority 1: Extending N_2O observations and manipulation experiments

Limited availability of cropland N_2O observations representing diverse agroecological conditions is one of the major barriers of better understanding and accurate quantification of N_2O emissions. Despite considerable efforts by international or regional research programs (e.g., GHG-Europe, GRACEnet, TRAnet), data availability still limits accurate N_2O quantification in a number of ways, such as spatiotemporal and crop-specific

coverage, observations at microbial levels, combined effects of multiple factors, and detailed information from experiments.

First, most available N_2O observations come from Europe, US, and China but are scarce in some developing countries (such as Sub-Saharan Africa, typically underfertilized),^{7,132} and globally averaged EFs would likely overestimate N_2O emissions for such regions. Previous observations are also unevenly distributed across different crop types (primarily focused on staple crops), while recent experiments have demonstrated that some cash crops (such as vegetables and tea) are becoming increasingly important N_2O emitters.^{22,157–159} These cropping systems are reported to feature excessive use of fertilization, high N losses, and low N-use efficiency and are located in areas with high precipitation and temperature favoring anaerobic conditions. The overuse of fertilizers would further cause legacy N_2O fluxes and lead to bias in N_2O estimates derived from growing seasons. Nonetheless, more observations are needed from some main planting areas (e.g., tea plantations in Africa, South Asia, and Central Asia) and from the non-growing season.^{160,161} Meanwhile, to filter observation data and reduce the bias, more detailed records of the status of control sites should be reported, including the year when each control site was first fertilized before the experiment and its level of soil residual N.^{7,8} Therefore, extending the global coverage of direct and indirect N_2O flux measurements to encompass all major agricultural land-use

types and climates and land-use changes and management practices, and conducting long-term high-frequency monitoring, are particularly important to increase the reliability of N_2O emissions as well as upscale results from site to regional scales.^{2,16}

Second, available manipulation experiments have primarily examined the effect of single factors, not multiple factors, on N_2O emissions, posing critical challenges for estimating N_2O and future projections. Extensive manipulation experiments should be conducted to investigate the interaction effects of multiple factors on N_2O emissions in the future, with microbial data (including abundance, diversity, structure, and activity) simultaneously measured. The combined effects should involve interactions among multiple climate change scenarios (such as warming, elevated O_3 concentration, acid rain, and extreme events^{162,163}) and management practices (such as fertilization, irrigation, drainage water management, land use, and cover change^{164–168}). The impacts of these factors on the indirect N_2O emissions from cropland need to be better evaluated.

Lastly, although the central role of microbial regulation on N_2O emissions has been increasingly recognized, microbial data are generally absent from most field-based and laboratory experiments at this time. This limits projecting the response of microbes to environmental changes and involvement of microbial pathways in estimating N_2O emissions.¹¹ More efforts are required for long-term and high-frequency monitoring of critical N_2O -relevant biomarkers and other corresponding biophysical characteristics from both field studies and soil incubation, using new approaches (i.e., stable isotope methods) and biological technology. With more available microbial data, the spatiotemporal dynamics of soil microbes, their responses to environmental parameters, and their regulation of N_2O fluxes could be quantified, enabling breakthroughs in the quantification of main process contributions, driving factors, and precise simulation and abatement.¹¹

Research priority 2: Improving model representations and quality of input datasets

Multiple process-based biogeochemical models have been developed to simulate N_2O emissions, and large discrepancies are noticed among these models in terms of magnitude, spatial pattern, and temporal trend as well as the contributions of natural and anthropogenic factors.¹⁴ These uncertainties indicate that dominant processes of N_2O emissions, associated driving factors, and their quantitative correlations are under-represented in these models. For example, effects of agronomic management practices (such as irrigation, tillage, and crop rotation) and natural disturbances (such as freeze-thaw cycle or the wet-dry cycle) are currently simplified or missing from NMIP models.⁴ Moreover, model capabilities are also largely limited by the quality of input datasets, especially for N inputs and management practices datasets at finer scales by crop type.^{169,170} Therefore, to improve model reliabilities in estimating cropland N_2O emissions, the top priority is to improve the model representations and the quality of input datasets.

To improve the model representations and performance, comprehensive understanding about the key processes regulating N_2O emissions, associated driving factors, underlying mechanisms, and interaction with climate change should be improved and represented explicitly and transparently in model

structures.¹⁴ Accurate representation of key processes (such as nitrification and denitrification) and their responses to climate change and human activities are of particular importance.⁴

Joint community efforts and multiple means (e.g., investigation, remote sensing, large-scale machine learning, data-model integration) are also required to develop detailed management datasets at the global scale. Last but not least, model-model and model-data intercomparison would be necessary to address uncertainty and discrepancies among estimates.

Research priority 3: Quantifying spatial adaptability and effectiveness of mitigation measures as well as their applicability

According to manipulation experiments and meta-analyses, mitigation effects on some improved practices from different climate-soil-plant-agricultural systems are significantly different with respect to magnitude or even sign, suggesting different spatial suitability of these agronomic practices, which must be taken in accordance with the local conditions to maximize mitigation benefits.¹⁷¹ Key mechanisms and associated factors underlying the varying mitigation effects should be clarified with enough observations across large-scale environments (see [research priority 1](#)). Furthermore, the adoption of the advanced practices should be done with caution to avoid other unintended consequences (i.e., compromised yield, increased risk of pollution swapping among reactive N losses, indirect N_2O emission, and greenhouse gas emissions [such as N_2O and CO_2]).^{23,24,132,172–176}

To comprehensively assess the effectiveness of potential mitigation practices, other environmental and socioeconomic considerations should also be incorporated. First, economic profits and opportunity costs are generally the first consideration of farmers, which would largely determine the successful implementation of these new practices. Other socioeconomic and cultural considerations, including farm size, mechanization, intensification of management practices, and dietary needs and preferences, also need to be considered.²² For example, more comprehensive dietary shift scenarios, considering variations in different age and gender groups as well as regional food cultures and cuisines, could be established to support a more robust assessment of food-related mitigation potentials.¹⁵² Third, the mitigation practices may have negative secondary impacts on food security through influencing agricultural markets and international trade.^{177,178} Therefore, to identify spatially optimal strategies of mitigation practices, cost-benefit analyses incorporating the aforementioned considerations should be conducted for each individual practice and their combinations. As such, it is possible to answer where, to what extent, and in what order these practices should be adopted.

Research priority 4: Exploration of new mitigation opportunities

Beyond the aforementioned agricultural practices for N_2O mitigation, exploring new opportunities to achieve the win-win goals of food security and environmental sustainability has become a prominent global challenge in the coming decades.¹⁷⁹ Recent studies have proposed some new opportunities from technological and management aspects. Improvements of new genetic

techniques to breed or engineer crops with higher N-use efficiency provide the possibility to match crop N demand and soil-available N pools, and thus could maintain yields with reduced N inputs and stem the leak of N into the environment.^{10,11,180,181} Some new microbes with the capability of reducing N₂O have been discovered, such as the denitrifier phenotype of *Paracoccus denitrificans* in batch cultures and nosZ-containing *Bradyrhizobium japonicum*.^{182,183} Enhancing the gene expression of nosZ or activities of N₂O-reductase enzyme by manipulating the availability of required metal (such as Cu, Fe, or Mo) could also ultimately reduce N₂O emissions.^{11,184,185} The application of coconut husks, as a natural solution, could increase the abundance of mites and further promote consumption of fungal N₂O producers, while its universality needs to be further tested using different types of fertilizer and soil.¹⁵⁶ However, we also emphasize that the capacity and effects of these new approaches across various environmental conditions should be carefully examined and exemplified before wide adoption.

Policy priorities

Despite the breakthroughs in agronomic and biological techniques on N₂O mitigation, N₂O emissions from croplands cannot be reduced unless these techniques are widely adopted.¹⁸⁴ Therefore, the development of N₂O mitigation techniques requires effective two-way cooperation between researchers and farmers and needs to be accompanied with innovative policies that provide enabling socioeconomic conditions for wide adoption of these techniques.²²

Efforts to quantify the mitigation potentials from combinations of measures would allow for prioritizing mitigation measures by region and by crops, which are generally helpful for policymakers to formulate reasonable and grounded policies. Regardless of regions, optimizing N fertilization has proved to be the most effective mitigation measure, and policy interventions must be conducive to the implementation of measures and mitigation. For example, for the hotspots of cropland N₂O emissions and mitigation (e.g., China), the implementation of optimizing N application rate applied to cropland fueled by Chinese government policies (e.g., the nationwide Soil Testing and Formulation Fertilization Program and Zero Growth of Chemical Fertilizers and Pesticides) have decelerated and reduced the cropland N₂O emissions by one-third.²² Some successful experience from China (i.e., a campaign with integrated soil-crop system management recommendations and establishment of the Science and Technology Backyard [STB] platform) could also be used as a reference for the other countries, especially the developing countries where smallholder farming dominates agricultural production (e.g., India and the Southeast Asian countries). The STB platform has particular implications for countries where soil is deprived of nutrients (e.g., Africa), where additional fertilizer application for food production is needed but may exhibit significant growth of N₂O emissions. Besides the technology extension services, some national subsidy programs should also be established and incentives to farmers provided to gradually adopt these advanced practices in developing countries.²³

With regard to high-income countries (e.g., Europe), capital assets and well-functioning financial markets have enabled them to become more aware of the need to address N pollution

earlier and with better mechanisms than developing countries. Special environmental protection measures have also been implemented in the EU Nitrates Directive (91/676/EEC) in the 1980s, originally focusing on the optimization and reduction of fertilizer use as well as the protection of groundwater. Attributed to the implementation of the EU Nitrates Directive, N₂O emissions from European agricultural soils decreased by 21% between 1990 and 2010.² Incentives and regulations have been implemented by individual member countries to meet the targets of the directive. The implementation of the directive in Germany sets out the specific details of the German fertilizer regulations, indicating that the adoption of measures requires not only the incentives such as subsidies available for farmers as in developing countries but also adequate inspection mechanisms if the results of a measure extend beyond the immediate (economic) benefit of the farmers.¹⁸⁶

Beyond the technical realm, shifting human diet based on the EAT-Lancet diet is another important mitigation measure, especially in maize cultivation for developed countries in Europe and the US. Since the EAT-Lancet diet was first proposed as a scientific goal for healthy diets based on sustainable food production systems, the FAO and WHO have also developed guidelines for promoting sustainable and healthy diets. However, it must be acknowledged that implementing a uniform global dietary standard is relatively challenging, with limited guidance for different regions and cultures. Therefore, in the future, countries should formulate regional guidelines for sustainable healthy diets tailored to their residents' social, economic, and cultural contexts as well as other factors. For regions with special cultures and dietary habits, leveraging international trade and investment mechanisms to enhance residents' dietary health is nonetheless a good choice. International food trade can serve as a tool to reduce N₂O emissions by reallocating crop production to regions where EF or N₂O emissions are relatively lower and yields are maintained.²² However, such reallocation needs to be incentivized by policies (e.g., a tax on N fertilizers or on N₂O emissions¹¹⁶), and the related GHG emissions for transport and preservation need to be accounted for. In addition, through the popularization of "food education" guiding residents' food consumption toward a sustainable and healthy pattern will contribute to achieving the goals of a "sustainable diet."

At the national level, it is also critical to transit from the traditional production-focused N policies to new policies with the consideration of the environmental costs of N pollution. Agriculture as the traditional primary production sector has been the target of multiple policies, many of which incline toward improving production conditions without consideration of environmental costs due to the food security concerns. In fact, an analysis from a global legal database found that about two-thirds of N-related policies aimed to facilitate and incentivize N production and consumption.¹⁸⁷ This exemplifies the challenges encountered when implementing meaningful measures to reduce N₂O emissions. While there are various strategies to directly tackle N₂O emissions with policy instruments,¹⁸⁸ they rarely work in the reality of political arguments. Instead, in most cases, instruments merely follow the opportunities for improving the efficiency of N use.¹⁰² Such improvement without compromising crop yields may be promoted by economic considerations, with mineral fertilizer as a valuable resource and a cost factor in agronomy.

Regarding international conventions, N₂O is among the GHGs targeted by the United Nations Framework Convention on Climate Change, but few countries include N₂O in their GHG mitigation plan.¹⁸⁸ To prevent ozone depletion, the Montreal Protocol was followed and suggestions have been made to mitigate N₂O emission under the existing international treaty,¹⁸⁹ but there was little progress. The lack of progress in policy could be partly attributed to the limited guidance available from science. Since the emissions vary temporally and spatially, it is challenging to calculate the exact attribution of emissions and precise quantification of mitigation. Due to the inherent challenges in adequate quantification and attribution of emissions to an individual polluter, N₂O mitigation generally is enabled in combination with tackling excess agricultural N. The global goal to “halve nitrogen waste”¹⁹⁰ has been laid out as a rule of thumb for policies in general, which could bring multiple activities of international agreements in line.

EXPERIMENTAL PROCEDURES

Resource availability

Lead contact

Requests and questions should be directed to Feng Zhou (zhouf@pku.edu.cn).

Materials availability

This study did not generate new unique materials.

Data and code availability

All original codes have been deposited at Figshare under <https://figshare.com/s/fee2a89e0bd62766879f> and are publicly available as of the date of publication. Requests for any additional information and resources required to re-analyze the data reported in this paper should be directed to and will be fulfilled by the lead contact, Feng Zhou (zhouf@pku.edu.cn).

Global cropland N₂O emissions and mitigation potentials

This review assessed the cropland-N₂O emissions and mitigation potentials based on the recent advances in mechanisms of N₂O production, modeling, and reduction effect size of measures from meta-analyses. The cropland N₂O emissions for the past six decades were evaluated and updated with a newly developed LME model and a set of high-resolution management datasets (e.g., fertilization rate, types, timing, placement).⁷ The mitigation potentials were achieved through optimizing N fertilization, improving irrigation practices, shifting human diets, or a combination of these methods. Here we briefly summarize the approaches for modeling, prediction, and mitigation assessments in this review.

Current N₂O emissions

Observation datasets

To obtain comprehensive insights into available N₂O emissions observation, we compiled an up-to-date N₂O emissions observation dataset from online data repositories and peer-reviewed meta-analyses (up to 2023) (Tables S1 and S2). Information on geographic location (i.e., longitude and latitude), crop, and observation year for each site were recorded. Such processes formed a global observation dataset including 515 sites from 37 countries during the period 1978–2018 (Figure S1 and Table S1). The spatial pattern of the observation sites by crop and region is presented in Figure S1.

Data synthesis of previous estimates

We summarized N₂O emission estimates from top-down and bottom-up approaches. Top-down approaches included estimates by Davidson⁹² and Thompson et al.⁹¹ (see Table 1). However, these results cannot separate natural and different anthropogenic sources, so they were not included in Figure 2. Bottom-up approaches include inventories, statistical models, and process models. (1) Emission inventory includes estimates from FAO,⁹⁴ EDGAR,¹⁹¹ and GAINS.⁹⁶ (2) Statistical models include the non-linear Δ EF model proposed by Shcherbak et al.,⁷⁶ the NL-N-RR model used by Gerber et al.,⁹⁰ the SRNM model from Wang et al.,¹⁸ and crop-specific models from Cui et al.⁷ In addition, machine-learning models (e.g., random forest) were also used with extended observations available. (3) Process-based models for estimating cropland N₂O emissions mainly included DLEM, DNDC, and DAYCENT (Table S2). Since estimates from process-based models were mostly unavailable except for NMIP, these modeling studies are summarized in Table S2 but excluded for estimates comparison in Figure 2. To facilitate comparison,

fertilizer-induced cropland N₂O emissions were isolated from estimates of FAOSTAT and GAINS, EDGAR, and NMIP following the methods in Wang et al.¹⁸

Modeling of crop-specific N₂O emissions for 1961–2020

We only calculated fertilized-induced (including synthetic fertilizer, manure, and crop residue) N₂O emissions from cropland while excluding background emissions. The former were calculated as the product of crop-specific EFs and fertilizer N inputs, in which the crop-specific N₂O EFs during 1961–2020 were updated with the constructed LME model by Cui et al.⁷ The EF model of Cui et al.⁷ was selected to account for impacts from various climatic, edaphic, and management-related factors while avoiding assumptions associated with the representation of complex N₂O production processes as in process-based models.⁷ In this model, EFs are mainly regulated by bulk density (BD), soil clay content (Clay), soil organic carbon (SOC) content, soil pH (pH), growing-season cumulative precipitation (Pre), growing-season mean daily air temperature (Tem), humidity index (PPE), fertilizer application per harvested area (Nrate), type (i.e., fraction of ammonium nitrate [AN], calcium ammonium nitrate [CAN], Manure), frequency, placement, irrigation fraction (i.e., rain-fed or irrigated for upland crops, continuously or intermittently flooded for rice), tillage fraction (i.e., no tillage or tillage), and sampling duration (Days, i.e., days from starting and ending dates). Detailed information for the modeling can be found in Cui et al.⁷

Projections

The global patterns of crop-specific N₂O emissions during 1961–2020 were estimated using the LME models by crop at 5-arcminute spatial resolution. Datasets for the estimation were classified into two groups (i.e., environmental data and agricultural management data), in which agricultural management data were specially developed for this review. Detailed information about the datasets is listed in Table S4.

Environmental data

Soil data (i.e., BD, Clay, SOC, and pH) were extracted from the Harmonized World Soil Database (HWSD) v1.2 (1 × 1 km).¹⁹¹ Cumulative precipitation, air temperature, and cumulative potential evapotranspiration over the growing season were acquired from the CRU TS V4.06 climate dataset (0.5° × 0.5°),¹⁹² where the growing season in each grid cell was identified as the period between the planting and harvesting dates obtained from Sacks et al.¹⁹³ Humidity index was calculated by cumulative precipitation divided by potential evapotranspiration over the growing season. All data were resampled into grid maps at 5-arcminute spatial resolution.

Agricultural management datasets

For fertilization, crop-specific N fertilizer inputs (including synthetic N fertilizers, crop residues, and manure), fertilizer types, and placement during 1961–2020 were obtained from Adalibieke et al.¹¹² The frequency (i.e., one or multiple times) of N fertilization was the same as in Cui et al.⁷ and we assumed that the frequency remained constant during the study period. For tillage, the fraction of tillage by crop during 1961–2020 was obtained from Adalibieke et al.,¹¹² which was constructed with the country and province (or state) level no-tillage area data during 1961–2020 and downscaled to grid with the method of Porwollik et al.¹⁹⁴ For irrigation, the History Database of the Global Environment (HYDE version 3.2)¹⁹⁵ and the MIRCA2000¹⁹⁶ dataset were used to compile the global crop-specific irrigation proportion data from the period 1961–2020. Categories of cropland in HYDE provided new distinctions with irrigated and rain-fed crops (upland crops other than rice) and irrigated and rain-fed rice during 1960–2017. The national-level dataset of “agricultural area actually irrigated” was obtained from FAOSTAT,⁹⁴ which was used to scale the baseline year 2015 maps of irrigated area from HYDE over the period 2016–2020. The area of irrigated upland crops from HYDE was first disaggregated into 21 crops based on the irrigated proportion from MIRCA2000 per grid cell. We assumed an even share of irrigated area by each upland crop during the period 1961–2020, like MIRCA2000. Finally, the crop-specific irrigated area was masked by reporting harvested area, from which the irrigated proportion of each crop can be calculated as the crop-specific irrigated area divided by the physical area of each crop. For rice, we further divided irrigated rice into continuously and intermittently flooded systems as provided by Cui et al.⁷ and we assumed that the irrigation proportion was kept the same during the study period.

Mitigation potentials

To explore the mitigation potentials of N₂O emissions from cropland at the present time, we estimated the N₂O reduction in 2020 by optimizing N fertilization, improving irrigation practices, shifting dietary structure, and their combinations.

Optimizing N fertilization

First, the global crop-specific mitigation potential of N₂O emissions from optimizing N inputs was estimated by reducing fertilizer N inputs to cap N surplus to the limit. To cap N surplus without compromising yield, improved N fertilization management practices and technologies (e.g., N fertilization placement, N fertilization frequency, enhanced efficiency fertilizers [EEFs]) should also be implemented. Here, N surplus refers to the total N inputs (i.e., fertilizer N inputs, N bio-fixation, and N deposition) minus N uptake. The limits of spatial N surplus were generated with the regional N boundaries referred to by Schulte-Uebbing et al.⁸ who mapped the critical N surplus for arable land to close the yield gap in regions where environmental thresholds for eutrophication of terrestrial and aquatic ecosystems and nitrate in groundwater were not exceeded at 0.5° × 0.5° resolution for the year 2010. Given that the mitigation potentials were assessed for the year 2020 and that the harvested area was different from the year 2010, we solely used the critical N surplus rate (units: kg N ha⁻¹ year⁻¹) rather than the critical N surplus amount. Here, we assumed that as the critical N surplus for each crop in 2020 was the same as “critical N surplus in arable land in view of all thresholds simultaneously (kg N ha⁻¹ year⁻¹)” (one output file in Schulte-Uebbing et al.⁸), we could ensure that N thresholds for the cropland are within the planetary boundaries. The targeted N fertilizer input by crop was calculated as the sum of N surplus and N uptake minus N deposition and biological fixation. The averaged gridded dataset of N deposition rate for the year 2020 was obtained from the input4MIPs project.¹⁹⁷ Crop N uptake and N fixation were calculated by crop yield multiplied by coefficient for crop N content and crop N fixation. Crop yield, coefficient for crop N content, and crop N fixation were obtained from FAOSTAT and Zhang et al.¹¹⁴ The LME model was used to evaluate baseline emissions in 2020 (*E*₀) and emissions after reducing fertilizer N inputs (*E*₁). *E*₀ minus *E*₁ was the mitigation potential from optimizing N inputs. Next, the mitigation potentials of N₂O emissions from improving management practices and technologies (such as EEFs [including CRFs, NIs, UIs, and DIs], deep placement, and increasing splitting frequency of N fertilizers) were estimated based on meta-analyses and the spatial-explicit adoption rate of each measure. We collected effect size of these measures on N₂O-EFs (equal to N₂O emissions due to the same N application rate between treatments and control) from meta-analyses results^{23–25,116–122} and calculated the weighted average effect size of each measure based on sample size. We then applied these measures to each cropland grid based on their current N balance and social-economic level, as described in the equation

$$MP_{c,m,g} = E_{c,g} \times ES_{c,m} \times IRate_m, \quad (\text{Equation 1})$$

where *MP*_{*c,m,g*} is the mitigation potential (kg N per grid) from measures *m* of crop *c* in grid *g*, *E*_{*c,g*} is the emissions of crop *c* in grid *g*, *ES*_{*c,m*} is effect size (%) of measures *m* on crop *c*, and *IRate*_{*m*} is the potential implementation rate (%) of measures *m*. Determination criteria of the potential implementation rate for different measures were from Gu et al.¹¹⁶ For details, see Table S5.

Improving irrigation practices

The global crop-specific potential of N₂O mitigation from improving irrigation practices was estimated based on the average effect size from meta-analyses and the spatial-explicit adoption rate of improving irrigation. We collected the effect size of improving irrigation on N₂O-EFs (equal to N₂O emissions due to the same N application rate between treatments and control) from meta-analyses results^{25,116,121} and calculated the weighted average effect size of each measure based on sample size. We then applied the improving irrigation into each cropland grid based on the cropland N harvest rate as described by the equation

$$MP_{c,g} = E_{c,g} \times ES_c \times IRate, \quad (\text{Equation 2})$$

where *MP*_{*c,g*} is the mitigation potential (kg N per grid) from improving irrigation of crop *c* in grid *g*, *E*_{*c,g*} is the emissions (kg N per grid) of crop *c* in grid *g*, *ES*_{*c*} is the effect size (%) of improving irrigation on crop *c*, and *IRate* is the potential implementation rate (%) of improving irrigation. Determination criteria of the potential implementation rate for improving irrigation were from Gu et al.¹¹⁶ For details, see Table S5.

Shifting diet structure

The global crop-specific potential of N₂O mitigation from shifting diet structure was estimated by changing the production of crops to meet the demand of food recommended by the Universal Healthy Reference Diet (EAT-Lancet). The emissions after shifting diet structure were calculated as per the following equation:

$$Ediet_c = \frac{E0_c}{Harea_c} \times Harea_{c,diet}, \quad (\text{Equation 3})$$

where, *E*_{0_c} and *Ediet*_{*c*} are the emissions (kg N per grid) before and after shifting diet structure, respectively, and *Harea*_{*c*} and *Harea*_{*c,diet*} are current and opti-

mized harvested area (ha) of crop *c*, respectively. Global or country-level changes of crop productions were from the EAT-Lancet Commission¹⁵² and Sun et al.²⁶ To calculate the changes of harvested area, we assumed the productions per harvested area here were consistent.

Combinations of measures

All measures from optimizing N fertilization (reducing N inputs, EEFs, N fertilization placement, and N fertilization frequency), irrigation practice, and shifting diet structure were combined to achieve the maximum mitigation potentials. The emissions of the combination can be calculated from the following equation:

$$E_{all} = Nrate \times \prod_m (1 - ES_m \times IRate_m) \times Harea_{diet} \times EF, \quad (\text{Equation 4})$$

where *E*_{*all*} is the emissions (kg N per grid) by combinations of all measures, *Nrate* is optimized fertilizer N application per harvested area (kg N/ha), *ES*_{*m*} is effect size (%) of management practices and technologies *m*, *IRate*_{*m*} is their potential implementation rate (%), *Harea*_{*diet*} is harvested area after shifting diet structure, and *EF* is emission factor (%).

SUPPLEMENTAL INFORMATION

Supplemental information can be found online at <https://doi.org/10.1016/j.oneear.2024.01.005>.

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AUTHOR CONTRIBUTIONS

F.Z.: conceptualization, writing – review and editing, supervision, funding acquisition, and project administration. X.C.: conceptualization, methodology, software, validation, investigation, formal analysis, visualization, writing – original draft, and funding acquisition. Y.B.: methodology, software, validation, formal analysis, investigation, writing – original draft, and visualization. W.A.: methodology, software, validation, formal analysis, investigation, writing – original draft, and visualization. W.W.: writing – review and editing. X.Z.: writing – review and editing. E.A.D.: writing – review and editing. Z.S.: data curation and writing – review and editing. H.T.: writing – review and editing. P.S.: writing – review and editing.

DECLARATION OF INTERESTS

The authors declare no competing interests.

REFERENCES

1. Intergovernmental Panel on Climate Change (IPCC) (2021). The Earth's Energy Budget, Climate Feedbacks and Climate Sensitivity. In *Climate Change 2021 – The Physical Science Basis: Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (Cambridge University Press), pp. 923–1054.
2. Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., Davidson, E.A., Ciais, P., Jackson, R.B., Janssens-Maenhout, G., et al. (2020). A comprehensive quantification of global nitrous oxide sources and sinks. *Nature* 586, 248–256.
3. Lobell, D.B., Di Tommaso, S., and Burney, J.A. (2022). Globally ubiquitous negative effects of nitrogen dioxide on crop growth. *Sci. Adv.* 8, eabm9909.
4. Xu, R., Tian, H., Pan, S., Prior, S.A., Feng, Y., and Dangal, S.R.S. (2020). Global N₂O emissions from cropland driven by nitrogen addition and

- environmental factors: Comparison and uncertainty analysis. *Global Biogeochem. Cycles* 34, e2020GB006698.
5. Hassan, M.U., Aamer, M., Mahmood, A., Awan, M.I., Barbanti, L., Seilman, M.F., Bakhsh, G., Alkharabsheh, H.M., Babur, E., Shao, J., et al. (2022). Management Strategies to Mitigate N₂O Emissions in Agriculture. *Life-Basel* 12, 439.
 6. Haider, A., Bashir, A., and Husnain, M.I.U. (2020). Impact of agricultural land use and economic growth on nitrous oxide emissions: Evidence from developed and developing countries. *Sci. Total Environ.* 741, 140421.
 7. Cui, X., Zhou, F., Ciais, P., Davidson, E.A., Tubiello, F.N., Niu, X., Ju, X., Canadell, J.G., Bouwman, A.F., Jackson, R.B., et al. (2021). Global mapping of crop-specific emission factors highlights hotspots of nitrous oxide mitigation. *Nat. Food* 2, 886–893.
 8. Schulte-Uebbing, L.F., Beusen, A.H.W., Bouwman, A.F., and de Vries, W. (2022). From planetary to regional boundaries for agricultural nitrogen pollution. *Nature* 610, 507–512.
 9. Chapuis-Lardy, L., Wrage, N., Metay, A., Chotte, J.-L., and Bernoux, M. (2007). Soils, a sink for N₂O? A review. *Global Change Biol.* 13, 1–17.
 10. Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls? *Philos. Trans. R. Soc. Lond. B Biol. Sci.* 368, 20130122.
 11. Hu, H.W., Chen, D., and He, J.Z. (2015). Microbial regulation of terrestrial nitrous oxide formation: understanding the biological pathways for prediction of emission rates. *FEMS Microbiol. Rev.* 39, 729–749.
 12. Wang, C., Amon, B., Schulz, K., and Mehdi, B. (2021). Factors that influence nitrous oxide emissions from agricultural soils as well as their representation in simulation models: A review. *Agronomy* 11, 770.
 13. Smith, K.A. (2017). Changing views of nitrous oxide emissions from agricultural soil: key controlling processes and assessment at different spatial scales. *Eur. J. Soil Sci.* 68, 137–155.
 14. Tian, H., Yang, J., Xu, R., Lu, C., Canadell, J.G., Davidson, E.A., Jackson, R.B., Arneeth, A., Chang, J., Ciais, P., et al. (2019). Global soil nitrous oxide emissions since the preindustrial era estimated by an ensemble of terrestrial biosphere models: Magnitude, attribution, and uncertainty. *Global Change Biol.* 25, 640–659.
 15. Bouwman, A.F., Boumans, L.J.M., and Batjes, N.H. (2002). Modeling global annual N₂O and NO emissions from fertilized fields. *Global Biogeochem. Cycles* 16, 28–1–28–9.
 16. Reay, D.S., Davidson, E.A., Smith, K.A., Smith, P., Melillo, J.M., Dentener, F., and Crutzen, P.J. (2012). Global agriculture and nitrous oxide emissions. *Nat. Clim. Change* 2, 410–416.
 17. Griffis, T.J., Lee, X., Baker, J.M., Russelle, M.P., Zhang, X., Venterea, R., and Millet, D.B. (2013). Reconciling the differences between top-down and bottom-up estimates of nitrous oxide emissions for the US Corn Belt. *Global Biogeochem. Cycles* 27, 746–754.
 18. Wang, Q., Zhou, F., Shang, Z., Ciais, P., Winiwarter, W., Jackson, R.B., Tubiello, F.N., Janssens-Maenhout, G., Tian, H., Cui, X., et al. (2020). Data-driven estimates of global nitrous oxide emissions from croplands. *Natl. Sci. Rev.* 7, 441–452.
 19. Leip, A., Busto, M., and Winiwarter, W. (2011). Developing spatially stratified N₂O emission factors for Europe. *Environ. Pollut.* 159, 3223–3232.
 20. Zhang, W., Cao, G., Li, X., Zhang, H., Wang, C., Liu, Q., Chen, X., Cui, Z., Shen, J., Jiang, R., et al. (2016). Closing yield gaps in China by empowering smallholder farmers. *Nature* 537, 671–674.
 21. Cui, Z., Zhang, H., Chen, X., Zhang, C., Ma, W., Huang, C., Zhang, W., Mi, G., Miao, Y., Li, X., et al. (2018). Pursuing sustainable productivity with millions of smallholder farmers. *Nature* 555, 363–366.
 22. Cui, X., Shang, Z., Xia, L., Xu, R., Adalibieke, W., Zhan, X., Smith, P., and Zhou, F. (2022). Deceleration of cropland-N₂O emissions in china and future mitigation potentials. *Environ. Sci. Technol.* 56, 4665–4675.
 23. Xia, L., Lam, S.K., Chen, D., Wang, J., Tang, Q., and Yan, X. (2017). Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis. *Global Change Biol.* 23, 1917–1925.
 24. Li, T., Zhang, W., Yin, J., Chadwick, D., Norse, D., Lu, Y., Liu, X., Chen, X., Zhang, F., Powlson, D., and Dou, Z. (2018). Enhanced-efficiency fertilizers are not a panacea for resolving the nitrogen problem. *Global Change Biol.* 24, e511–e521.
 25. Kuang, W., Gao, X., Tenuta, M., and Zeng, F. (2021). A global meta-analysis of nitrous oxide emission from drip-irrigated cropping system. *Global Change Biol.* 27, 3244–3256.
 26. Sun, Z., Scherer, L., Tukker, A., Spawen-Lee, S.A., Bruckner, M., Gibbs, H.K., and Behrens, P. (2022). Dietary change in high-income nations alone can lead to substantial double climate dividend. *Nat. Food* 3, 29–37.
 27. Semba, R.D., de Pee, S., Kim, B., McKenzie, S., Nachman, K., and Bloem, M.W. (2020). Adoption of the ‘planetary health diet’ has different impacts on countries’ greenhouse gas emissions. *Nat. Food* 1, 481–484.
 28. Braker, G., and Conrad, R. (2011). Chapter 2 - Diversity, Structure, and Size of N₂O-Producing Microbial Communities in Soils—What Matters for Their Functioning? In *Advances in Applied Microbiology*, A.I. Laskin, S. Sariaslani, and G.M. Gadd, eds. (Academic Press), pp. 33–70.
 29. Kuypers, M.M.M., Marchant, H.K., and Kartal, B. (2018). The microbial nitrogen-cycling network. *Nat. Rev. Microbiol.* 16, 263–276.
 30. Thakur, I.S., and Medhi, K. (2019). Nitrification and denitrification processes for mitigation of nitrous oxide from waste water treatment plants for biovalorization: Challenges and opportunities. *Bioresour. Technol.* 282, 502–513.
 31. Ward, B.B. (2005). Temporal variability in nitrification rates and related biogeochemical factors in Monterey Bay, California, USA. *Mar. Ecol. Prog. Ser.* 292, 97–109.
 32. Stein, L.Y. (2020). The long-term relationship between microbial metabolism and greenhouse gases. *Trends Microbiol.* 28, 500–511.
 33. Ma, Z., Gao, X., Tenuta, M., Kuang, W., Gui, D., and Zeng, F. (2018). Urea fertigation sources affect nitrous oxide emission from a drip-fertigated cotton field in northwestern China. *Agric. Ecosyst. Environ.* 265, 22–30.
 34. Medhi, K., Singhal, A., Chauhan, D.K., and Thakur, I.S. (2017). Investigating the nitrification and denitrification kinetics under aerobic and anaerobic conditions by *Paracoccus denitrificans* ISTOD1. *Bioresour. Technol.* 242, 334–343.
 35. Zhang, Q.L., Liu, Y., Ai, G.M., Miao, L.L., Zheng, H.Y., and Liu, Z.P. (2012). The characteristics of a novel heterotrophic nitrification–aerobic denitrification bacterium, *Bacillus methylotrophicus* strain L7. *Bioresour. Technol.* 108, 35–44.
 36. Zumft, W.G. (2005). Nitric oxide reductases of prokaryotes with emphasis on the respiratory, heme–copper oxidase type. *J. Inorg. Biochem.* 99, 194–215.
 37. Blomberg, M.R.A., and Adelroth, P. (2018). Mechanisms for enzymatic reduction of nitric oxide to nitrous oxide - A comparison between nitric oxide reductase and cytochrome c oxidase. *Biochim. Biophys. Acta Bioenerg.* 1859, 1223–1234.
 38. Suharti, Strampaad, M.J.F., Schröder, I., and de Vries, S. (2001). A novel copper containing menaquinol no reductase from *Bacillus azotoformans*. *Biochemistry* 40, 2632–2639.
 39. Wrage, N., Velthof, G.L., van Beusichem, M.L., and Oenema, O. (2001). Role of nitrifier denitrification in the production of nitrous oxide. *Soil Biol. Biochem.* 33, 1723–1732.
 40. Colliver, B.B., and Stephenson, T. (2000). Production of nitrogen oxide and dinitrogen oxide by autotrophic nitrifiers. *Biotechnol. Adv.* 18, 219–232.
 41. Poth, M., and Focht, D.D. (1985). N-15 Kinetic-Analysis of N₂O Production by *Nitrosomonas-Europaea* - an Examination of Nitrifier Denitrification. *Appl. Environ. Microbiol.* 49, 1134–1141.
 42. Kraft, B., Tegetmeyer, H.E., Sharma, R., Klotz, M.G., Ferdelman, T.G., Hettich, R.L., Geelhoed, J.S., and Strous, M. (2014). The environmental controls that govern the end product of bacterial nitrate respiration. *Science* 345, 676–679.
 43. Smith, M.S., and Zimmerman, K. (1981). Nitrous-oxide production by non-denitrifying soil nitrate reducers. *Soil Sci. Soc. Am. J.* 45, 865–871.
 44. Simon, J. (2002). Enzymology and bioenergetics of respiratory nitrite ammonification. *FEMS Microbiol. Rev.* 26, 285–309.
 45. Heil, J., Liu, S., Vereecken, H., and Brüggemann, N. (2015). Abiotic nitrous oxide production from hydroxylamine in soils and their dependence on soil properties. *Soil Biol. Biochem.* 84, 107–115.
 46. Rubasinghege, G., Spak, S.N., Stanier, C.O., Carmichael, G.R., and Grassian, V.H. (2011). Abiotic mechanism for the formation of atmospheric nitrous oxide from ammonium nitrate. *Environ. Sci. Technol.* 45, 2691–2697.
 47. Heil, J., Vereecken, H., and Brüggemann, N. (2016). A review of chemical reactions of nitrification intermediates and their role in nitrogen cycling and nitrogen trace gas formation in soil. *Eur. J. Soil Sci.* 67, 23–39.
 48. Harris, E., Diaz-Pines, E., Stoll, E., Schloter, S., Schulz, S., Duffner, C., Li, K., Moore, K.L., Ingrisch, J., Reinthaler, D., et al. (2021). Denitrifying pathways dominate nitrous oxide emissions from managed grassland during drought and rewetting. *Sci. Adv.* 7, eabb7118.
 49. Wei, H., Song, X., Liu, Y., Wang, R., Zheng, X., Butterbach-Bahl, K., Venterea, R.T., Wu, D., and Ju, X. (2023). In situ N-15-N₂O site preference

- and O-2 concentration dynamics disclose the complexity of N₂O production processes in agricultural soil. *Global Change Biol.* 29, 4910–4923.
50. Venterea, R.T., Clough, T.J., Coulter, J.A., Breuillin-Sessoms, F., Wang, P., and Sadowsky, M.J. (2015). Ammonium sorption and ammonia inhibition of nitrite-oxidizing bacteria explain contrasting soil N₂O production. *Sci. Rep.* 5, 12153.
51. Maharjan, B., and Venterea, R.T. (2013). Nitrite intensity explains N management effects on N₂O emissions in maize. *Soil Biol. Biochem.* 66, 229–238.
52. Venterea, R.T. (2007). Nitrite-driven nitrous oxide production under aerobic soil conditions: kinetics and biochemical controls. *Global Change Biol.* 13, 1798–1809.
53. Meurer, K.H.E., Franko, U., Stange, C.F., Rosa, J.D., Madari, B.E., and Jungkunst, H.F. (2016). Direct nitrous oxide (N₂O) fluxes from soils under different land use in Brazil—a critical review. *Environ. Res. Lett.* 11, 023001.
54. Lesschen, J.P., Velthof, G.L., de Vries, W., and Kros, J. (2011). Differentiation of nitrous oxide emission factors for agricultural soils. *Environ. Pollut.* 159, 3215–3222.
55. Kool, D.M., Dolfing, J., Wrage, N., and Van Groenigen, J.W. (2011). Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. *Soil Biol. Biochem.* 43, 174–178.
56. Stieglmeier, M., Mooshammer, M., Kitzler, B., Wanek, W., Zechmeister-Boltenstern, S., Richter, A., and Schleper, C. (2014). Aerobic nitrous oxide production through N-nitrosating hybrid formation in ammonia-oxidizing archaea. *ISME J.* 8, 1135–1146.
57. Zhu, X., Burger, M., Doane, T.A., and Horwath, W.R. (2013). Ammonia oxidation pathways and nitrifier denitrification are significant sources of N₂O and NO under low oxygen availability. *Proc. Natl. Acad. Sci. USA* 110, 6328–6333.
58. Liu, C., Wang, K., Meng, S., Zheng, X., Zhou, Z., Han, S., Chen, D., and Yang, Z. (2011). Effects of irrigation, fertilization and crop straw management on nitrous oxide and nitric oxide emissions from a wheat–maize rotation field in northern China. *Agric. Ecosyst. Environ.* 140, 226–233.
59. Liu, L., Hu, C., Olesen, J.E., Ju, Z., Yang, P., and Zhang, Y. (2013). Warming and nitrogen fertilization effects on winter wheat yields in northern China varied between four years. *Field Crops Res.* 151, 56–64.
60. Liu, L., Hu, C., Yang, P., Ju, Z., Olesen, J.E., and Tang, J. (2016). Experimental warming-driven soil drying reduced N₂O emissions from fertilized crop rotations of winter wheat–soybean/fallow, 2009–2014. *Agric. Ecosyst. Environ.* 219, 71–82.
61. Wagner-Riddle, C., Congreves, K.A., Abalos, D., Berg, A.A., Brown, S.E., Ambadan, J.T., Gao, X., and Tenuta, M. (2017). Globally important nitrous oxide emissions from croplands induced by freeze–thaw cycles. *Nat. Geosci.* 10, 279–283.
62. Butterbach-Bahl, K., and Wolf, B. (2017). Warming from freezing soils. *Nat. Geosci.* 10, 248–249.
63. Butterbach-Bahl, K., and Dannenmann, M. (2011). Denitrification and associated soil N₂O emissions due to agricultural activities in a changing climate. *Curr. Opin. Environ. Sustain.* 3, 389–395.
64. Iqbal, J., Necpalova, M., Archontoulis, S.V., Anex, R.P., Bourguignon, M., Herzmann, D., Mitchell, D.C., Sawyer, J.E., Zhu, Q., and Castellano, M.J. (2018). Extreme weather-year sequences have nonadditive effects on environmental nitrogen losses. *Global Change Biol.* 24, e303–e317.
65. Knapp, A.K., Beier, C., Briske, D.D., Classen, A.T., Luo, Y., Reichstein, M., Smith, M.D., Smith, S.D., Bell, J.E., Fay, P.A., et al. (2008). Consequences of More Extreme Precipitation Regimes for Terrestrial Ecosystems. *Bioscience* 58, 811–821.
66. BORKEN, W., and MATZNER, E. (2009). Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. *Global Change Biol.* 15, 808–824.
67. Li, L., Zheng, Z., Wang, W., Biederman, J.A., Xu, X., Ran, Q., Qian, R., Xu, C., Zhang, B., Wang, F., et al. (2020). Terrestrial N₂O emissions and related functional genes under climate change: A global meta-analysis. *Global Change Biol.* 26, 931–943.
68. Pereira, J., Figueiredo, N., Goufo, P., Carneiro, J., Morais, R., Carranca, C., Coutinho, J., and Trindade, H. (2013). Effects of elevated temperature and atmospheric carbon dioxide concentration on the emissions of methane and nitrous oxide from Portuguese flooded rice fields. *Atmos. Environ.* 80, 464–471.
69. Bijoor, N.S., Czimczik, C.I., Pataki, D.E., and Billings, S.A. (2008). Effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn. *Global Change Biol.* 14, 2119–2131.
70. Bhattacharyya, P., Roy, K.S., Neogi, S., Dash, P.K., Nayak, A.K., Mohanty, S., Baig, M.J., Sarkar, R.K., and Rao, K.S. (2013). Impact of elevated CO₂ and temperature on soil C and N dynamics in relation to CH₄ and N₂O emissions from tropical flooded rice (*Oryza sativa* L.). *Sci. Total Environ.* 461–462, 601–611.
71. Wang, B., Li, J., Wan, Y., Li, Y., Qin, X., Gao, Q., Waqas, M.A., Wilkes, A., Cai, W., You, S., and Zhou, S. (2018). Responses of yield, CH₄ and N₂O emissions to elevated atmospheric temperature and CO₂ concentration in a double rice cropping system. *Eur. J. Agron.* 96, 60–69.
72. Stith, S., Cox, P.M., Collins, W.J., and Huntingford, C. (2007). Indirect radiative forcing of climate change through ozone effects on the land-carbon sink. *Nature* 448, 791–794.
73. Grave, R.A., Nicoloso, R.D.S., Cassol, P.C., da Silva, M.L.B., Mezzari, M.P., Aita, C., and Wuaden, C.R. (2018). Determining the effects of tillage and nitrogen sources on soil N₂O emission. *Soil Tillage Res.* 175, 1–12.
74. Snyder, C.S., Bruulsema, T.W., Jensen, T.L., and Fixen, P.E. (2009). Review of greenhouse gas emissions from crop production systems and fertilizer management effects. *Agric. Ecosyst. Environ.* 133, 247–266.
75. Hoben, J.P., Gehl, R.J., Millar, N., Grace, P.R., and Robertson, G.P. (2011). Nonlinear nitrous oxide (N₂O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. *Global Change Biol.* 17, 1140–1152.
76. Shcherbak, I., Millar, N., and Robertson, G.P. (2014). Global metaanalysis of the nonlinear response of soil nitrous oxide (N₂O) emissions to fertilizer nitrogen. *Proc. Natl. Acad. Sci. USA* 111, 9199–9204.
77. Song, X., Liu, M., Ju, X., Gao, B., Su, F., Chen, X., and Rees, R.M. (2018). Nitrous oxide emissions increase exponentially when optimum nitrogen fertilizer rates are exceeded in the North China Plain. *Environ. Sci. Technol.* 52, 12504–12513.
78. Qin, S., Yuan, H., Hu, C., Oenema, O., Zhang, Y., and Li, X. (2014). Determination of potential N₂O-reductase activity in soil. *Soil Biol. Biochem.* 70, 205–210.
79. Schwenke, G.D., and Haigh, B.M. (2019). Can split or delayed application of N fertiliser to grain sorghum reduce soil N₂O emissions from sub-tropical Vertosols and maintain grain yields? *Soil Res.* 57, 859.
80. Rochette, P., Angers, D.A., Chantigny, M.H., and Bertrand, N. (2008). Nitrous oxide emissions respond differently to no-till in a loam and a heavy clay soil. *Soil Sci. Soc. Am. J.* 72, 1363–1369.
81. Pareja-Sánchez, E., Cantero-Martínez, C., Álvaro-Fuentes, J., and Plaza-Bonilla, D. (2020). Impact of tillage and N fertilization rate on soil N₂O emissions in irrigated maize in a Mediterranean agroecosystem. *Agric. Ecosyst. Environ.* 287, 106687.
82. Ye, X., Liu, H., Zhang, X., Ma, J., Han, B., Li, W., Zou, H., Zhang, Y., and Lin, X. (2020). Impacts of irrigation methods on greenhouse gas emissions/absorptions from vegetable soils. *J. Soils Sediments* 20, 723–733.
83. Tang, J., Wang, J., Li, Z., Wang, S., and Qu, Y. (2018). Effects of irrigation regime and nitrogen fertilizer management on CH₄, N₂O and CO₂ emissions from saline-alkaline paddy fields in northeast china. *Sustainability* 10, 475.
84. Hagedorn, J.G., Davidson, E.A., Fisher, T.R., Fox, R.J., Zhu, Q., Gustafson, A.B., Koontz, E., Castro, M.S., and Lewis, J. (2022). Effects of drainage water management in a corn–soy rotation on soil N₂O and CH₄ fluxes. *Nitrogen* 3, 128–148.
85. Trost, B., Prochnow, A., Drastig, K., Meyer-Aurich, A., Ellmer, F., and Baumecker, M. (2013). Irrigation, soil organic carbon and N₂O emissions. *Agron. Sustain. Dev.* 33, 733–749.
86. Saleh-Lakha, S., Shannon, K.E., Henderson, S.L., Goyer, C., Trevors, J.T., Zebbarh, B.J., and Burton, D.L. (2009). Effect of pH and temperature on denitrification gene expression and activity in *Pseudomonas mandelii*. *Appl. Environ. Microbiol.* 75, 3903–3911.
87. Liu, B., Mørkved, P.T., Frostegård, A., and Bakken, L.R. (2010). Denitrification gene pools, transcription and kinetics of NO, N₂O and N₂ production as affected by soil pH. *FEMS Microbiol. Ecol.* 72, 407–417.
88. Zhou, F., Shang, Z., Zeng, Z., Piao, S., Ciais, P., Raymond, P.A., Wang, X., Wang, R., Chen, M., Yang, C., et al. (2015). New model for capturing the variations of fertilizer-induced emission factors of N₂O. *Global Biogeochem. Cycles* 29, 885–897.
89. Shang, Z., Zhou, F., Smith, P., Saikawa, E., Ciais, P., Chang, J., Tian, H., Del Grosso, S.J., Ito, A., Chen, M., et al. (2019). Weakened growth of cropland-N₂O emissions in China associated with nationwide policy interventions. *Global Change Biol.* 25, 3706–3719.
90. Gerber, J.S., Carlson, K.M., Makowski, D., Mueller, N.D., Garcia de Cortazar-Atauri, I., Havlik, P., Herrero, M., Launay, M., O’Connell, C.S., Smith, P., and West, P.C. (2016). Spatially explicit estimates of N₂O emissions from croplands suggest climate mitigation opportunities from improved fertilizer management. *Global Change Biol.* 22, 3383–3394.

91. Thompson, R.L., Lassaletta, L., Patra, P.K., Wilson, C., Wells, K.C., Gressent, A., Koffi, E.N., Chipperfield, M.P., Winiwarter, W., Davidson, E.A., et al. (2019). Acceleration of global N_2O emissions seen from two decades of atmospheric inversion. *Nat. Clim. Change* 9, 993–998.
92. Davidson, E.A. (2009). The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nat. Geosci.* 2, 659–662.
93. Tesfaye, K., Takele, R., Sapkota, T.B., Khatri-Chhetri, A., Solomon, D., Stirling, C., and Albanito, F. (2021). Model comparison and quantification of nitrous oxide emission and mitigation potential from maize and wheat fields at a global scale. *Sci. Total Environ.* 782, 146696.
94. FAOSTAT (2022). Food and Agricultural Organization of the United Nations.
95. Janssens-Maenhout, G., Crippa, M., Guizzardi, D., Muntean, M., Schaaf, E., Dentener, F., Bergamaschi, P., Pagliari, V., Olivier, J.G.J., Peters, J.A.H.W., et al. (2019). EDGAR v4.3.2 Global Atlas of the three major greenhouse gas emissions for the period 1970–2012. *Earth Syst. Sci. Data* 11, 959–1002.
96. Winiwarter, W., Höglund-Isaksson, L., Klimont, Z., Schöpp, W., and Amann, M. (2018). Technical opportunities to reduce global anthropogenic emissions of nitrous oxide. *Environ. Res. Lett.* 13, 14011.
97. Tian, X., Cong, J., Wang, H., Zheng, H., Wang, Z., Chu, Y., Wang, Y., Xue, Y., Yin, Y., and Cui, Z. (2023). Cropland nitrous oxide emissions exceed the emissions of RCP 2.6: A global spatial analysis. *Sci. Total Environ.* 858, 159738.
98. Yang, Y., Liu, L., Zhang, F., Zhang, X., Xu, W., Liu, X., Wang, Z., and Xie, Y. (2021). Soil Nitrous Oxide Emissions by Atmospheric Nitrogen Deposition over Global Agricultural Systems. *Environ. Sci. Technol.* 55, 4420–4429.
99. Del Grosso, S.J., Ojima, D.S., Parton, W.J., Stehfest, E., Heistermann, M., DeAngelo, B., and Rose, S. (2009). Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. *Global Planet. Change* 67, 44–50.
100. Tian, H., Bian, Z., Shi, H., Qin, X., Pan, N., Lu, C., Pan, S., Tubiello, F.N., Chang, J., Conchedda, G., et al. (2022). History of anthropogenic Nitrogen inputs (HaNi) to the terrestrial biosphere: a 5 arcmin resolution annual dataset from 1860 to 2019. *Earth Syst. Sci. Data* 14, 4551–4568.
101. Van Grinsven, H.J.M., Spiertz, J.H.J., Westhoek, H.J., Bouwman, A.F., and Erisman, J.W. (2014). Nitrogen use and food production in European regions from a global perspective. *J. Agric. Sci.* 152, S9–S19.
102. Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., and Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. *Environ. Res. Lett.* 9, 105011.
103. Davidson, E.A., Galloway, J.N., Millar, N., and Leach, A.M. (2014). N-related greenhouse gases in North America: innovations for a sustainable future. *Curr. Opin. Environ. Sustain.* 9–10, 1–8.
104. Deng, J., Guo, L., Salas, W., Ingraham, P., Charrier-Klobas, J.G., Frolking, S., and Li, C. (2022). A decreasing trend of nitrous oxide emissions from California cropland from 2000 to 2015. *Earth's Future* 10, e2021EF002526.
105. Lim, J.Y., Islam Bhuiyan, M.S., Lee, S.B., Lee, J.G., and Kim, P.J. (2021). Agricultural nitrogen and phosphorus balances of Korea and Japan: Highest nutrient surplus among OECD member countries. *Environ. Pollut.* 286, 117353.
106. Saritas, O., and Kuzminov, I. (2017). Global challenges and trends in agriculture: impacts on Russia and possible strategies for adaptation. *Fore-sight* 19, 218–250.
107. Ivanova, S., and Nosov, V. (2011). Development of Agriculture in Russia and its Impact on Fertilizer Use.
108. Green, D.J., and Vokes, R.W. (1997). Agriculture and the transition to the market in Asia. *J. Comp. Econ.* 25, 256–280.
109. Ito, A., Nishina, K., Ishijima, K., Hashimoto, S., and Inatomi, M. (2018). Emissions of nitrous oxide (N_2O) from soil surfaces and their historical changes in East Asia: a model-based assessment. *Prog. Earth Planet. Sci.* 5, 55.
110. Ma, R., Yu, K., Xiao, S., Liu, S., Ciais, P., and Zou, J. (2022). Data-driven estimates of fertilizer-induced soil NH_3 , NO and N_2O emissions from croplands in China and their climate change impacts. *Global Change Biol.* 28, 1008–1022.
111. Lu, C., Yu, Z., Zhang, J., Cao, P., Tian, H., and Nevison, C. (2022). Century-long changes and drivers of soil nitrous oxide (N_2O) emissions across the contiguous United States. *Global Change Biol.* 28, 2505–2524.
112. Adalbieke, W., Cui, X., Cai, H., You, L., and Zhou, F. (2023). Global crop-specific nitrogen fertilization dataset in 1961–2020. *Sci. Data* 10, 617.
113. Wang, Y., Tao, F., Yin, L., and Chen, Y. (2022). Spatiotemporal changes in greenhouse gas emissions and soil organic carbon sequestration for major cropping systems across China and their drivers over the past two decades. *Sci. Total Environ.* 833, 155087.
114. Zhang, X., Davidson, E.A., Mauzerall, D.L., Searchinger, T.D., Dumas, P., and Shen, Y. (2015). Managing nitrogen for sustainable development. *Nature* 528, 51–59.
115. Vitousek, P.M., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger, J., Martinelli, L.A., Matson, P.A., et al. (2009). Nutrient imbalances in agricultural development. *Science* 324, 1519–1520.
116. Gu, B., Zhang, X., Lam, S.K., Yu, Y., van Grinsven, H.J.M., Zhang, S., Wang, X., Bodirsky, B.L., Wang, S., Duan, J., et al. (2023). Cost-effective mitigation of nitrogen pollution from global croplands. *Nature* 613, 77–84.
117. Fan, D., He, W., Smith, W.N., Drury, C.F., Jiang, R., Grant, B.B., Shi, Y., Song, D., Chen, Y., Wang, X., et al. (2022). Global evaluation of inhibitor impacts on ammonia and nitrous oxide emissions from agricultural soils: A meta-analysis. *Global Change Biol.* 28, 5121–5141.
118. Thapa, R., Chatterjee, A., Awale, R., McGranahan, D.A., and Daigh, A. (2016). Effect of enhanced efficiency fertilizers on nitrous oxide emissions and crop yields: A meta-analysis. *Soil Sci. Soc. Am. J.* 80, 1121–1134.
119. Tufail, M.A., Naeem, A., Arif, M.S., Farooq, T.H., Shahzad, S.M., Dar, A.A., Albasher, G., and Shakoor, A. (2022). Unraveling the efficacy of nitrification inhibitors (DCD and DMPP) in reducing nitrogen gases emissions across agroecosystems: A three-decade global data synthesis (1993–2021). *Fuel* 324, 124725.
120. Yang, M., Zhu, X., Bai, Y., Sun, D., Zou, H., Fang, Y., and Zhang, Y. (2021). Coated controlled-release urea creates a win-win scenario for producing more staple grains and resolving N loss dilemma worldwide. *J. Clean. Prod.* 288, 125660.
121. Yangjin, D., Wu, X., Bai, H., and Gu, J. (2021). A meta-analysis of management practices for simultaneously mitigating N_2O and NO emissions from agricultural soils. *Soil Tillage Res.* 213, 105142.
122. Chen, H., Liao, Q., and Liao, Y. (2021). Response of area- and yield-scaled N_2O emissions from croplands to deep fertilization: a meta-analysis of soil, climate, and management factors. *J. Sci. Food Agric.* 101, 4653–4661.
123. Millar, N., Robertson, G.P., Grace, P.R., Gehl, R.J., and Hoben, J.P. (2010). Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (Maize) production: an emissions reduction protocol for US Midwest agriculture. *Mitig. Adapt. Strategies Glob. Change* 15, 185–204.
124. Zhang, C., Ju, X., Powlson, D., Oenema, O., and Smith, P. (2019). Nitrogen surplus benchmarks for controlling N pollution in the main cropping systems of China. *Environ. Sci. Technol.* 53, 6678–6687.
125. Shi, X., Hu, H.-W., Müller, C., He, J.-Z., Chen, D., and Suter, H.C. (2016). Effects of the nitrification inhibitor 3,4-Dimethylpyrazole Phosphate on nitrification and nitrifiers in two contrasting agricultural soils. *Appl. Environ. Microbiol.* 82, 5236–5248.
126. Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B.L., Lassaletta, L., de Vries, W., Vermeulen, S.J., Herrero, M., Carlson, K.M., et al. (2018). Options for keeping the food system within environmental limits. *Nature* 562, 519–525.
127. West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K., and Siebert, S. (2014). Leverage points for improving global food security and the environment. *Science* 345, 325–328.
128. Charles, A., Rochette, P., Whalen, J.K., Angers, D.A., Chantigny, M.H., and Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agric. Ecosyst. Environ.* 236, 88–98.
129. Burger, M., and Venterea, R.T. (2011). Effects of nitrogen fertilizer types on nitrous oxide emissions. In *Understanding Greenhouse Gas Emissions from Agricultural Management*, L. Guo, A.S. Gunasekara, and L.L. McConnell, eds. (American Chemical Society), pp. 179–202.
130. Harrison, R., and Webb, J. (2001). A review of the effect of N fertilizer type on gaseous emissions. In *Advances in Agronomy*, pp. 65–108.
131. Li, X., Inubushi, K., and Sakamoto, K. (2002). Nitrous oxide concentrations in an Andisol profile and emissions to the atmosphere as influenced by the application of nitrogen fertilizers and manure. *Biol. Fertil. Soils* 35, 108–113.
132. Maaz, T.M., Sapkota, T.B., Eagle, A.J., Kantar, M.B., Bruulsema, T.W., and Majumdar, K. (2021). Meta-analysis of yield and nitrous oxide outcomes for nitrogen management in agriculture. *Global Change Biol.* 27, 2343–2360.

133. Association of American Plant Food Control Officials (AAPFCO) (1997). Official Publication No. 50. In Association of American Plant Food Control Officials, Inc.
134. Subbarao, G.V., Ito, O., Sahrawat, K.L., Berry, W.L., Nakahara, K., Ishikawa, T., Watanabe, T., Suenaga, K., Rondón, M., and Rao, I.M. (2006). Scope and strategies for regulation of nitrification in agricultural systems—challenges and opportunities. *Crit. Rev. Plant Sci.* 25, 303–335.
135. Bo, Y., Jägermeyr, J., Yin, Z., Jiang, Y., Xu, J., Liang, H., and Zhou, F. (2022). Global benefits of non-continuous flooding to reduce greenhouse gases and irrigation water use without rice yield penalty. *Global Change Biol.* 28, 3636–3650.
136. Cayuela, M.L., Aguilera, E., Sanz-Cobena, A., Adams, D.C., Abalos, D., Barton, L., Ryals, R., Silver, W.L., Alfaro, M.A., Pappa, V.A., et al. (2017). Direct nitrous oxide emissions in Mediterranean climate cropping systems: Emission factors based on a meta-analysis of available measurement data. *Agric. Ecosyst. Environ.* 238, 25–35.
137. Kennedy, T.L., Suddick, E.C., and Six, J. (2013). Reduced nitrous oxide emissions and increased yields in California tomato cropping systems under drip irrigation and fertigation. *Agric. Ecosyst. Environ.* 170, 16–27.
138. Guardia, G., Cangani, M.T., Sanz-Cobena, A., Junior, J.L., and Vallejo, A. (2017). Management of pig manure to mitigate NO and yield-scaled N₂O emissions in an irrigated Mediterranean crop. *Agric. Ecosyst. Environ.* 238, 55–66.
139. Sánchez-Martin, L., Arce, A., Benito, A., Garcia-Torres, L., and Vallejo, A. (2008). Influence of drip and furrow irrigation systems on nitrogen oxide emissions from a horticultural crop. *Soil Biol. Biochem.* 40, 1698–1706.
140. International Commission on Irrigation and Drainage (ICID) (2020). Sprinkler and Micro Irrigated area. <https://www.icid.org/sprinklerandmicro.pdf>.
141. Hedenus, F., Wirsén, S., and Johansson, D.J.A. (2014). The importance of reduced meat and dairy consumption for meeting stringent climate change targets. *Climatic Change* 124, 79–91.
142. Theurl, M.C., Lauk, C., Kalt, G., Mayer, A., Kaltenegger, K., Morais, T.G., Teixeira, R.F.M., Domingos, T., Winiwarter, W., Erb, K.-H., and Haberl, H. (2020). Food systems in a zero-deforestation world: Dietary change is more important than intensification for climate targets in 2050. *Sci. Total Environ.* 735, 139353.
143. Navarre, N., Schrama, M., de Vos, C., and Mogollón, J.M. (2023). Interventions for sourcing EAT-Lancet diets within national agricultural areas: A global analysis. *One Earth* 6, 31–40.
144. Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M.C., Thornton, P.K., Blümmel, M., Weiss, F., Grace, D., and Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proc. Natl. Acad. Sci. USA* 110, 20888–20893.
145. Xu, P., Houlton, B.Z., Zheng, Y., Zhou, F., Ma, L., Li, B., Liu, X., Li, G., Lu, H., Quan, F., et al. (2022). Policy-enabled stabilization of nitrous oxide emissions from livestock production in China over 1978–2017. *Nat. Food* 3, 356–366.
146. Herrero, M., Henderson, B., Havlík, P., Thornton, P.K., Conant, R.T., Smith, P., Wirsén, S., Hristov, A.N., Gerber, P., Gill, M., et al. (2016). Greenhouse gas mitigation potentials in the livestock sector. *Nat. Clim. Change* 6, 452–461.
147. Davidson, E.A. (2012). Representative concentration pathways and mitigation scenarios for nitrous oxide. *Environ. Res. Lett.* 7, 024005.
148. Kim, B.F., Santo, R.E., Scatterday, A.P., Fry, J.P., Synk, C.M., Cebron, S.R., Mekonnen, M.M., Hoekstra, A.Y., de Pee, S., Bloem, M.W., et al. (2020). Country-specific dietary shifts to mitigate climate and water crises. *Global Environ. Change* 62, 101926.
149. Leach, A.M., Galloway, J.N., Bleeker, A., Erisman, J.W., Kohn, R., and Kitzes, J. (2012). A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. *Environmental Development* 1, 40–66.
150. Bouwman, L., Goldewijk, K.K., Van Der Hoek, K.W., Beusen, A.H., Van Vuuren, D.P., Willems, J., Rufino, M.C., and Stehfest, E. (2013). Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. *Proc. Natl. Acad. Sci. USA* 110, 20882–20887.
151. Ritchie, H., Reay, D.S., and Higgins, P. (2018). The impact of global dietary guidelines on climate change. *Global Environ. Change* 49, 46–55.
152. Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., et al. (2019). Food in the Anthropocene: the EAT-Lancet Commission on healthy diets from sustainable food systems. *Lancet* 393, 447–492.
153. Davidson, E.A., and Kanter, D. (2014). Inventories and scenarios of nitrous oxide emissions. *Environ. Res. Lett.* 9, 105012.
154. Oenema, O., Ju, X., Klein, C., Alfaro, M., del Prado, A., Lesschen, J.P., Zheng, X., Velthof, G., Ma, L., and Gao, B. (2013). Reducing N₂O emissions from agricultural sources. In *Drawing Down N₂O to Protect Climate and the Ozone Layer: A UNEP Synthesis Report*, pp. 17–25.
155. Sun, Z., Sängler, A., Rebensburg, P., Lentzsch, P., Wirth, S., Kaupenjohann, M., and Meyer-Aurich, A. (2017). Contrasting effects of biochar on N₂O emission and N uptake at different N fertilizer levels on a temperate sandy loam. *Sci. Total Environ.* 578, 557–565.
156. Shen, H., Shiratori, Y., Ohta, S., Masuda, Y., Isobe, K., and Senoo, K. (2021). Mitigating N₂O emissions from agricultural soils with fungivorous mites. *ISME J.* 15, 2427–2439.
157. Li, Y., Zheng, X., Fu, X., and Wu, Y. (2016). Is green tea still 'green'. *Geography and Environment* 3, e00021.
158. Wang, Y., Yao, Z., Pan, Z., Wang, R., Yan, G., Liu, C., Su, Y., Zheng, X., and Butterbach-Bahl, K. (2020). Tea-planted soils as global hotspots for N₂O emissions from croplands. *Environ. Res. Lett.* 15, 104018.
159. Qasim, W., Xia, L., Lin, S., Wan, L., Zhao, Y., and Butterbach-Bahl, K. (2021). Global greenhouse vegetable production systems are hotspots of soil N₂O emissions and nitrogen leaching: A meta-analysis. *Environ. Pollut.* 272, 116372.
160. Shang, Z., Abdalla, M., Kuhnert, M., Albanito, F., Zhou, F., Xia, L., and Smith, P. (2020). Measurement of N₂O emissions over the whole year is necessary for estimating reliable emission factors. *Environ. Pollut.* 259, 113864.
161. Pelster, D.E., Thiagarajan, A., Liang, C., Chantigny, M.H., Wagner-Riddle, C., Congreves, K., Lemke, R., Glenn, A., Tenuta, M., Hernandez-Ramirez, G., et al. (2023). Ratio of non-growing season to growing season N₂O emissions in Canadian croplands: an update to national inventory methodology. *Can. J. Soil Sci.* 103, 344–352.
162. Wang, Y., Hu, Z., Islam, A.R.M.T., Chen, S., Shang, D., and Xue, Y. (2019). Effect of Warming and Elevated O₃ Concentration on CO₂ Emissions in a Wheat-Soybean Rotation Cropland. *Int. J. Environ. Res. Publ. Health* 16, 1755.
163. Wang, Y., Hu, Z., Liu, C., Islam, A.R.M.T., Chen, S., Zhang, X., and Zhou, Y. (2021). Responses of CO₂ and N₂O emissions from soil-plant systems to simulated warming and acid rain in cropland. *J. Soils Sediments* 21, 1109–1126.
164. Jiang, J., Hu, Z., Sun, W., and Huang, Y. (2010). Nitrous oxide emissions from Chinese cropland fertilized with a range of slow-release nitrogen compounds. *Agric. Ecosyst. Environ.* 135, 216–225.
165. Yao, Z., Zhou, Z., Zheng, X., Xie, B., Mei, B., Wang, R., Butterbach-Bahl, K., and Zhu, J. (2010). Effects of organic matter incorporation on nitrous oxide emissions from rice-wheat rotation ecosystems in China. *Plant Soil* 327, 315–330.
166. Wu, D., Senbayram, M., Well, R., Brüggemann, N., Pfeiffer, B., Loick, N., Stempfhuber, B., Dittert, K., and Bol, R. (2017). Nitrification inhibitors mitigate N₂O emissions more effectively under straw-induced conditions favoring denitrification. *Soil Biol. Biochem.* 104, 197–207.
167. Ren, X., Zhu, B., Bah, H., and Raza, S.T. (2020). How Tillage and Fertilization Influence Soil N₂O Emissions after Forestland Conversion to Cropland. *Sustainability* 12, 7947.
168. Lee, S.I., Park, H.J., Jeong, Y.J., Seo, B.S., Kwak, J.H., Yang, H.I., Xu, X., Tang, S., Cheng, W., Lim, S.S., and Choi, W.J. (2021). Biochar-induced reduction of N₂O emission from East Asian soils under aerobic conditions: Review and data analysis. *Environ. Pollut.* 291, 118154.
169. Zhang, X., Zou, T., Lassaletta, L., Mueller, N.D., Tubiello, F.N., Lisk, M.D., Lu, C., Conant, R.T., Dorich, C.D., Gerber, J., et al. (2021). Quantification of global and national nitrogen budgets for crop production. *Nat. Food* 2, 529–540.
170. Kaltenegger, K., and Winiwarter, W. (2020). Global gridded nitrogen indicators: Influence of crop maps. *Global Biogeochem. Cycles* 34, e2020GB006634.
171. Li, Y., and Ju, X. (2020). Rational nitrogen application is the key to mitigate agricultural nitrous oxide emission. *Journal of Agro-Environment Science* 39, 842–851.
172. Lam, S.K., Suter, H., Mosier, A.R., and Chen, D. (2017). Using nitrification inhibitors to mitigate agricultural N₂O emission: a double-edged sword? *Global Change Biol.* 23, 485–489.
173. Ke, J., He, R., Hou, P., Ding, C., Ding, Y., Wang, S., Liu, Z., Tang, S., Ding, C., Chen, L., and Li, G. (2018). Combined controlled-released nitrogen fertilizers and deep placement effects of N leaching, rice yield and N recovery in machine-transplanted rice. *Agric. Ecosyst. Environ.* 265, 402–412.
174. Rychel, K., Meurer, K.H.E., Börjesson, G., Strömberg, M., Getahun, G.T., Kirchmann, H., and Kätterer, T. (2020). Deep N fertilizer placement

- mitigated N₂O emissions in a Swedish field trial with cereals. *Nutrient Cycl. Agroecosyst.* **118**, 133–148.
175. Zhang, C., Song, X., Zhang, Y., Wang, D., Rees, R.M., and Ju, X. (2022). Using nitrification inhibitors and deep placement to tackle the trade-offs between NH₃ and N₂O emissions in global croplands. *Global Change Biol.* **28**, 4409–4422.
 176. Lyu, H., Zhang, H., Chu, M., Zhang, C., Tang, J., Chang, S.X., Masek, O., and Ok, Y.S. (2022). Biochar Affects Greenhouse Gas Emissions in Various Environments: A Critical Review. *Land Degradation & Development* **33**, 3327–3342.
 177. Fujimori, S., Wu, W., Doelman, J., Frank, S., Hristov, J., Kyle, P., Sands, R., van Zeist, W.-J., Havlik, P., Dominguez, I.P., et al. (2022). Land-based climate change mitigation measures can affect agricultural markets and food security. *Nat. Food* **3**, 110–121.
 178. Yao, G., Zhang, X., Davidson, E.A., and Taheripour, F. (2021). The increasing global environmental consequences of a weakening US-China crop trade relationship. *Nat. Food* **2**, 578–586.
 179. Sutton, M.A., Oenema, O., Erisman, J.W., Leip, A., van Grinsven, H., and Winiwarter, W. (2011). Too much of a good thing. *Nature* **472**, 159–161.
 180. Xu, Y., Wang, R., Tong, Y., Zhao, H., Xie, Q., Liu, D., Zhang, A., Li, B., Xu, H., and An, D. (2014). Mapping QTLs for yield and nitrogen-related traits in wheat: influence of nitrogen and phosphorus fertilization on QTL expression. *Theor. Appl. Genet.* **127**, 59–72.
 181. Zebbarth, B.J., Tai, H., Luo, S., Millard, P., De Koeyer, D., Li, X.-Q., and Xiong, X. (2011). Differential gene expression as an indicator of nitrogen sufficiency in field-grown potato plants. *Plant Soil* **345**, 387–400.
 182. Bakken, L.R., Bergaust, L., Liu, B., and Frostegård, A. (2012). Regulation of denitrification at the cellular level: a clue to the understanding of N₂O emissions from soils. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **367**, 1226–1234.
 183. Itakura, M., Uchida, Y., Akiyama, H., Hoshino, Y.T., Shimomura, Y., Morimoto, S., Tago, K., Wang, Y., Hayakawa, C., Uetake, Y., et al. (2013). Mitigation of nitrous oxide emissions from soils by *Bradyrhizobium japonicum* inoculation. *Nat. Clim. Change* **3**, 208–212.
 184. Gregorich, E., Janzen, H.H., Helgason, B., and Ellert, B. (2015). Nitrogenous gas emissions from soils and greenhouse gas effects. In *Advances in Agronomy*, pp. 39–74.
 185. Richardson, D., Felgate, H., Watmough, N., Thomson, A., and Baggs, E. (2009). Mitigating release of the potent greenhouse gas N₂O from the nitrogen cycle – could enzymic regulation hold the key? *Trends Biotechnol.* **27**, 388–397.
 186. Löw, P., Osterburg, B., and Klages, S. (2021). Comparison of regulatory approaches for determining application limits for nitrogen fertilizer use in Germany. *Environ. Res. Lett.* **16**, 055009.
 187. Kanter, D.R., Chodos, O., Nordland, O., Rutigliano, M., and Winiwarter, W. (2020). Gaps and opportunities in nitrogen pollution policies around the world. *Nat. Sustain.* **3**, 956–963.
 188. Kanter, D.R., Ogle, S.M., and Winiwarter, W. (2020). Building on Paris: integrating nitrous oxide mitigation into future climate policy. *Curr. Opin. Environ. Sustain.* **47**, 7–12.
 189. Kanter, D., Mauzerall, D.L., Ravishankara, A.R., Daniel, J.S., Portmann, R.W., Grabel, P.M., Moomaw, W.R., and Galloway, J.N. (2013). A post-Kyoto partner: Considering the stratospheric ozone regime as a tool to manage nitrous oxide. *Proc. Natl. Acad. Sci. USA* **110**, 4451–4457.
 190. Sutton, M.A., Howard, C.M., Kanter, D.R., Lassaletta, L., Möring, A., Raghuram, N., and Read, N. (2021). The nitrogen decade: mobilizing global action on nitrogen to 2030 and beyond. *One Earth* **4**, 10–14.
 191. Fischer G., Nachtergaele F., Prieler S., Van Velthuisen H.T., Verelst L., Wiberger D. *Global Agro-Ecological Zones Assessment for Agriculture (GAEZ 2008)*. Rome, Italy: FAO; Laxenburg, Austria: IIASA; 2008.
 192. Harris, I., Osborn, T.J., Jones, P., and Lister, D. (2020). Version 4 of the CRU TS monthly high-resolution gridded multivariate climate dataset. *Sci. Data* **7**, 109.
 193. Sacks, W.J., Deryng, D., Foley, J.A., and Ramankutty, N. (2010). Crop planting dates: an analysis of global patterns. *Global Ecol. Biogeogr.* **19**, 607–620.
 194. Porwollik, V., Rolinski, S., Heinke, J., and Müller, C. (2019). Generating a rule-based global gridded tillage dataset. *Earth Syst. Sci. Data* **11**, 823–843.
 195. Klein Goldewijk, K., Beusen, A., Doelman, J., and Stehfest, E. (2017). Anthropogenic land use estimates for the Holocene - HYDE 3.2. *Earth Syst. Sci. Data* **9**, 927–953.
 196. Portmann, F.T., Siebert, S., and Döll, P. (2010). MIRCA2000-Global monthly irrigated and rainfed crop areas around the year 2000: A new high-resolution data set for agricultural and hydrological modeling. *Global Biogeochem. Cycles* **24**, Gb1011.
 197. Hegglin, M., Kinnison, D., and Lamarque, J.-F. (2019). input4MIPs.C-MIP6.ScenarioMIP.NCAR.NCAR-CCMI-ssp119-1-0 (Earth System Grid Federation).