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REVIEW

GASEOUS REACTIVE NITROGEN LOSSES FROM ORCHARDS, VEGETABLES AND TEA PLANTATIONS

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KEYWORDS

fruit, greenhouse gas, green development, fertilizer management, climate change

HIGHLIGHTS

- Gaseous N emissions from orchards, vegetables and tea plantations (OVT) are reviewed.
- Gaseous N emissions from OVT are greater in China than the rest of the world.
- OVT are hotspots for gaseous N emissions from the agricultural sector in China.

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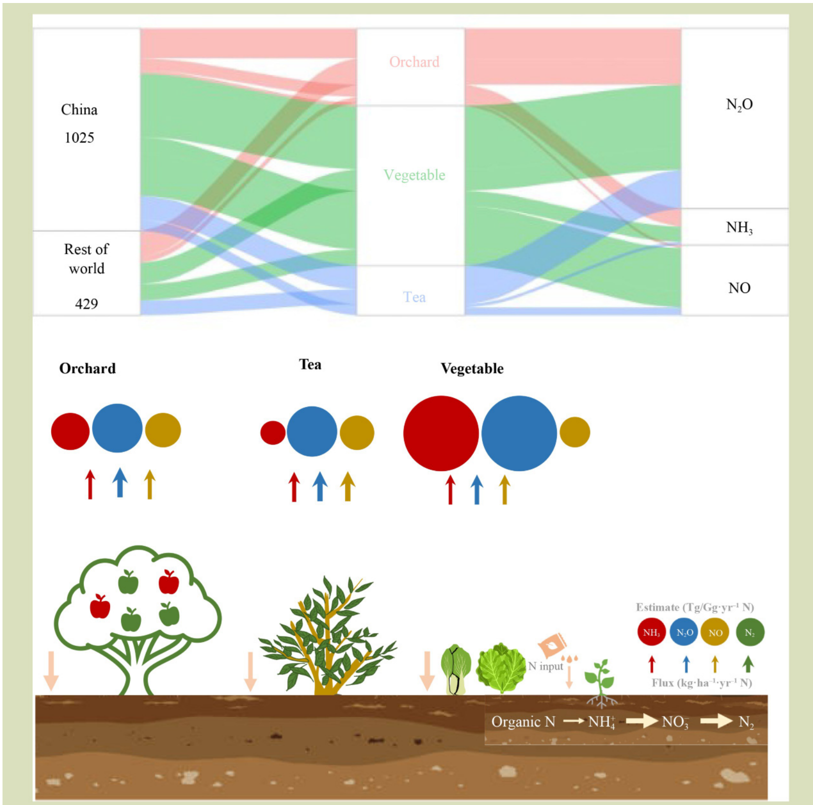
GRAPHICAL ABSTRACT



ABSTRACT

Nitrogen fertilizer application has accelerated the agricultural soil N cycle while ensuring food security. Gaseous reactive N emissions from orchards, vegetables and tea plantations (OVT) are less understood than those from cereal crops. This paper presents a compilation of data on soil ammonia, nitrous oxide, and nitric oxide emissions from 1454 OVT systems at 184 unique experimental locations worldwide aiming to investigate their emission characteristics, emission factors (EF), and contribution to total farmland emissions. NH_3 and N_2O emissions from orchards and N_2O and NO emissions from vegetable production were significantly higher in China than in the rest of the world, regardless of fertilizer application, while N_2O emissions from tea plantations were lower than for vegetables. The EF of NH_3 for vegetables was close to the global mean value with urea application but significantly higher than that of orchards. The EF of N_2O in orchards and vegetables was comparable to the global median value, while in tea plantations, the value was 2.3 times higher than the global median value. Current estimates suggest that direct emissions of NH_3, N_2O, and NO from OVT systems are equivalent to approximately a quarter, two thirds and a half of the total farmland in China, respectively. Future research needs to strengthen observational field studies in establishing standard sampling methods for gaseous N emissions and implementing knowledge-based management measures to help achieve the green development of agriculture.

1 INTRODUCTION

Increasing anthropogenic-induced reactive N concentrations have accelerated the global N cycle[1]. During the last few decades, anthropogenic processes such as Haber-Bosch industrial processes, biological N fixation from agricultural cultivation and fossil fuel combustion have caused reactive N production comparable to natural processes, such as lightning and biological N fixation[2]. Total reactive N inputs to terrestrial ecosystems have increased significantly over the last century, with synthetic N fertilizer inputs dominating in agroecosystems[3]. The total global production of anthropogenic reactive N is estimated to be 210 $Tg \cdot yr^{-1}$, of which fertilizer production accounts for about 48%[2,4]. The application of large amounts of chemical fertilizers in agriculture ensures food production but also inevitably has a negative impact on the global environment by increasing reactive N emissions and exacerbating surface source pollution[5].

Gaseous reactive N losses in the form of NH_3 volatilization and nitrogen oxide emissions are the main pathways for N losses from agricultural fields. NH_3 volatilization is one of the main pathways for N losses from agricultural fields in China[6]. It is estimated that ammonia emissions from agricultural sources in China account for about 90% of the total national emissions, with approximately equal contributions from planting and livestock farming emissions[7]. NH_3 volatilization from agricultural soils mainly comes from the ammonification process of urea hydrolysis and nitrate isomerization reduction to ammonium, which is influenced by various factors such as temperature, moisture, crop type, fertilizer application and fertilization practices, and soil properties. Increased ammonia emissions due to excessive or inappropriate application of N fertilizers not only reduce N use efficiency, but also contribute to problems such as haze, dry and wet atmospheric deposition, and indirect greenhouse gas emissions[4]. In addition, as two trace gases of concern, N_2O and NO emissions are directly or indirectly involved in global warming with adverse effects on human health and ecosystem function[8,9]. Agricultural soils are the main anthropogenic source of N_2O and NO emissions, as N fertilizers applied in agricultural production are essential to increase the substrate for nitrification and denitrification processes[10–12]. In 2007–2016, N_2O emissions from anthropogenic sources averaged 43% of total emissions (7.3 $Tg \cdot yr^{-1}$ N), with direct and indirect emissions from N inputs in agriculture accounting for about 52%[13]. NO emissions from N fertilizer application in agriculture are estimated to be 3.7 $Tg \cdot yr^{-1}$ N, which is about 10% of the global total[9]. Therefore, the use of rational fertilization techniques to improve N use efficiency is a fundamental way to reduce gaseous active N losses from agricultural fields.

Over the past three decades, the changing cropping structure of arable land in China has been driven by the expanding area under cash crops, including orchards, vegetables and tea plantations (OVT). The area planted with cereals increased by 14%, while the cultivated areas of OVT increased by 1.2, 2.3, and 2.2 times, respectively (Fig. 1(a)). Studies based on national research questionnaires showed that the average N fertilizer application rate under farmer practices in Chinese OVT was 592, 439 and 491 $kg \cdot ha^{-1}$ N, respectively, higher than that of major cereal crops such as rice (224 $kg \cdot ha^{-1}$ N), wheat (212 $kg \cdot ha^{-1}$ N) and maize (249 $kg \cdot ha^{-1}$ N)[14–16]. High N inputs for cash crop cultivation result in N surpluses about 4-fold higher than those for cereals[15]. A high N surplus usually means an increased potential loss of N from agricultural soils to the environment, leading to a lower N use efficiency. A growing body of experimental evidence indicates that the intensities of gaseous reactive N emissions, especially N_2O and NO, from soils grown in OVT are generally greater than from cereal cropping systems[17–24]. Therefore, a better

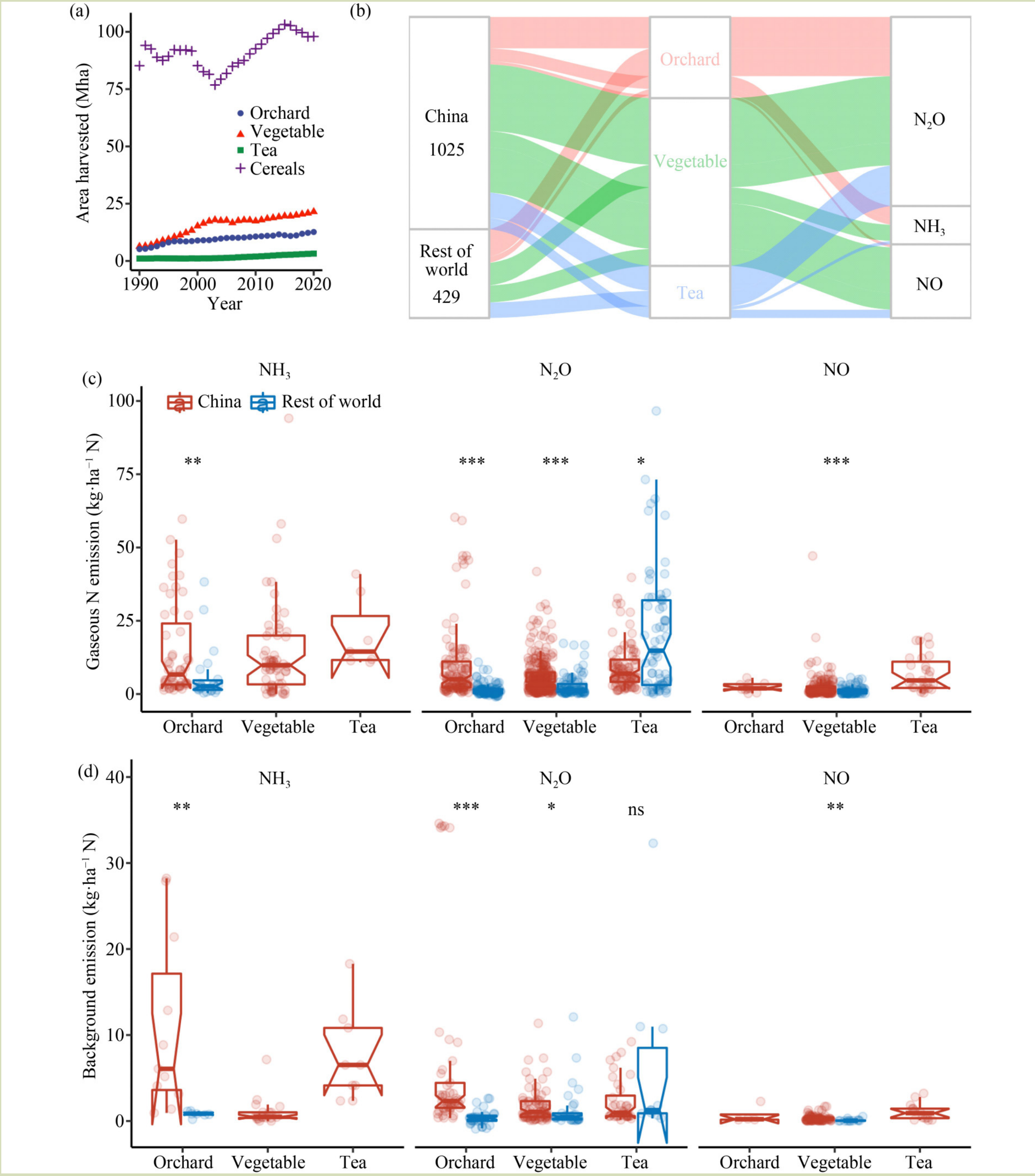


Fig. 1 Gaseous reactive N losses from orchards, vegetables and tea plantations (OVT). (a) Harvested area of OVT in China during 1990–2020. (b) Distribution of experimental observations for measuring gaseous reactive N emissions from OVT. (c) Comparison of gaseous reactive N emissions between cropping systems or regions under fertilization conditions. (d) Comparison of background gaseous reactive N emissions (unfertilized plots) between cropping systems or regions. Asterisks indicate results of non-parametric Wilcoxon signed-rank tests. ns, $P > 0.05$; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$.

understanding of the characteristics of gaseous reactive N emissions from intensive cash crop cultivation and its contribution to the agricultural sector is an essential step toward the green development of agriculture.

In this analysis, we aimed to systematically analyze the intensity, direct emission factors, and total amount of soil gaseous reactive N emissions from OVT systems and compared their differences between China and other regions of the world. To accomplish this, we extracted observations of gaseous N emissions from OVT systems from the databases of the synthesis and meta-analysis studies and established a data set. We focused on answering the following questions. (1) What is the intensity of gaseous reactive N emissions from OVT systems and how do they differ between regions and crop types? (2) What are the fertilizer-induced direct emission factors (EF) for these systems? (3) What is the contribution of gaseous reactive N emissions from OVT systems to croplands in China and the world? Finally, we also briefly highlight some issues that need to be addressed for future research in OVT systems.

2 MATERIALS AND METHODS

We collected data on NH_3, N_2O, and NO emissions from OVT systems reported in synthesis and meta-analysis studies. Data on NH_3 and N_2O emissions from orchards were obtained from two recent synthesis studies[20,25]. Data on NH_3 emissions from orchards and vegetables were obtained from a database used for an empirical model analysis[26]. Emissions of N_2O from tea plantations and NO from OVT systems were obtained from our own database[27] and a recent synthesis study[18], respectively. Available databases for NH_3 emissions from vegetable and tea plantations and NO emissions from orchards and tea plantations are only available from China. In these databases, we extracted the following information for each study: coordinates of the test site, crop type, the amount and type of N fertilizer applied and total gaseous N emissions during the test observation period. We excluded some data containing treatments that affected gaseous N emissions, namely, slow-release fertilizers, urease or nitrification inhibitors, biochar and bioorganic fertilizers. The gaseous N emissions data were divided into emissions from fertilized treatments and background emissions (i.e., unfertilized plots) according to whether N fertilizer was applied. We obtained 1454 observations including 185 for NH_3, 913 for N_2O and 356 for NO.

To compare the differences in gaseous N emissions due to fertilizer application in OVT systems, we collected EF data reported in the literature. These EF values were divided into two categories corresponding to the Tier 1 and Tier 3 methods in the IPCC guidelines for national GHG inventories. Specifically, the Tier 1 approach is based on the arithmetic mean of EFs from multiple independent studies, while Tier 3 is an empirical model that fits the relationship between N fertilizer inputs as a function of gaseous N emissions. We also collected the results of existing studies on estimating total gaseous N emissions from OVT systems. Data on the area cultivated of OVT systems in China from 1990 to 2020 were obtained from the National Statistical Yearbook[28].

3 STATISTICAL ANALYSIS

We used the non-parametric Wilcoxon signed-rank test to test for differences in gaseous N emissions across regions or between crops. All statistical analyses and plotting were performed in R software[29].

4 RESULTS AND DISCUSSION

4.1 Background emissions and fertilization-induced gaseous N emissions

We compiled observational data on gaseous N emissions in OVT systems over the past four decades. Our database contains 1454 observations from 184 unique experimental locations worldwide, of which 70% are from China and the rest from other regions of the world (Fig. 1(b)). Among these observations, OVT accounted for 27%, 55% and 18%, respectively. In terms of gas type, the dominant gas type was N_2O (63%), followed by NO (24%) and NH_3 (13%). Thus, these findings suggest that China is a hotspot for *in situ* field observations for these cash crop cropping systems. These studies not only help to understand the characteristics and drivers of gaseous N emissions in diverse cropping systems but also provide a basis for understanding the impact of increasing anthropogenic reactive N on climate and air quality and developing abatement strategies.

The magnitude of gaseous N emissions under fertilization conditions varied significantly among gas types, crops and regions (Fig. 1(c)). Overall, the magnitude of gaseous N emissions due to fertilizer application was ranked as NH_3 > N_2O > NO. The median NH_3 and N_2O emissions from Chinese orchards were 6.7 $kg \cdot ha^{-1}$ N (mean ± SD: 14.6 ± 15.9 $kg \cdot ha^{-1}$ N) and 5.0 $kg \cdot ha^{-1}$ N (9.7 ± 12.9 $kg \cdot ha^{-1}$ N), respectively, significantly higher than the corresponding emissions of 2.7 $kg \cdot ha^{-1}$ N (6.2 ± 9.6 $kg \cdot ha^{-1}$ N; Wilcoxon signed-rank test, $P < 0.01$) and 0.9 $kg \cdot ha^{-1}$ N (1.3 ± 1.9 $kg \cdot ha^{-1}$ N; $P < 0.001$) in the rest of the world. Similarly, the median N_2O and NO emissions from Chinese vegetable fields were 3.6 $kg \cdot ha^{-1}$ N (5.8 ± 6.3 $kg \cdot ha^{-1}$ N) and 1.3 $kg \cdot ha^{-1}$ N

(2.4 ± 4.5 kg·ha^{-1} N), significantly greater than the corresponding emissions of 1.2 kg·ha^{-1} N (2.9 ± 3.8 kg·ha^{-1} N; $P < 0.001$) and 0.65 kg·ha^{-1} N (1.2 ± 1.3 kg·ha^{-1} N; $P < 0.001$) in the rest of the world. These results suggest that Chinese orchards and vegetable cultivation are global hotspots for gaseous N emissions from the agricultural sector[20,30]. This is mainly attributed to the differences in agronomic management practices such as fertilizer application, irrigation, and tillage among different regions. For example, in our data set the average N application to orchards and vegetables in China (median: 393 and 313 kg·ha^{-1} N, respectively) was significantly higher than in the rest of the world (80 and 220 kg·ha^{-1} N, respectively; $P < 0.001$). Water-efficient irrigation (e.g., drip and sprinkler irrigation) can reduce gaseous N emissions from intensive cropping systems compared to furrow and flood irrigation[31]. In contrast, the median and mean values of N_2O emissions from tea plantations in China were 7.0 and 9.4 (± 8.0) kg·ha^{-1} N, respectively, which were significantly lower than those of 14.8 and 20.6 (±21.1) kg·ha^{-1} N, respectively, in the rest of the world ($P < 0.05$). We attribute this difference mainly to the fact that N fertilizer application in tea plantations was lower in China than in the rest of the world (450 vs 514 kg·ha^{-1} N; $P < 0.001$). In addition, tea plantation soils in other regions of the world were more acidic compared to China and thus tended to emit more N_2O[32,33]. For China, NH_3 emissions did not differ among the three cropping systems, while N_2O emissions showed a pattern of tea plantations > orchards > vegetables ($P < 0.05$). Importantly, we found that the magnitude of NO emissions was significantly higher in tea plantations than in the other two cropping systems ($P < 0.05$), which may be due to the acidic soil environment in tea plantations being more conducive to the occurrence of chemodenitrification with NO as a product[34].

Similarly, under unfertilized conditions, gaseous N emissions varied among gas types, crops, and regions (Fig. 1(d)). The median background emissions of NH_3 and N_2O in Chinese orchards were 6.1 kg·ha^{-1} N (10.9 ± 10.3 kg·ha^{-1} N) and 2.3 kg·ha^{-1} N (6.4 ± 10.2 kg·ha^{-1} N), significantly higher than the corresponding emissions of 0.8 kg·ha^{-1} N (0.8 ± 0.3 kg·ha^{-1} N; $P < 0.01$) and 0.3 kg·ha^{-1} N (0.4 ± 0.9 kg·ha^{-1} N; $P < 0.001$) in the rest of the world. The median background emissions of N_2O and NO from Chinese vegetable production were 1.1 kg·ha^{-1} N (1.8 ± 2.1 kg·ha^{-1} N) and 0.2 kg·ha^{-1} N (0.3 ± 0.4 kg·ha^{-1} N), which were significantly higher than the corresponding emissions of 0.5 kg·ha^{-1} N (1.5 ± 2.6 kg·ha^{-1} N; $P < 0.05$) and 0.04 kg·ha^{-1} N (0.1 ± 0.2 kg·ha^{-1} N; $P < 0.01$) in the rest of the world. However, background N_2O emissions from tea plantations did not differ between China and the rest of the world (median: 1.0 vs 1.3 kg·ha^{-1} N). For China, we found significantly higher NH_3 background emissions from orchards and tea plantations than from vegetable fields ($P < 0.001$). In comparison, N_2O background emissions from orchards were significantly higher than from the other two cropping systems ($P < 0.01$). NO background emissions from tea plantations were significantly higher than those from vegetables ($P < 0.01$) but not different from those from orchards. Overall, we found that the magnitude of background emissions of gaseous reactive N was generally greater in Chinese OVT systems than in upland cereals[35].

4.2 EFs of reactive gaseous N

We summarized and compared the EFs of gaseous N among different cropping systems and between China and the world (Table 1). Three tier methods are included in the national GHG inventory guidelines prepared by the IPCC[48]. Specifically, the default EFs in the Tier 1 approach is calculated by aggregating a large number of EFs from independent observational studies, which are obtained by dividing the difference in emissions of a given gas between fertilized and unfertilized treatments by the total N input. The EFs used in the Tier 2 approach is more refined EFs, such as region- or country-specific, crop-specific, and fertilizer-specific EF. The EFs in this tier approach are the EFs corresponding to different crops in different regions in our case. EFs in the Tier 3 approach are generally obtained based on empirical models or process-based model simulations, as in this review the EFs for this tier approach are all obtained using empirical models.

The EF of different gaseous N varied significantly across crops and regions (Table 1). The mean EF of N_2O emission in Chinese orchards was 1.2% (range: 0.7%–1.72%), which was slightly lower than the world average of 1.4% (0.84%–1.86%)[20,25,37]. The mean EF of N_2O emission from Chinese vegetable fields was 0.85% (0.55%–1.7%), and notably lower than the world average of 1.4% (0.93%–2.4%)[17,35,38–42,44,45]. The EF of N_2O in orchards and vegetables and its variability were comparable with the global mean (1%, range: 0.1%–1.8%)[48]. In contrast, the respective mean EF values of 2.3% and 2.4% for N_2O emission in Chinese and global tea plantations were significantly higher than the other two cropping systems and the global mean values[46,47]. This disparity is mainly due to the fact that tea plantations are generally grown in subtropical and tropical regions, and their optimal soil pH range is 4.5–5.5, which are environmental conditions that favor soils with high N_2O emissions[49].

Unlike N_2O, the EF data available to date for NO and NH_3 emissions are relatively sparse (Table 1). To the best of our

Table 1 Summary of emission factor (EF) of gaseous N loss (N_2O, NO, and NH_3) from orchards, vegetables, and tea plantations

Plant type	Emission	Region	EF (%)	IPCC method	Reference
Orchard	N_2O	China	0.70	Tier 3	[36]
			1.08	Tier 1	[20]
			1.72	Tier 3	
		World	0.84	Tier 3	
			1.19	Tier 1	
			1.36	Tier 3	[37]
			1.39	Tier 1	[36]
			1.62	Tier 3	
			1.86	Tier 3	[20]
Vegetable	N_2O	China	0.55	Tier 3	[38]
			0.63	Tier 3	[39]
			0.69	Tier 1	[40]
			0.69	Tier 3	[41]
			0.73	Tier 1	
			0.86	Tier 1	[42]
			0.95	Tier 3	
			1.69	Tier 1	[35]
		World	0.93	Tier 3	[43]
			0.94	Tier 1	[44]
			1.41	Tier 1	[45]
			1.50	Tier 3	
			2.42	Tier 1	[17]
Tea	N_2O	China	1.89	Tier 3	[27]
			2.19	Tier 1	
			2.72	Tier 1	[46]
		World	1.81	Tier 3	[27]
			2.31	Tier 1	[47]
			2.31	Tier 1	[27]
Vegetable	NO	China	0.87	Tier 1	
			1.26	Tier 1	[35]
		World	0.75	Tier 1	[18]
			1.71	Tier 1	[17]
Orchard	NO	China	0.42	Tier 1	[18]
		World	0.42	Tier 1	
Tea	NO	China	1.54	Tier 1	
Orchard	NH_3	World	3.64	Tier 1	[36]
			5.22	Tier 3	
Vegetable	NH_3	China	11.60	Tier 1	[35]
			13.36	Tier 1	[30]
		World	13.34	Tier 1	

Note: A blank in the reference column indicates that the data in this row are from the same study as the above row.

knowledge, only Chinese studies have reported NO emissions from orchards and tea plantations and calculated their EFs, corresponding to mean EF values of 0.42 and 1.5%[18]. The EFs of NO emission from vegetables production in China and the world are very similar (1.1% vs 1.2%), although there are significant differences between studies[17,18,35]. The global EF for NH_3 emission from orchards was 4.4%, which was significantly lower than the comparable EFs for Chinese and world vegetables (12.5% and 13.3%)[30,35,36]. The lower EF of NH_3 emission from orchards may be associated with fertilization and irrigation practices, such as hole or furrow application and water-saving irrigation that tend to reduce NH_3 volatilization[31]. We found that the EF of NH_3 emission from vegetables production was comparable to the global mean proportion of NH_3 loss due to urea application (14.2%)[48]. Therefore, the higher or comparable EF of OVT systems compared to cereal crops implies the need to estimate emissions due to fertilizer application in these systems, as it is a prerequisite for the development of emission reduction strategies.

4.3 Estimation of gaseous N emissions due to fertilizer application

Existing studies have estimated gaseous N emissions due to fertilizer application in OVT systems in China and globally (Table 2). The total NH_3 emissions due to fertilizer application in Chinese OVT systems were about 1.4 $Tg\cdot yr^{-1}$ N, of which vegetable fields accounted for more than half and orchards accounted for the least[26,30,35]. The total NH_3 emission due to synthetic N fertilizer application in Chinese vegetable fields was estimated to be 0.52 $Tg\cdot yr^{-1}$ N using a uniform EF[30], which is lower than the empirical model-based estimate of 1.1 $Tg\cdot yr^{-1}$ N[26]. This discrepancy could be explained because the former included only synthetic N fertilizer application and the latter used high-resolution activity data. The large variation between total NH_3 emissions from non-cereal crops other than vegetables is indicative of the relatively small number of observations to date. The estimated annual mean NH_3 emission from Chinese farmland was 4.7 ± 2.1 $Tg\cdot yr^{-1}$ N[26]. Thus, current estimates suggest that direct NH_3 emissions from synthetic N fertilizer application in OVT systems account for about 23% of the total farmland in China (Fig. 2).

In the 2010s, the total direct N_2O emissions due to fertilizer application in Chinese vegetable cropping systems was about 62 $Gg\cdot yr^{-1}$ N but estimates varied significantly between studies, ranging from 50.4 to 74.7 $Gg\cdot yr^{-1}$ N (Table 2). This estimate is larger than that based on region-specific EFs (55 $Gg\cdot yr^{-1}$ N)[51], which may be mainly due to the increase in the area under vegetable cultivation over the past 20 years. Direct N_2O emissions from fertilizer application in Chinese vegetable production accounts for about two thirds of the global total[44]. Total direct N_2O emissions from fertilizer application in Chinese orchards and tea plantations are comparable in the 2010s with a mean value of about 41 $Gg\cdot yr^{-1}$ N. However, the lower estimate is because the authors estimated only the top two fruits in China, that is, apples and citrus[50]. Similarly, we found that direct N_2O emissions from Chinese tea plantations due to fertilizer application accounted for about two thirds of the global total. Overall, N_2O emissions from OVT accounted for 18%, 27% and 18% of total farmland emissions in China, respectively (Fig. 2).

To date, two studies have estimated direct NO emissions associated with fertilizer application in OVT systems in China and globally[18,35] (Table 2). Based on the corresponding EFs mentioned above, the direct NO emissions due to fertilizer application in China are 55.7 and 40.2 $Gg\cdot yr^{-1}$ N for vegetables and other cash crops, respectively, with NO emissions from Chinese vegetable production accounting for about two thirds of the global total. Apart from vegetables, NO emissions from other cash crops in China are equivalent to 40% of the global total emissions from orchards and tea plantations. The total NO emissions from vegetables and other cash crops in China are estimated to be about half of the national total[35]. Therefore, these estimations indicate that the OVT cropping system in China is a significant source of gaseous N emissions not only in China (Fig. 2) but also in the global agricultural sector.

5 PERSPECTIVES AND SUMMARY

Over the last three decades, many field observation studies have been conducted globally on gaseous N emissions from OVT systems, especially in China. Their findings have provided a basis for understanding the characteristics, magnitude, and influencing factors of gaseous N emissions from OVT systems. Based on these observations, different studies have also attempted to estimate direct gaseous N emissions due to fertilizer application in these cropping systems, but the differences and uncertainties between the estimates are large. More importantly, OVT systems differ from cereal crops in terms of agronomic management such as fertilizer, water and tillage during cropping, which in turn strongly influence gaseous N emissions. For this reason, to better understand the characteristics of gaseous N emissions from OVT systems and their contribution to emission reduction in the agricultural sector, and thus provide a

Table 2 Summary of estimates of gaseous reactive N emissions from orchards, vegetables and tea plantations in China and the world

Emission and plant type	Region	Period	Estimates	Reference
NH_3 ($Tg \cdot yr^{-1}$ N)				
Vegetable	China	2018	0.52 ± 0.05	[35]
		2014	0.63 ± 0.04	[30]
		2017	1.10	[26]
	World	2014	1.10 ± 0.16	[30]
Orchard	China	2017	0.31	[26]
Tea	China	2017	0.03	
Other cash crops	China	2018	0.07 ± 0.01	[35]
Other non-cereal crops	World	2014	1.38 ± 0.14	[30]
N_2O ($Gg \cdot yr^{-1}$ N)				
Vegetable	China	2018	74.7 ± 7.62	[35]
		2015	69.0	[40]
		2009	67.0	[38]
		2016	50.8 ± 11.6	[50]
		2008	35.6 ± 5.09	[39]
		1990s	55.0 (11.8–129)	[51]
	World	2010	95.0	[44]
Orchard	China	2016	25.8 ± 2.68	[50]
		2000s	41.0 (20.0–78.0)	[20]
Tea	China	2013	41.0	[46]
		2010s	41.6	[27]
	World	2018	84.0	[47]
		2018	57.0	
		2010s	46.5	[27]
NO ($Gg \cdot yr^{-1}$ N)				
Vegetable	China	2018	55.7 ± 12.9	[35]
Other cash crops	China	2018	40. ± 11.0	
Vegetable	World	2010	83.3 (50.5–130)	[18]
Orchard	World	2010	75.8 (42.6–110)	
Tea	World	2010	24.2 (16.7–32)	

Note: Values are mean ± SD or with 95% confidence interval in parentheses. A blank in the reference column indicates that the data in this row are from the same study as the above row.

scientific reference for non-CO_2 GHG emission reduction in agriculture, we propose that future research needs to focus on the following aspects.

Firstly, there is an urgent need to develop and establish standard sampling guidelines for active gaseous N emissions from OVT systems. In orchards and tea plantations, fertilizer application is usually applied in strips, holes or furrows, which will directly affect the sampling point layout and representativeness of gas collection using the static chamber method. For example, most previous observations in tea plantations were made at the location of fertilizer application between the rows, which inevitably leads to an overestimation of gaseous N emissions, since the intensity of soil emissions between the rows is remarkably higher than under the canopy. For vegetable production, the observed gaseous N emissions under greenhouse conditions are not necessarily representative of what is eventually released into the atmosphere. In this

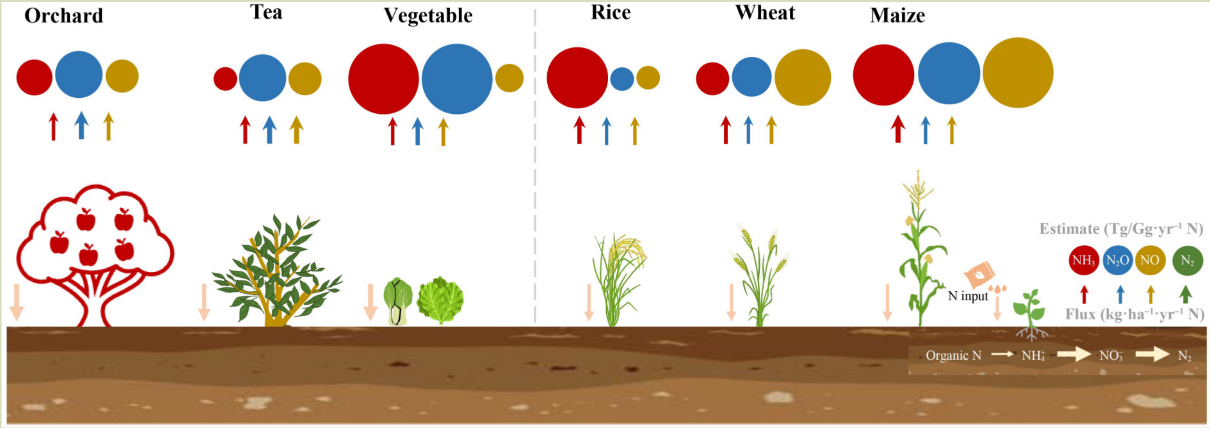


Fig. 2 Comparison of fertilizer application-induced reactive N emissions between orchard, vegetable, and tea plantation systems and cereal crops in China. Downward arrows indicate nitrogen fertilizer application. The three upward arrows in each group from left to right indicate the fluxes of NH_3, N_2O and NO emissions, respectively. The three circles in each group from left to right indicate total NH_3, N_2O and NO emissions, respectively. The arrow widths and circle sizes indicate the size of each gas when normalized across crops. For cereals, data on N application rate, emissions of N_2O and NH_3 are adapted from Chen et al.[16] and NO emissions are from Wang et al.[18]. Estimates of NH_3, N_2O and NO for cereals are adapted from Wang et al.[26], Aliyu et al.[40] and Ma et al.[35].

regard, it is unclear whether the differences between gaseous N losses from open-field and greenhouse cultivation reported in existing studies are indicative of actual differences.

Secondly, research is needed to assess the impact of optimized nutrient management and water efficient use practices on gaseous N emissions from OVT systems. The higher N fertilizer application in OVT systems than in cereal crops in existing studies is the major driver of higher gaseous N losses. Although the OVT cultivation area is smaller than other dryland crops, the higher emissions make them deserve priority for emission reduction. Therefore, future research needs to investigate the effects of optimized fertilization practices (e.g., 4R technology), the application of efficiency-enhancing N fertilizers (slow-release fertilizers, urease or nitrification inhibitors), and soil amendments (biochar and bioorganic fertilizers) on gaseous N emissions from OVT systems. Although available experimental evidence indicates that water-efficient irrigation can reduce soil N_2O emissions, this reduction effect is weakly represented for horticultural crops, and it remains to be clarified what effect water-efficient irrigation has on NH_3 and NO emissions.

Finally, more *in situ* observations in OVT systems are needed to improve data coverage globally. Evidently, the existing observational studies, regardless of crop type, have mostly been conducted in China, and only sporadically in other regions of the world, although this is partially attributed to differences in cash crop coverage in different countries. Climate conditions and soil properties are the key control factors affecting soil gaseous N emissions on a global scale. Therefore, it is necessary to conduct studies in other major growing regions of the world, thereby reducing the bias in their estimation based on data synthesis. In China, unlike vegetables, the observations of existing studies on orchards and tea plantations are mainly concentrated in a few provinces, such as the lack of observations on tea plantations in south-western and southern and orchards in north-western China.

In summary, we explored the gaseous N emissions, EF and emission estimates from Chinese and global OVT systems using existing research data sets. Our results show that NH_3 and N_2O emissions from Chinese orchards are notably higher than those from other regions of the world. NH_3 emissions from vegetable production in China accounted for more than half of the global total. Nevertheless, we also found that these estimates are still highly uncertain, which is mainly attributed to the low spatial coverage of available data making them less representative. Therefore, we recommend that more observational studies be conducted with broader coverage focusing on establishing standard sampling methods and assessing knowledge-based management measures. These studies will provide a scientific basis for developing strategies when implementing priority abatement in OVT systems.

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Compliance with ethics guidelines
Jinyang Wang, Pinshang Xu, Haiyan Lin, Shumin Guo, Zhaoqiang Han, and Jianwen Zou declare that they have no conflicts of interest or financial conflicts to disclose. This article does not contain any studies with human or animal subjects performed by any of the authors.

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