Review

Ground-level O₃ pollution and its impacts on food crops in China: A review Zhaozhong Feng^{a,*}, Enzhu Hu^a, Xiaoke Wang^a, Lijun Jiang^a, Xuejun Liu^b^a State Key Laboratory of Urban and Regional Ecology, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, Shuangqing Road 18, Haidian District, Beijing 100085, China^b College of Resources and Environmental Sciences, China Agricultural University, Beijing 100193, China

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ABSTRACT

Ground-level ozone (O₃) pollution has become one of the top environmental issues in China, especially in those economically vibrant and densely populated regions. In this paper, we reviewed studies on the O₃ concentration observation and O₃ effects on food crops throughout China. Data from 118 O₃ monitoring sites reported in the literature show that the variability of O₃ concentration is a function of geographic location. The impacts of O₃ on food crops (wheat and rice) were studied at five sites, equipped with Open Top Chamber or O₃-FACE (free-air O₃ concentration enrichment) system. Based on exposure concentration and stomatal O₃ flux–response relationships obtained from the O₃-FACE experimental results in China, we found that throughout China current and future O₃ levels induce wheat yield loss by 6.4–14.9% and 14.8–23.0% respectively. Some policies to reduce ozone pollution and impacts are suggested.

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1. Introduction

Ground-level ozone (O₃) has been assumed to be the most phytotoxic air pollutant due to significant damage to the plants and rising trend in the concentration at a regional scale (The Royal Society, 2008). With the fast industrialization and urbanization in the last two decades, O₃ concentration is rising at a higher rate in China than other countries and the mean of the daily 24 h average O₃ concentration reaches more than 50 ppb during the crop growing season in some regions (Tang et al., 2013; Zhao et al., 2009). Published experiments have shown that ambient O₃ concentrations with an average of 40 ppb have significantly decreased the yield of major food crops (including wheat, rice, soybean, potato) by about 10% compared with ozone-free air (Feng and Kobayashi, 2009). A field survey around Beijing found a total of 28 species or cultivars exhibiting typical ozone symptoms (Feng et al., 2014). Therefore, it can be inferred that food security in China is being or has already been threatened by current O₃ concentration and this damage will continue in the future. In this manuscript, we review the current and future O₃ pollution, and its impacts on the food crop production throughout China.

2. Ambient O₃ and NO_x concentration throughout China

Until 1990 ozone concentrations in Chinese cities were low compared to the US and Europe, but they have increased quite rapidly since then due to increased emissions from automobile traffic and the use of fossil fuels in electricity generation and industry. Regional O₃ pollution has become one of the top environmental concerns in China, especially in those economically vibrant and densely populated regions. Fig. 1 summarizes the O₃ monitoring sites in the published literature that have provided data for at least three consecutive months during the past decades. Some major cities in China, such as Beijing, Shanghai, Jinan, Hong Kong, and Guangzhou are faced with photochemical threat. High surface O₃ concentrations are frequently reported throughout China (e.g., Lu et al., 2002; Ma and Zhang, 2000; Shan et al., 2006, 2009; Streets et al., 2007; Tang et al., 2009, 1995, 1989; Wang et al., 2007; Xu et al., 2008; Zhang et al., 1998).

The variability of O₃ concentration is a function of geographic location. Table 1 depicts the annual and seasonal average of O₃ concentration of various regions. In the central and northern part of China, the O₃ reaches a maximum in summer. However, in southern China, the ozone concentration is generally characterized by a peak in fall and a trough in summer. On the monthly-mean basis, surface O₃ peaks in May in the Yangtze River Delta (YRD), June in the North China Plain (NCP), and October in the Pearl River Delta (PRD),

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Fig. 1. The distribution of O₃ monitoring stations in China.

Table 1

Average O₃ concentrations (ppb) in different regions of China.

Region (abbreviation)	Measurement period	Average concentration (95% confidence interval)					Hourly maximum
		Annual	Spring	Summer	Fall	Winter	
Northeast/North (NE/N) ^a	1994–2012	23.5 (21.9–25.2)	30.3 (19.9–40.7)	31.5 (23.1–39.8)	19.4 (15.2–23.7)	15.4 (12.2–18.7)	164.5
North China Plain and Central/Western (NCP/C/W) ^b	1993–2011	32.4 (29.5–35.3)	36.8 (34.0–39.6)	45.7 (42.5–48.9)	29.6 (26.4–32.7)	20.2 (17.5–22.9)	316.0
South (S) ^c	1990–2012	33.7 (30.7–36.7)	33.5 (29.8–37.1)	26.5 (22.4–30.6)	39.9 (36.2–43.6)	32.5 (28–37.1)	213.5
Middle and Lower Reaches of Yangtze River (ML-YR) ^d	1991–2011	26.0 (24.4–27.6)	34.7 (32.5–36.8)	33.3 (31.1–35.5)	30.4 (28.5–32.2)	21.1 (19.0–23.2)	208.0
Southwest (SW) ^e	1993–2010	30.5	31.2	40.5	24.1	18.7	

^a Includes Heilongjiang, Jilin, Liaoning, Inner Mongolia, Sinkiang. Sources: (Cui, 2008; Duan et al., 2011; Li et al., 2003, 1999; Liu et al., 2013c; Meng et al., 2013).

^b Includes Hebei, Beijing, Tianjin, Shandong, Henan, Shanxi, Shaanxi, Gansu, Ningxia, Qinghai, Tibet. Sources: (An et al., 2007; Chen et al., 2010; Ding et al., 2008; Duan et al., 2008, 2011; Gao et al., 2005; Hao and Wang, 2005; Huang et al., 2012, 2013; Jin et al., 2008; Li et al., 2007, 1999; Lin et al., 2008; Liu et al., 2008a; Ma and Zhang, 2000; Ma et al., 2007, 2011; Meng et al., 2013; Nie et al., 2004; Shan et al., 2006, 2008, 2009; Tang et al., 2009; Tang et al., 2002; Wang et al., 2009a, 2006b, 2008, 2011b; Wu et al., 2011; Yang and Gao, 2008; Ye et al., 2008; Yin et al., 2004, 2005, 2006; Yu et al., 2011; Zhang et al., 1998; Zhao et al., 2013b; Zhu, 2004; Zong et al., 2007).

^c Includes Fujian, Guangdong, Hainan, Guangxi, Hong Kong, Macau, Taiwan. Sources: (Chan et al., 1998; Chang and Lee, 2006; Chen, 2006; Chen et al., 2011a; Chou et al., 2006; Deng, 2006; Duan et al., 2011; Huang et al., 2013; Lam et al., 2001; Lu, 2013; Meng et al., 2013; Song et al., 2012; Wang et al., 2012a, 2011a, 2009b, 2003; Wu and Huang, 2006; Zhang et al., 2011, 1998; Zhou et al., 2013; Zhou et al., 2008).

^d Includes Jiangsu, Anhui, Shanghai, Zhejiang, Hubei, Jiangxi, Hunan. Sources: (An et al., 2010; Chen et al., 2011b; Cheung and Wang, 2001; Duan et al., 2011; Geng et al., 2010; Hang, 2012; Hong et al., 2009; Huang et al., 2013; Li et al., 2007, 1999; Liu et al., 2008b; Liu et al., 2013b; Meng et al., 2013; Wan and Yang, 2007; Wang et al., 2006a, 2001; Xu et al., 2008; Yan et al., 2003; Yang et al., 2008; Zhang et al., 2003; Zhou, 2004).

^e Includes Chongqing, Guizhou, Sichuan, Yunnan. The number of records here is too small to estimate the confidence interval accurately. Sources: (Duan et al., 2011; Liu et al., 2013a).

respectively (Wang et al., 2011b). As shown in Table 1, the annual mean background O₃ over China shows a spatial gradient from 33.7 ppb in the South China to 23.5 ppb in the Northeast/North China. Compared with that of the mid- or high latitudes, tropospheric O₃ at low latitudes (e.g., in the tropical region) is found to be lower as a result of the inactivity of the stratospheric and tropospheric transport (Piotrowicz et al., 1986) and the low photochemical formation due to the lack of precursor sources in the large oceanic areas (Chan et al., 1998; McFarland et al., 1979).

As the main precursor of O₃, NO_x emissions have increased the most rapidly of any air pollutant in China over the last two or three decades (Liu et al., 2013d; Ohara et al., 2007; Zhang et al., 2012, 2009, 2007). The estimation of total NO_x emissions in China from 1990 to 2030 is illustrated in Fig. 2. Driven by the continuously rapid growth of economic activity and coal-dominated energy consumption, the NO_x emissions in China increased rapidly from 8.3 Tg in 1990 to 21.9–26.1 Tg in 2010, with an annual growth rate of 5%. From 1996 to 2011, the tropospheric columns of NO₂ over the

surrounding areas of Guangzhou, Shanghai, and Beijing increased by 82%, 292%, and 307%, respectively (Huang et al., 2013). China is presently the largest emitter of NO_x in Asia and will remain so if the current increasing trend of NO_x emissions continues in the future. Based on current legislation and current implementation status, China's total NO_x emissions are estimated to increase by a factor of 1.5–2 over the next 2 decades, making the largest NO_x emitter in the world (Aunan et al., 2000; Tian and Hao, 2003).

The background O₃ level over the mid latitudes of the Northern Hemisphere has risen in the range of approximately 0.5–2% per year over the past three decades (Vingarzan, 2004). Ding et al. (2008) analyzed aircraft O₃ data obtained from the MOZAIC (Measurement of Ozone and Water Vapor by Airbus In-Service Aircraft) program for the 1995–2005 period and found that summertime O₃ in the boundary layer near Beijing had increased by about 2% per year. The O₃ concentration data obtained from a coastal site in Hong Kong during 1994–2007 showed a relatively slower annual increase of 0.58 ppb, whereas comparing means in years 1994–2000 and 2001–2007 gives an annual increase of

0.87 ppb for a 7-year period (Lin et al., 2009; Wang et al., 2009b). The surface O₃ at Waliguan district from August 1994 to December 2005 showed an average annual growth rate of 0.23 ppb (De and Zhao, 2007). Model projections using IPCC emission scenarios for the 21st century indicate that background O₃ may rise to levels that would exceed internationally accepted environmental criteria for human health and the environment. Using five of the less conservative IPCC emission scenarios, the average global surface O₃ concentration is predicted to be in the range of 35–48 ppb by 2040, 38–71 ppb by 2060, 41–87 by 2080 and 42–84 ppb by 2100 (Vingarzan, 2004).

3. Impacts of elevated O₃ concentrations on food crop yields

To date, experiments on crops (winter wheat, rice and oil-rape) using different O₃ concentrations have been conducted at five different sites within China using Open Top chamber (OTC) or free-air ozone concentration enrichment system (O₃-FACE) facilities

Fig. 2. NO_x emissions in China during 1990–2030 (Tg NO₂ yr^{−1}). Sources are from Aardenne et al. (1999), Tian and Hao (2003), Zhang et al. (2007), Zhao et al. (2013a).

(Tables 2 and 3). Among these experiments, two experimental sites for winter wheat were situated in both Northern and Southern China, and one and three experimental sites for rice were situated in Northern and Southern China, respectively. The experiments in both Jiaxing and Jiangdu lasted five years to study the impacts of elevated O₃ concentrations on the growth, physiological characteristics and yield components of rice and winter wheat. Both experiments provided a lot of information for regional assessment of ambient and elevated O₃ effects on the food crop yield loss in the Yangtze River Delta. The experiment in Jiangdu site also investigated four cultivars of wheat and rice (including *Japonica*, *Indica* and *hybrid* cultivars) in response to elevated O₃ concentrations.

At the Gucheng site, non-filtered ambient air (NF), 50 and 100 ppb O₃ induced the yield loss in wheat by −4.7%, 10.5% and 58.6%, respectively, and in rice by 7.3%, 8.2% and 26.1%, respectively, relative to charcoal-filtered air (CF) (Feng et al., 2003). At the Jiaxing site, wheat yields were decreased by 8.5–58% and 40–73% and rice yields by 10–34% and 16–43% as compared with CF for O₃-1 (75 or 100 ppb) and O₃-2 (150 or 200 ppb) treatments, respectively, with the yearly variations in O₃ concentrations and AOT40 (accumulated O₃ concentration above a threshold level of 40 ppb) (Wang et al., 2012b). While at the Jiangdu site, a mean 25% enhancement above the ambient O₃ concentration (A-O₃, 45.7 ppb) significantly reduced the grain yield of winter wheat by 20% with significant variation in the range from 10% to 35% among the combinations of cultivar and season (Zhu et al., 2011). In rice, elevated O₃ also reduced the grain yield by 12% on average across four cultivars,

Table 3

A summary on the impacts of ozone on rice growth conducted in China in comparison to studies in Europe.

Site	Rooting	Duration	Facility	O ₃ treatments	Dose-response	Reference
Gucheng (39°08'N, 115°48'E)	Pot	2000	OTC	CF, NF, 50, 100, 200 ppb	RY = 100 −0.53 AOT40	(Feng et al., 2003)
Jiaxing (31°53'N, 121°18'E)	Field	2004–2008	OTC	CF, NF, 75/100, 150/200 ppb	RY = 100 −0.95 AOT40	(Wang et al., 2012b)
Jiangdu (32°35'N, 119°42'E)	Field	2007–2011	FACE	Ambient air (AA) & 1.5*AA (E-O ₃)	—	
Dongguan (23°01'N, 113°45'E)	Field	2010	OTC	NF, NF+40, NF+80, NF+120 ppb	RY = 100 −3.9 AOT40; RY = 100 −2.3 POD ₂	(Tong, 2011)
Synthesis from Europe	Pot & Field		OTC		RY = 94 −0.39 AOT40	(Mills et al., 2007)

The abbreviations are as defined in Table 2.

however, there were large differences in the extent of the yield losses (Shi et al., 2009).

To quantitatively estimate effects of O₃ on crop yield at regional, national and global scales, the relationships between the relative yield (RY) and different O₃ exposure indices of major crop species have been developed in North America and Europe based on controlled O₃ fumigation experiments under near-field conditions using Open Top chambers (Heagle, 1989; Jäger et al., 1992). Among O₃ exposure indices, AOT40 has been used widely during the last two decades as it has been found to have a strong relationship with relative yield of many crop species (Mills et al., 2007). Notably, the AOT40 index reflects the O₃ concentrations ([O₃]) in the air near to the plants during daylight hours, but it does not consider biological and climatic factors influencing daytime stomatal O₃ uptake. Recently, stomatal O₃ flux has been assumed to be more suitable index to develop the relationship with relative yield compared to AOT40 because the flux index depends not only on [O₃] at plant height, but also on stomatal conductance (g_{sto}), and its dependence on species or genotype as well as on phenological and climatic factors (Danielsson et al., 2003; Fuhrer, 2000; LRTAP Convention, 2010).

On the basis of field measurements, O₃ dose–response relationships for both wheat and rice (Tables 2 and 3) were developed for each experiment (with an exception of rice at the Jiangdu site). The field-grown winter wheat showed a similar O₃ concentration-based dose–response relationship for the three different sites, inducing much higher yield loss than for plants rooted in the pot (Gucheng site) and those conducted in Europe. However, large

Table 2

A summary on the impacts of ozone on wheat growth conducted in China in comparison to studies in Europe.

Site	Rooting	Duration	Facility	O ₃ treatments ^a	Dose-response ^b	Reference
Gucheng (39°08'N, 115°48'E)	Pot	1999	OTC	CF, NF, 50, 100, 200 ppb	RY = 100–1.30 AOT40	(Feng et al., 2003)
Jiaxing (31°53'N, 121°18'E)	Field	2004–2008	OTC	CF, NF, 75/100, 150/200 ppb	RY = 100–2.28 AOT40	(Wang et al., 2012b)
Jiangdu (32°35'N, 119°42'E)	Field	2007–2011	FACE	Ambient air (AA) & 1.5*AA (E-O ₃)	RY = 96.1–2.5 AOT40 RY = 100–8.2 POD ₁₂	(Feng et al., 2012)
Changping (40°12'N, 116°08'E)	Field	2010	OTC	NF, NF+30, NF+60, NF+90 ppb	RY = 100–2.2 AOT40 RY = 100–3.7 POD ₄	(Tong, 2011)
Synthesis from Europe	Pot & Field		OTC		RY = 99–1.6 AOT40 RY = 100–3.8 POD ₆	(Mills et al., 2007; Mills et al., 2011)

^a CF, charcoal-filtered ambient air; NF, non-charcoal-filtered ambient air; E-O₃, elevated O₃ concentration.

^b RY, relative yield; AOT40, accumulated O₃ concentration over a threshold of 40 ppb; POD_y (phytotoxic ozone dose), the accumulated stomatal O₃ flux above a flux threshold of y nmol O₃ m^{−2} s^{−1} (LRTAP Convention, 2010).

differences were found in the AOT40–response relationship for rice (Table 3), as indicated by the large difference in the slope of the AOT40 dose–response relationship at the different sites. This was possibly due to significant differences in response to O₃ among *Japonica*, *Indica* and hybrid genotypes of rice (Shi et al., 2009).

Only two sites (Jiangdu and Changping) developed stomatal O₃ flux response relationships due to the large amount of data required for complicated g_{sto} parameterizations (Table 2). There was a large difference in the stomatal O₃ flux threshold 'y' value (4, 6 and 12) which has been assumed the highest correlation with relative yield in each study including European experiments. This can be attributed to the different stomatal conductance functions and parameterizations among these studies. For rice, a stomatal O₃ flux response relationship was only developed at the Dongguan site. Stomatal O₃ flux models need to be developed at different sites throughout China to more accurately estimate the impacts of O₃ on wheat and rice yield at both regional and national scales.

4. Regional estimate of ozone effects on crop yield

So far, there have been three studies making an estimate on the relative yield loss of food crops by current and projected O₃ concentrations at national scale, despite a large difference between O₃ dose indices, even within each study (Table 4). Taking winter wheat as an example, the O₃ concentration in 2020 is projected to induce the yield loss by 2.9–7% based on M7/M12 function, 2.3–63% based on SUM06, 13.4–16.6% based on AOT40 and 19.2–23.0% based on stomatal O₃ flux. The large difference can be attributed to the different O₃ concentration formation models and sources in dose–response relationships. Only one dose–response relationship was built upon the results from the field experiment of four local cultivars in subtropical regions of China (Tang et al., 2013).

In Anun et al. (2000), a global three-dimensional photochemical tracer/transport model (CTM) of the troposphere was used to model surface ozone levels, combining ozone dose–response relationships from European and American experiments. In the 1990s, ambient O₃ concentration is thought to have led to little yield loss of the studied crops, with the exception of soybean (12% yield loss). However, high O₃ by 2020 was projected to induce losses throughout China of 18–21% (soybean), 29.3% (spring wheat), 13.4% (winter wheat), 7.2% (corn), and 5% (rice) (Table 4). In a recent study (Wang and Mauzerall, 2004), MOZART-2 (Model of Ozone and Related Chemical Tracers, Version 2) was used to simulate ambient O₃ concentrations for 1990 and 2020. The exposure indices and corresponding exposure–response relationships were obtained from the US NCLAN studies for wheat, corn

and soybean, and from Adams et al. (1989) for rice. Based on M7/M12 indices, the O₃ concentration in 1990 is calculated to have caused yield losses of less than 8%, while the four grain crop production losses due to O₃ in 2020 are estimated to be 47.4 million metric tons in China (Table 4).

A multiplicative model of leaf stomatal O₃ flux was parameterized using measurements of the stomatal conductance in the O₃-FACE study (Feng et al., 2012). Using the ozone concentration and stomatal O₃ flux relationship (75-days AOT40 and POD12, Feng et al., 2012) and those (90-days AOT40 and POD6) derived from European OTC experiments (LRTAP Convention, 2010), the impacts of tropospheric O₃ on wheat production throughout China were estimated, where surface [O₃] was estimated with a high resolution (40 × 40 km) chemical transport model coupled with the Regional Emission inventory in Asia (REAS). Results indicated that the [O₃] in 2000 caused an estimated yield loss in wheat of 6.4–14.9% throughout China. For the future projection, the O₃ concentration is estimated to induce the yield loss of winter wheat by 14.8–23.0% for China, which is much higher than the other two projections (Table 4). An O₃ dose response relationship is yet to be established for rice on the basis of five field experiments conducted in the O₃-FACE. From the results to date, the O₃ impact is estimated to double based on O₃ flux or triple based on O₃ exposure across a majority of rice producing areas in the middle and lower reaches of Yangtze River and the South China between the years 2000 and 2020 (Tang et al., 2013).

In addition to the national estimates, a regional yield loss of main crops by current ozone has been assessed. Liu et al. (2009) estimated the yield loss in wheat and rice by ozone using simulated O₃ data (based on CTM [Global three-dimensional Chemical Transport Model] and MOZART-2), and the O₃ dose relationship from the OTC study in Gucheng. The yield losses in Chongqing for wheat and rice were 12.0% and 10.8% respectively, and 12.0% and 9.2% in the Yangtze River Delta (Liu et al., 2009). Yao et al. (2008) estimated regional wheat, oil seed rape and rice yield losses of 17.1%, 5.9% and 2.3%, respectively, in the Yangtze River Delta on the basis of O₃ dose-relationships obtained from field measurements during 2004–2006 in Jiaxing and O₃ concentration data from 1999 to 2000.

In summary, there is a large uncertainty about the relative yield loss by current and future surface O₃ at a regional and national scale in China, due to the lack of observed O₃ concentrations in the rural regions and few controlled O₃-FACE experiments, but the studies to date have shown that the potential problem is very large and therefore warrants further study. Further information is also needed to consider the effects of multiple climate types and

Table 4

A summary on the national estimate on the relative yield loss (%) by ambient of O₃ concentration in 1990 or 2000 and a projection ozone in 2020 (number in bracket).

	Anun et al. (2000) 1990 (2020)			Wang and Mauzerall (2004) 1990 (2020)			Tang et al. (2013) 2000 (2020)			
	M7/M12 ^a	SUM06 ^b	AOT40 ^c	M7/M12	SUM06	W126 ^c	AOT40 (90d)	AOT40 (75d)	POD ₆	POD ₁₂
Winter wheat	1.6 (2.9)	0.0 (2.3)	1.7 (13.4)	6 (7)	13 (63)	12 (41)	6.4 (14.8)	7.2 (16.6)	14.9 (23.0)	10.3 (19.2)
Spring wheat	3.2 (8.2)	0.1 (29.3)	9.1 (29.3)	0.8 (2)	0.5 (30)	3 (22)				
Single rice	1.5 (4.5)			4 (8)						
Double E rice	1.1 (3.7)			3 (7)						
Double L rice	1.5 (4.4)			5 (10)						
Spring corn				8 (16)	3.5 (39)	1 (24)				
Summer corn	2.8 (7.2)	0.0 (7.2)		8 (16)	9.2 (64)	4 (45)				
Soybean	11.7 (20.9)	1.9 (17.8)		23 (33)	19 (45)	15 (37)				

^a M7/M12: seasonal 7 (wheat and rice) or 12 (corn and soybean) h d⁻¹ mean O₃ concentration.

^b SUM06: the sum, of the hourly ozone concentrations for hours when the concentration is at or above 60 ppb.

^c W126: sum of the hourly concentrations from 8 am to 8 pm from May to September in a year, where each concentration c_i is weighted by a sigmoidal function $w_i = 1 / (1 + 4403 \exp -0.126 c_i)$, to assign greater emphasis to the higher concentrations.

genotypes of food crops, and therefore it is necessary to launch large joint programs related to ozone. These should aim to further establish the relationship between ozone dose (AOT40 and PODy) and yield loss using field experiments at multiple sites throughout China in order to quantitatively assess how food security could be threatened by ozone pollution.

5. Policy recommendations to reduce impacts of ground-level O₃

5.1. Strictly controlling emissions of main precursors of O₃

In order to effectively reduce the concentration of ground-level O₃, we have to make a strict law to decrease the emissions of O₃ precursors (e.g., VOCs and NO_x) which are mainly from motor vehicle exhaust. Two suggestions are provided. One is that Euro standard V for motor vehicle exhaust should be enforced throughout China by 2020. The other is to reduce the motor vehicles on the road by further developing the public transportation at major and populous cities, increasing the price of petrol, and increasing the parking charges.

5.2. Breeding O₃ tolerant crop genotypes

Plants vary greatly in resistance to O₃ damage between species and cultivars. An effective method to reduce yield loss could be to breed O₃-resistant genotypes by adding the O₃-resistant genes using modern molecular biology techniques. Some genes conferring O₃-resistance have been determined in rice and soybean (Frei et al., 2010; Frei, 2015; Gillespie et al., 2011).

5.3. Applying chemical protective agents at key phenological stages

The damage caused to plants as a consequence of high O₃ concentrations could be mitigated or prevented by application of antioxidants, such as ascorbic acid, glutathione, and the antioxidant ethylenediurea (EDU) (Didyk and Blum, 2010; Feng et al., 2010; Lisko et al., 2014; Manning, 2000; Manning et al., 2011; Paoletti et al., 2009; Saitanis et al., 2015). These antioxidants and EDU have been extensively used in USA and many European countries to protect crop yield loss from ambient ozone. However, the toxicity of EDU in the food chain has not been tested, and it is also known that phytotoxicity can occur at high doses (Manning et al., 2011). Therefore, the most effective approach for utilization of these chemical protective agents could be applied at key phenological stages, e.g., grain filling in both wheat and soybean, and tillering in rice, where these crops are grown in regions with high ozone concentrations occurring frequently during the growing season. However, field trials to establish the most efficient usage of these chemical protective agents on crops have yet to be assessed.

6. Further research needs in the future

6.1. Establishing the O₃ dose model and model parameterization

It is difficult to assess the impact of ambient O₃ on crops productivities over the vast territory of China using a unified dose–response model, since the crop variety and climate differs greatly by latitude and region. There is limited research over various regions, except for some short-term studies on wheat and rice in the Yangtze River Delta (YRD), Pearl River Delta (PRD), and the regions of Beijing, Tianjin and Hebei. A more comprehensive study covering each of the major agricultural regions and staple food crops is a crucial task. To more accurately evaluate the yield and economic losses of crops caused by O₃ pollution, a series of

mathematical models that describe the relationship between crop productivity and ozone exposure under different conditions is required to be established and validated through local field investigations.

6.2. Developing the air quality standard for food security in China

Currently there are not any available standards to protect crops from O₃ in China. To develop a standard, we need to do more monitoring and measurements in rural areas and major crop production regions, including Hebei, Inner Mongolia, Jilin, Heilongjiang, Jiangsu, Anhui, Shandong, Henan, Hunan, Sichuan, Liaoning, Jiangxi and Hubei Provinces, all of which have a sowing area of more than 3 Mha and in combination account for approximately 76% of national food production. Combined with the controlled O₃ experiments at different sites mentioned above, ozone standards for cereal production could be developed for China.

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