Impacts of emission changes in China from 2010 to 2017 on domestic and intercontinental air quality and health effect

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Abstract. China has experienced dramatic changes in emissions since 2010, which accelerated following the implementation of the Clean Air Action in 2013. These changes have resulted in significant air quality improvements that are reflected in both surface networks and satellite observations. Air pollutants, such as $PM_{2.5}$, surface ozone, and their precursors, have long enough lifetimes in the troposphere to be easily transported downwind. Emission changes in China will thus not only change the domestic air quality, but will also affect the air quality in other regions. In this study, we use a global chemistry transport model (CAM-chem) to simulate the influence of Chinese emission changes from 2010 to 2017 on both domestic and foreign air quality. By applying health impact functions derived from epidemiology studies, we then quantify the changes in air pollution-related (including both $PM_{2.5}$ and O_3) mortality burdens at regional and global scales. Within our simulation period, the population-weighted annual $PM_{2.5}$ concentration in China peaks in 2011 (94.1 μg m^{-3}) and decrease to 69.8 μg m^{-3} by 2017. These estimated national $PM_{2.5}$ changes in China are comparable with previous studies using fine-resolution regional models, though our model tends to overestimate $PM_{2.5}$ from 2013 to 2017 when evaluated with surface observations. Relative to 2010, emission changes in China increased the global $PM_{2.5}$-related mortality burdens through 2013, by 27,700 (95%CI: 23,900—31, 400) deaths yr^{-1} in 2011, and 13, 300 (11,400—15,100) deaths yr^{-1} in 2013, among which 93% occurred in China. The sharp emission decreases after 2013 generated significant benefits for human health. By 2017, emission changes in China reduced premature deaths associated with $PM_{2.5}$ by 108, 800 (92,800—124,800) deaths yr^{-1} globally, relative to 2010, among which 92% were realized in China. In contrast, the population-weighted annual averaged maximum daily 8-hr ozone concentration peaked in 2014 and did not reach 2010-level by 2017. As such, O_3 generated nearly 8,500 (6,500—9,900) more

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premature deaths yr^{-1} in 2017 (143% of the global total), compared to the year 2010. Downwind regions, such as South Korea, Japan, and the United States, generally experienced O_3 improvements following 2013 as a result of mixed effects of increased export of ozone and decreased export of ozone precursors from China. Overall, we conclude that the sharp emission reductions in China over the past decade have generated substantial benefits for air quality and reduced premature deaths associated with air pollution at a global scale. The benefits are dominated by the $PM_{2.5}$ decreases in China since the ozone shows increasing trend while the emissions decrease.

1 Introduction

Fine particulate matter with an aerodynamic diameter of less than 2.5 μm ($PM_{2.5}$) has been of particular interest to the atmospheric chemistry research community due to its environmental impacts, such as visibility impairment and material damages (Hand et al., 2013, 2014; Wu and Zhang, 2018), and effects on human health (e.g. Pope et al., 2002; Krewski et al., 2009 and so on). Associations between short-term and long-term $PM_{2.5}$ exposures and various deleterious human health impacts have been widely and consistently reported (Krewski et al., 2009; Burnett et al., 2014; Liu et al., 2019), including premature mortality via several endpoints (e.g., cardiopulmonary and respiratory disease). Exposure to ozone (O_3) also impacts human health, with studies reporting an association between short-term O_3 exposure and hospital admissions, emergency room visits for respiratory causes and school absences (Katsouyanni et al., 2009), and long-term O_3 exposure with premature mortality from respiratory disease (Jerrett et al., 2009; Turner et al., 2016).

$PM_{2.5}$ and its precursors, can travel long distances, affecting air quality and human health in other receptor regions (Ewing et al. 2010; Pfister et al., 2011; Anenberg et al., 2014), despite its relatively short lifetime in the atmosphere (days to weeks). Tropospheric ozone features longer lifetime compared with $PM_{2.5}$, with approximately a global average of ~23 days (Young et al., 2013), and the research community has expressed particular interest in studying its intercontinental transport (e.g. Zhang et al., 2008, 2014; Cooper et al., 2015; Lin M. et al., 2012, 2017; Parrish et al., 2014). Numerous studies have investigated the source-reception relationship on air quality and associated premature mortality burden from emission changes in one source region on others (West et al., 2009a,b; Fry et al., 2013; Liang et al., 2018; Crippa et al., 2019). Liang et al. (2018) used the ensemble model outputs from the Task Force on Hemispheric Transport of Air Pollution (TF HTAP, Janssens-Maenhout et al., 2015), and estimated the source-receptor relationship between air quality and avoided premature deaths from a 20% reductions in anthropogenic emissions in East Asia. They estimated that 96,600 premature deaths would be avoided globally associated with $PM_{2.5}$ reductions, with 6% (5, 500 deaths) occurring in downwind regions. For long-term O_3 exposure, these emission reductions could lead to 11, 400 fewer premature deaths globally, with 15% (1,700 deaths) occurring downwind.

To tackle the severe air pollution problem in China, the Chinese government has implemented strict clean air policies in recent years (State Council of the People's Republic of China, 2013). Before 2013, the clean air control policies mainly focused on the emission standards of industry and power sectors. After 2013, eight more stringent control measures were developed and aimed to further reduce $PM_{2.5}$ pollution (see Fig. 1 in Zheng B. et al., 2018b). As a result, significant emission reductions have

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been observed in China, and the air quality has substantially improved especially after the "Air Pollution Prevention and Control Action Plan" (APPCAP) since 2013 (Zheng Y. et al., 2017; Zhang et al., 2019; UN Environment 2019; Zheng B. et al., 2018a,b). The relative change of China's anthropogenic emissions for major air pollutants during the 2010–2017 period include: −35 % for primary $PM_{2.5}$, -62 % for SO_2, −27 % for BC, −35 % for OC, −17 % for NO_x, and −27 % for CO (Zheng B. et al., 2018b). Surface monitors indicate that these reductions have resulted in significant decreases in ambient $PM_{2.5}$ concentrations, which are further reflected in observations from satellite and results from model simulations (e.g. Song et al., 2017; Huang et al., 2018; Li Y. et al., 2015; Li C. et al., 2017; Lin C. et al., 2018; Zheng Y. et al., 2017; Zhang Q. et al., 2019). The rapid reductions of major air pollutants in China were also confirmed using a long-term, robust observational record at Fukue Island, Japan (Kanaya et al., 2020). A recent study using high-resolution regional chemical transport model (CTM) showed that the estimated national population–weighted annual mean $PM_{2.5}$ concentrations decreased from 61.8 to 42.0 μg m^{-3} from 2013 to 2017 (Zhang Q. et al. 2019). Meanwhile, summertime daily maximum 8-h average (MDA8) O_3 in China has been shown an increasing trend since 2013 (Lu et al., 2018, 2020). The increasing trend in surface ozone may be partially explained by the slowing down of the aerosol sink of hydroperoxyl radicals (Li et al., 2019), though this chemical pathway remains disputed (Tan et al., 2020).

Previous studies have evaluated the benefits of China's APPCAP on improved air quality including, both both $PM_{2.5}$ and O_3 pollution, and avoided premature mortality (e.g. Huang et al., 2018; Zhang et al., 2019; Lu et al., 2020). However, limited studies have investigated the benefits of these actions on global air quality and the air pollution related premature mortality burden. In this study, we use a global CTM to simulate the global air quality changes from 2010 to 2017 that result from the historical emission changes in China during the same time. We then estimate the air pollution-related premature mortality burden, both within China and its downwind regions. This particular time period was selected because the emissions in China slightly increased from 2010 to 2012, and then substantially decrease thereafter, which allows for a comparison of different emission trends in China on global air quality and health burdens.

2 Data and Methods

2.1 Model simulation using the CAM-chem

Global air quality is simulated from 2010 to 2017 using the CAM-chem model (CAM version 4, Lamarque et al., 2012) at a horizontal resolution of 1.9◦ (latitude) × 2.5◦ (longitude), and 56 vertical levels that span the surface to 4 hPa (≈40 km), and driven by the NASA GEOS5 Global Atmosphere Forcing Data (Tilmes 2016, http://rda.ucar.edu/datasets/ds313.0/, last Accessed 20 April, 2020). Additional modeling configurations, including the lower boundary conditions for long-lived species such as CO_2 and CH_4, the online biogenic emission inventory, and other natural emissions, are described in our previous studies (Zhang et al., 2016, 2021; Tilmes et al., 2016). In this version of CAM-chem, the bulk aerosol model was applied based on the work of Tie et al. (2001, 2005), in which the sulfate aerosol is formed by the oxidation of SO_2 in both the gas and aqueous phase (Lamarque et al., 2012; Tilmes et al., 2016). Ammonium nitrate is also represented depending on the amount of sulfate

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present in the air mass following the parameterization of gas/aerosol partitioning by Metzger et al. (2002). The secondary organic aerosols (SOA) are derived using the two-product model approach with laboratory determined yields for SOA formation from monoterpene, isoprene and aromatic photooxidation (Heald et al., 2008; Times et al., 2016), coupled to both the chemistry and biogenic emissions. The uncertainties in estimating accurate anthropogenically influenced SOA (Cao et al., 2018; Zheng et al., 2018) could make our model simulated $PM_{2.5}$ concentrations conservative (Nault et al., 2021). $PM_{2.5}$ concentration is calculated as the sum of SO_4+NO_3+NH_4+OC+BC+SOA (secondary organic aerosol)+0.2*Dust+Seasalt (West et al., 2013; Silva et al., 2016). For dust and sea salt, only the size fractions relevant for $PM_{2.5}$ (size bins 1-3) are used. Dust in desert regions was found to be too high in the model, so global dust concentrations were multiplied by 0.2 to achieve rough consistency with the $PM_{2.5}$ concentrations estimated with Brauer et al. (2012). A comprehensive evaluation of the model performance in simulating temporal and spatial distribution of global ozone and aerosols by comparing surface observation, balloon, aircraft and satellite was carried out in previous studies (Tilmes et al., 2015, 2016; Zhang et al., 2016, 2021). Here, we evaluate the model using the surface ozone and $PM_{2.5}$ concentrations in China from 2013 to 2017. The lowest modeled level from CAM-chem (extending to approximately 58 m above the surface) is taken to indicate ground-level concentrations. Base case simulation (named CEDS_MEIC in Table 1) was run consistently from 2010 to 2017 with one year spin-up in 2009 using time-varying global anthropogenic emission inventory as a combination of the Community Emissions Data System (CEDS, v2017-05-18, Hoesly et al., 2018) developed by the Pacific Northwest National Laboratory, and the Multi-resolution Emission Inventory (MEIC, http://www.meicmodel.org/, last access 20 April 2020) developed by Tsinghua University, China. We performed first sensitivity simulation (CEDS_MEIC_ChinaFix) which keeps China emissions constant at the level of 2010, and the differences between the CEDS_MEIC and CEDS_MEIC_ChinaFix are the influences of anthropogenic emission changes in China since 2010 on global air quality and human health (Table 1). By making comparisons between these two scenarios, we can also rule out the influences of meteorological variabilities on the global distribution of $PM_{2.5}$ and ozone. The global anthropogenic emissions other than China after 2014 were kept constant, since no global emissions inventory was available after 2014 when we first prepared the study. McDuffie et al. (2020) updated the global CEDS anthropogenic emissions through 2017 with continued update into 2019 (https://github.com/JGCRI/CEDS, last access 6 May 2021). However, this should have negligible effect on our conclusions since our focus is the emission changes from China on influence of domestic and international air transport. We also performed another sensitivity simulation with global anthropogenic emissions keeping constant at 2010 level (CEDS_MEIC_GlobalFix, Table 1). The air quality changes from 2011 to 2017 relative to 2010 from this sensitivity could allow us to take a look at the meteorological changes on $PM_{2.5}$ and ozone changes in China at the same time.

It also came to our attention that the CEDS emissions tends to overestimate the magnitude of the Chinese emissions, and underestimate the emission decreasing trend in China (Zheng B. et al., 2018b; Paulot et al., 2018). From Fig. S1, we see that in 2014, the emissions from CEDS are at least 20% higher than the estimations from MEIC for most of the air pollutants, except for non-methane volatile organic compounds (1%). More specifically, the SO_2, OC and BC emissions estimated in CEDS are 84%, 81% and 58% higher individually than those estimated in MEIC. For the emission trend, CEDS estimated a

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continued increasing trend while MEIC showed a peak for the majority air pollutants before the year 2012 (Liu et al., 2016; Zheng B. et al., 2018b). From Fig. S2 which shows the spatial patterns of the emission differences between CEDS and MEIC in 2014, we can see that the emissions in CEDS are higher in the western and south China, and lower in the eastern China. We performed another sensitivity simulation in which we used global anthropogenic emission from CEDS only from 2010 to 2014 (CEDS_Global), to evaluate the model's performance in simulating $PM_{2.5}$ and ozone in China, and also discuss the relative air quality changes applying different emission inventory in China.

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2.2 Surface observation for $PM_{2.5}$ and O_3 in China

Surface observation for hourly $PM_{2.5}$ and ozone concentration were downloaded from China National Environmental Monitoring Center (CNEMC) Network (http://106.37.208.233:20035/) from 2013 to 2017, since data was not available before 2013. We evaluated the model's performance in simulating annual $PM_{2.5}$ and maximum daily 8 h average (MDA8 O_3) as these two metrics are related with the health impact analysis we performed in our study.

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2.3 Health impact assessment for surface $PM_{2.5}$ and O_3

We applied the CRF derived from long-term cohort studies, together with the baseline mortality rates and exposure population to quantify the air quality related mortality burden changes. The mortality burdens related to ambient air pollution including $PM_{2.5}$ and O_3 are calculated following Eq. (1):

$$\Delta Mort = Y_0 \times AF \times Pop\ , \qquad (1)$$

Where $\Delta Mort$ is mortality burden attributed to surface $PM_{2.5}$ or O_3, Y_0 is the baseline mortality rates for cause of specific disease, AF is the attribution fraction calculated as $1 - \frac{1}{RR}$ with RR as the relative risk, and Pop is the exposed population with ages greater than 25 years old. The RR for the $PM_{2.5}$ is calculated using the integrated exposure response model (IER, Burnett et al., 2014), following the previous methods in our group (Shindell et al., 2018). We used the 1000 simulations for the parameter distributions of α, β and γ retrieved from the Global Burden of Disease 2017 (GBD2017) study (Stanaway et al., 2018) to derive the mean RR with 95% uncertainty intervals. The RR for the long-term exposure to annual average of maximum daily 8 h average ozone concentration (hereafter MDA8) is from the updated cohort study (Turner et al., 2016), with RR of 1.12 (95 % confidence interval (CI): 1.08, 1.16) for respiratory disease. Country-age-specific baseline mortality rates (Y_0) in 2010 were retrieved from the GBD2017 project (Stanaway et al., 2018), and remapped to match the 10th International Statistical Classification of Diseases and Related Health Problems codes as used in the cohort study (Turner et al., 2016; Seltzer et al., 2020). The theoretical minimum risk exposure level for $PM_{2.5}$ exposure assessment is drawn from a uniform distribution with a lower bound of 5.8 μg m^{-3} and an upper bound of 8.8 μg m^{-3}, and for ozone exposure assessment is 26.7 ppbv. Previous studies have shown that the global CTMs with coarse resolutions, e.g. 1.9°×2.5° in our study, would likely bring low biases in estimating health effects especially in urban areas (Li et al., 2016; Punger and West, 2013; Silva et al., 2013, 2016). However, less is known how these underestimates would affect the relative contributions of downwind transportation (Liang et al., 2018).

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Meanwhile, Jin et al. (2019) concluded that in estimating the ambient $PM_{2.5}$-related mortality burdens, the uncertainties from the CRFs are much larger than the uncertainties from the $PM_{2.5}$ concentration estimations.

3 Results

3.1 Model evaluation with surface observation in China

From the model evaluation metrics, we see that the CAM-chem base simulation (CEDS_MEIC scenario, see Table 1) generally overestimates the annual $PM_{2.5}$ concentration in China, with mean bias (MB) of 19.3 μg m^{-3} and normalized mean bias (NMB) of 37.2% for all the 5 years. The MBs are around 20 μg m^{-3} from 2014 to 2017 with lowest values in year 2013 (MB of 7.6 μg m^{-3}) and highest in 2015 (21.7 μg m^{-3}). The lower MB and NMB in 2013 could be explained by the much less data availability. The positive NMBs for all the years shows that the overestimations are systematic and may not affect our main conclusions since we focus on the changes among years. The higher modelling bias for surface $PM_{2.5}$ from the CAM-chem were also seen in other studies, for example, He and Zhang (2014) reported NMB of 37.6% and 41.85% for the contiguous United States and Europe in 2001. The bias in simulating the surface $PM_{2.5}$ in the CAM-chem (version 4) was mainly caused by the inaccurate prediction of SO_4^{2-}, NH_4^+, and organic aerosols, and missing major inorganic aerosol species such as nitrate and chloride (He and Zhang, 2014; Tilmes et al., 2016). By including advanced inorganic aerosol treatments, such as condensation of volatile species, explicit inorganic aerosol thermodynamics for sulfate, ammonium, nitrate, sodium, and chloride (He and Zhang, 2014), and more comprehensive secondary organic aerosols approach (Volatility Basis Set scheme, Times et al., 2019; Liu et al., 2020), the performance for simulating surface $PM_{2.5}$ could be significantly improved. The different metrics also exhibits small annual variabilities, with NMB around 40% and NME around 50% for each year, except for 2013 and 2014 which have smaller NMB values due to the limited size of the observations (Table 2). The CAM-chem can generally reproduce the spatial patterns of the annual $PM_{2.5}$ distributions, with correlation coefficient (R) greater than 0.7. With CEDS emissions applying globally (CEDS_Global scenario, see Table 1), we have better performance in both 2013 and 2014 for the annual $PM_{2.5}$ evaluation. Part of the reason is that we have less available data in these two years, while we suspect the main reason is that though total emissions in China are higher in CEDS than those in MEIC, CEDS tends to underestimate the emissions in eastern and central China (Fig. S2), where the majority of observations are available. With CEDSs emission applied, the annual $PM_{2.5}$ is lower in eastern China and higher in western and northwestern China (Fig. S3). The performance for CAM-chem in simulating annual MDA8 O_3 is slighter better than the surface $PM_{2.5}$, with NMB lower than 20% for all the years (Table 3). From Table 3, we can also see that the CAM-chem overestimates the annual MDA8 O_3 in China, which means our estimation for the O_3-related mortality burden will have positive bias. A high bias of about 10 ppb can be attributed to the coarse model resolution, which leads to an overestimate of ozone production, because of diluted emissions of ozone precursors (Tilmes et al, 2015). For annual MDA8 O_3, the CEDS simulation (CEDS_Global scenario) has a poorer performance compared with using MEIC emissions, which maybe caused by the overestimation of the surface ozone in western China (Fig. S4). For both $PM_{2.5}$ and ozone, we also find that the NMBs are lower in the eastern China compared with other inland regions (Figs. S5-S6).

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3.2 Air quality changes in China from 2010 to 2017

We first report the trends for the national annual population-weighted (Pop-weighted) $PM_{2.5}$ which decreased by by 17.6%, changing from 84.7 μg m^{-3} in 2010 to 69.8 μg m^{-3} in 2017 (Fig. 1). The Pop-weighted $PM_{2.5}$ was highest in 2011 (annual average of 94.1 μg m^{-3}), and decreases until 2017 (69.8 μg m^{-3}). From 2013 to 2017, the national annual Pop-weighted $PM_{2.5}$ decreased by 15.9 μg m^{-3}, comparable to the values (19.8 μg m^{-3}) estimated by Zhang Q. et al. (2019) which used a high-resolution regional air quality model. The Pop-weighted national average $PM_{2.5}$ concentrations shares a similar trend as the area-weighted average but much higher (Fig. 1a), demonstrating the fact that higher $PM_{2.5}$ concentrations happen in more density region. The annual average of area-weighted $PM_{2.5}$ concentration decreased by 7.6 μg m^{-3}, consistent with the estimations by Ding et al. (2019a) at 9.0 μg m^{-3}, which also demonstrates the fact the overestimations of annual $PM_{2.5}$ from CAM-chem are systematic and will unlikely affect our trend analysis. The surface $PM_{2.5}$ changes in China due to emission changes usually peak in the fall and winter (Fig. S7a). For the spatial patterns of the $PM_{2.5}$ changes, we see that significant annual $PM_{2.5}$ changes (increases before 2013 and decreases after then) mainly occur in the eastern China (Fig. 2; Table S1), which were the focused regions for China APPCAP (Ding et al., 2019a,b). When distinguishing the contributions from anthropogenic emissions vs. meteorology, we find that the annual $PM_{2.5}$ decreases in China are mainly dominated by the emission changes, consistent with previous studies (Dang and Liao, 2019; Ding et al., 2019a; Zhai et al., 2019; Zhang Q. et al., 2019). Compared with the year 2010, interannual meteorology led to annual $PM_{2.5}$ decreases as high as 4.4 μg m^{-3} in 2011, and increases as high as 3.1 μg m^{-3} in 2015, while the emission changes led to annual $PM_{2.5}$ decreases as high as 16.7 μg m^{-3} in 2017, and increases as high as 4.9 μg m^{-3} in 2015 (Fig. 4a). There were also isolated increases in $PM_{2.5}$ in northwest China from 2010 to 2013, which were mainly caused by the dust storms (Meng et al., 2019; Luo et al., 2020; Zhao et al., 2020). Different from the $PM_{2.5}$ trend, the annual Pop-weighted average MDA8 O_3 has a continued increasing trend since 2010 (Fig. 1b), with peaks in 2014 (59.5 ppbv), and then decreases to 2017 (57.1 ppbv). The area-weighted MDA8 O_3 was comparable or even larger than the Pop-weighted (e.g. 2012), as a result of more uniform O_3 distribution in China or even higher ozone events in western China because of stratosphere-troposphere exchange with less population (Wang et al., 2011; Li et al., 2019). For ozone, the emission changes from 2010 to 2013 exacerbate summer ozone pollution in China, but alleviate ozone pollution in the other three regions (Fig. S8a). After 2013, the emission decreases in China exacerbate the ozone pollution for all the seasons, especially in winter. For the spatial patterns, the ozone increases mainly in Beijing–Tianjin–Hebei and Yangtze-River-Delta, and slightly decreases in the south (Fig. 3; Fig. S9; Table S2). The anthropogenic emission reductions in China leads to ozone increases (Fig. 4b), which could be partially explained by aerosol sink of hydroperoxyl radicals slowing down due to $PM_{2.5}$ decreases (Li et al., 2018). The interannual meteorological condition changes have a much larger positive effect on the annual MDA8 O_3, compared with the anthropogenic emission changes, leading to ozone increases as high as 8.1 ppbv in 2014 and as low as 0.7 ppbv in year 2011. The meteorology-induced ozone increases can be attributed to the increasing temperature which enhances the ozone production and biogenic NMVOCs emissions (Ding et al., 2019b; Liu and Wang, 2020), and the increasing solar radiation (Wang et al., 2020; Ma et al., 2021).

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3.3 Emission changes in China on global air quality and health

3.3.1 Global and regional air quality

The simulated global tropospheric ozone burden (total column ozone below the chemical tropopause of 150 ppbv) from 2010 to 2017 calculated from the CEDS_MEIC simulation is 327.5 ± 5.2 Tg, agreeing well with the previous estimations using ensemble models (ACCENT: 336 ± 27 Tg; ACCMIP:337 ± 23 Tg; TOAR: 340 ± 34 Tg, and CMIP6: 348 ± 15 Tg; Griffiths et al., 2021). From Fig. 5, the change for global tropospheric ozone burden from the emission changes in China ranges from 0.6 Tg (2011 and 2012) to -1.9 Tg (2017). The tropospheric ozone burden changes are not only seen in China, but also in downwind regions, especially in Northern Hemisphere, such as Pacific Ocean and U.S., as a result of the vertical transport of the air pollutants (Fig. S10).

Due to the prevailing western wind, the air pollutants in China could be easily transported to downwind regions, especially during springtime (Lin M. et al., 2012; Liang et al., 2018). From Fig. 6, we see that for downwind regions, South Korea has the largest pop-weighted $PM_{2.5}$ changes, ranging from 0.7 μg m^{-3} in 2012 to -2.63 μg m^{-3} in 2017, following by Japan and U.S. (Fig. 6a). The influences are largest in spring than the other seasons for all the downwind regions (Fig. S7b-d). However, for ozone, we see that the emission changes in China have increased the surface ozone in South Korea from 2010 to 2017, mainly dominated by the increased export of ozone. Both Japan and U.S. are firstly seen an increase in ozone and then decrease (Fig. 6b). However, the increases and the decreases are not usually following the same year as emissions and $PM_{2.5}$ concentration changes, demonstrating the non-linearity of the ozone productions. For the downwind regions, the season with peak ozone changes also varies (Fig. S8b-d).

3.3.1 Global and regional air pollution-related mortality burden changes

The global ambient $PM_{2.5}$-related mortality burden in 2010 is 4.08 million (95%CI: 2.15—6.0 million), similar with previous estimated in the same year applying the same IER method (3.6±1.0 million in 2010, Shindell et al., 2018). Compared with 2010, the emission changes in China (by comparing CEDS_MEIC and CEDS_MEIC_ChinaFix) lead to mortality increases by 27, 700 deaths yr^{-1} (95%CI: 23, 900—31, 400 deaths yr^{-1}) in 2011, with 93% occurring in China (25, 800, 95%CI: 22, 300—29, 200 deaths yr^{-1}; Table 4). From Table 4 (last column), we can see that China takes the largest majority of global $PM_{2.5}$-mortality burden changes from 2011 to 2017 (more than 90%), because of the relative linear relations between emission and concentration for $PM_{2.5}$, and its relative shorter lifetime in the troposphere. Compared with year 2010, the emission changes in China leads to 108, 800 (95%CI: 92,800—124, 800) avoided premature deaths in 2017, with 92% (95%CI: 85, 900—114, 300) occurring in China. Among the three downwind regions, Japan displays the largest influences for the $PM_{2.5}$-related mortality burden changes, ranging from 197 deaths yr^{-1} in 2011 to -875 deaths yr^{-1} in 2017. The emission changes in China have comparable effect on the $PM_{2.5}$-related mortality burden change among South Korea and U.S. (Table 4).

The global ozone-related mortality in 2010 is 1.02 million (95% CI: 0.73—1.28 million), consistent with previous estimations using other global CTMs, such as GEOS-Chem (1.04—1.23 million) and GISS (0.8—1.3 million), applying the same relative

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risk (Mally et al., 2017; Shindell et al., 2018). The emission changes in China increased the global ozone-related mortality by 4,900 (95%CI, 3,700—5,900) deaths yr^{-1} in 2011 (Table 5), among which 73% is happening in China (3600 deaths yr^{-1}, 95%CI: 2,700—4,300). For the three downwind region, South Korea, Japan, and U.S., the added O_3-related mortality burdens in each country are 23, 200 and 131 deaths yr^{-1} individually. In 2017, the reduced anthropogenic emissions in China greatly increased the ozone-related mortality burden, by 8,500 deaths yr^{-1}, which is 43% higher than the global total added O_3-related mortality burden (5,920 deaths yr^{-1}). In general, the downwind regions have decreased O_3-related mortality burden (-65 and -289 deaths yr^{-1} for Japan and U.S. individually, Table 5), except for South Korea, which also has increased mortality burden by 17 deaths yr^{-1} (Table 5).

4 Conclusions

Dramatic changes are observed for anthropogenic emissions in China since 2010, with majority of air pollutants, such as NO_X, SO_2, peaking around 2012 and 2013, and decreasing significantly afterwards. In this study, we use a global chemistry transport model (CAM-chem) to simulate the effects of emission changes in China on domestic and downwind regions' air quality and air pollution-related mortality burden from 2010 to 2017. Compared with surface $PM_{2.5}$ observation networks in China, our model tends to overestimate the annual $PM_{2.5}$ concentrations, with normalized mean bias (NMB) of 37.2% and normalized mean error (NME) of 52.0% from 2013 to 2017. The $PM_{2.5}$ overestimation from CAM-chem was likely caused by the uncertainties in the bottom-up emission inventories (Shen et al., 2019; Zhang Q. et al., 2019), and missing chemical mechanisms for the $PM_{2.5}$ components (Tilmes et al., 2015, 2016; Liu et al., 2020), and are in the same magnitudes as seen in other high resolution regional models (25%-30% in Shen et al., 2019; ~20% in Zhang Q. et al., 2019). We also evaluated the model performances by applying two sets of emission inventories: the regional emission inventory (MEIC) developed by Tsinghua University and the global emission inventory (CEDS) developed by PNNL which was extensively in the CMIP6 experiments. For surface $PM_{2.5}$, we find that the CAM-chem with CEDS emissions tends to have a lower NMB and NME in 2013-2014 since CEDS emissions are lower in urban areas in China than that from MEIC, though the national totals from CEDS are much higher than those from MEIC. For surface O_3, CAM-chem with MEIC emissions in China has a lower NMB (13.7%) and NME (21.9%) for the annual MDA8 O_3. The simulations with CEDS in China tend to have larger NMB (15.2%) and NME (36.6%) in 2013-2014.

From 2010 to 2017, we simulated that, the annual average population-weighted (Pop-weighted) $PM_{2.5}$ increases in China till 2011 (94.1 μg m^{-3}), and then decreases sharply afterwards. The annual Pop-weighted $PM_{2.5}$ in 2017 decreases by 17.6% (-14.9 μg m^{-3}), compared with the value in 2010 (84.7 μg m^{-3}), and 18.5% lower than 2013 (85.8 μg m^{-3}). Though CAM-chem overestimates the $PM_{2.5}$ concentrations in China, the simulated decreasing trend for annual $PM_{2.5}$ from 2013 to 2017 (-15.9 μg m^{-3} for Pop-weighted, and 7.6 μg m^{-3} for area-weighted) is comparable with previous studies using high resolution regional air quality models (-19.8 μg m^{-3} estimated from Zhang Q. et al. (2019), and -9.0 μg m^{-3} from Ding et al. (2019a)). The emission changes in China from 2010 to 2013 increased the global $PM_{2.5}$-related mortality burdens, varying from 27, 700 deaths yr^{-1}

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(95% confidence interval (CI): 23,900—31, 400) in 2011, and 13, 300 (95%CI: 11,420—15, 110) deaths yr^{-1} in 2013. The overestimation of the surface $PM_{2.5}$ concentration in China from our model was unlikely to affect the estimations of the magnitudes and trends of the $PM_{2.5}$-related health benefits, because of the linearity of the IER functions at higher $PM_{2.5}$ concentration in China (Zhang Q. et al., 2019). Among the increased premature deaths, at least 93% occurring in China. The sharp emission decreases after 2013 bring significant benefits of reduced premature mortality in 2017, reaching 108, 800 (92,800—124, 800) deaths yr^{-1}, among which 92% (100, 100 deaths yr^{-1} with 95%CI: 85, 900—114, 300) happening in China. Downwind regions, such as South Korea, Japan, and U.S. share similar $PM_{2.5}$ trend as China. The transport of $PM_{2.5}$ and its precursors could change the annual Pop-weighted $PM_{2.5}$ ranges from 0.7 μg m^{-3} in 2011 and -2.6 μg m^{-3} in 2017, leading to added 98 premature deaths in 2011, and 386 avoided premature deaths in 2017. Japan has a smaller change for the annual Pop-weighted $PM_{2.5}$, but much larger changes in $PM_{2.5}$-related mortality burden changes, ranging from 197 added premature deaths in 2011, and 875 avoided premature deaths in 2017, mainly caused by the much higher population in Japan than that in South Korea (https://countryeconomy.com/countries/compare/japan/south-korea?sc=XE23, last accessed Sep 3rd, 2021). The influence for U.S. ranges from 44 added premature deaths in 2011, and 381 avoided premature deaths in 2017. Different trend as $PM_{2.5}$, the emission changes in China increased the annual maximum daily 8-hr ozone concentration in China, and also the ozone-related premature deaths, ranging from 3, 600 deaths yr^{-1} in 2011 (75% of global total increased premature deaths), and 8, 500 deaths yr^{-1} in 2017 (143% of global total). Downwind regions, such as South Korea, Japan, and U.S. are generally seen decreased O_3-related mortality burden after 2013 as a combination of increased export of ozone and decreased export of ozone precursors. In general, we conclude that the sharp emission reductions in China after 2013 bring benefits of improved air quality and reduced premature deaths associated with air pollution at the global scale. The benefits are dominated by the $PM_{2.5}$ decreases.

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Data availability. Global anthropogenic emissions data from CEDS are available from https://www.geosci-model-dev.net/11/369/2018/ (accessed May 4th, 2020). MEIC emission inventory is available from http://meicmodel.org/?page_id=560 (last access May 6th, 2021). Baseline health and population data are available from the World Health Organization and the United Nations, respectively. The CAM-chem model is available at http://www.cesm.ucar.edu/models/cesm1.2/ (accessed May 4th, 2020). Data from CESM modelling that support the findings of this study are available from the corresponding author upon request.

Author contributions. YZ and DS originally designed the study, and YZ conducted all simulations, created all figures and wrote the manuscript, with comments and edits from all authors. BZ and QZ contributed to develop the MEIC emission inventory.

Competing interests. The authors declare that they have no conflict of interest.

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Table 1: Model simulation performed for this study.

Name	Anthropogenic Emissions	Meteorology
CEDS_MEIC	MEIC in China from 2010 to 2017 CEDS outside China from 2010 to 2017[1]	2010 to 2017
CEDS_MEIC_ChinaFix	MEIC in China constant as in 2010 CEDS outside China from 2010 to 2017[1]	2010 to 2017
CEDS_MEIC_GlobalFix	Emissions kept constant at 2010 level from both MEIC and CEDS	2010 to 2017
CEDS_Global	2010-2014 from CEDS globally	2010 to 2014

[1]The global emissions other than in China after 2014 stay constant as the year 2014 since the global anthropogenic emissions after 2014 were not available when we carried out study.

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Table 2: Model performance for the annual $PM_{2.5}$ concentration compared with surface observation in China from 2013 to 2017, with mean bias (MB, μg m^{-3}), normalized mean bias (NMB, %), normalized mean error (NME, %), and root-mean-square error (RMSE, μg m^{-3}).

Year	# of stations	MB (μg m^{-3})	NMB (%)	NME (%)	RMSE (μg m^{-3})
2013	378	6.9	9.5	29.4	27.3
2014	537	21.8	34.3	45.6	39.7
2015	1438	21.6	41.3	54.0	39.5
2016	1431	19.2	40.0	54.5	36.5
2017	1461	19.4	42.2	55.7	34.8
2013-2017	5245	19.3	37.2	52.0	26.7
CAM_Chem using CEDS emissions only[1]					
2013[1]	378	0.8	1.0	23.5	20.1
2014[1]	537	19.5	30.1	42.1	34.3
2013-2014[1]	915	11.8	17.5	33.9	29.5

[1]The CAM-chem simulations applying global CEDS emissions only, which only has been ran from 2010 to 2014.

Table 3: As Table 2 but for annual MDA8 O_3, with mean bias (MB, ppbv), normalized mean bias (NMB, %), normalized mean error (NME, %), and root-mean-square error (RMSE, ppbv).

Year	# of stations	MB (ppbv)	NMB (%)	NME (%)	RMSE (ppbv)
2013	1029	4.8	11.6	20.2	10.6
2014	1033	7.2	17.5	24.1	12.4
2015	1026	6.0	14.4	23.0	11.9
2016	1031	6.5	15.7	22.7	11.8
2017	1042	3.8	9.3	19.5	10.3
2013-2017	5161	5.7	13.7	21.9	11.4
CAM_Chem using CEDS emissions only[1]					
2013[1]	1029	14.1	33.8	34.5	16.4
2014[1]	1033	16.3	39.2	39.6	18.3
2013-2014[1]	2062	15.2	36.6	37.0	17.4

[1] The CAM-chem simulations applying global CEDS emissions only, which only has been ran from 2010 to 2014.

Table 4. The changes for the $PM_{2.5}$- mortality burden under the emission changes in China from 2010 to 2017 in China, as well as three other downwind regions—South Korea, Japan and U.S. The mortality burden changes at global level are also included. Positive values mean emission change in China increase the $PM_{2.5}$-related mortality burden in this region, and negative values mean decreases the $PM_{2.5}$-related mortality burden.

Year	China	South Korea	Japan	U.S.	Global	% (China/Global)
2011	25,800	98	197	44	27,700	93%
2012	26,200	80	143	24	27,900	94%
2013	12,600	-21	-122	-39	13,300	95%
2014	-19,620	-147	-306	-138	-21,800	90%
2015	-47,670	-226	-541	-191	-51,600	92%
2016	-77,600	-264	-615	-284	-83,200	93%
2017	-100,100	-386	-875	-381	-108,800	92%

Table 5. As Table 4 but for ozone-related mortality burden changes.

Year	China	South Korea	Japan	U.S.	Global	% (China/Global)
2011	3,600	23	172	131	4,900	73%
2012	3,400	15	113	140	4,900	70%
2013	5,500	17	115	93	6,600	83%
2014	6,400	25	68	-5	6,500	99%
2015	7,400	22	34	-100	6,500	113%
2016	7,500	11	-56	-188	5,600	133%
2017	8,500	17	-65	-289	5,900	143%

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Figure 1: National population-weighted (Pop-Weighted) and area-weighted annual $PM_{2.5}$ (a), and MDA8 O_3 (b) from 2010 to 2017 from our base model simulation (CEDS_MEIC). The units are μg m^{-3} for $PM_{2.5}$ and ppbv for ozone.

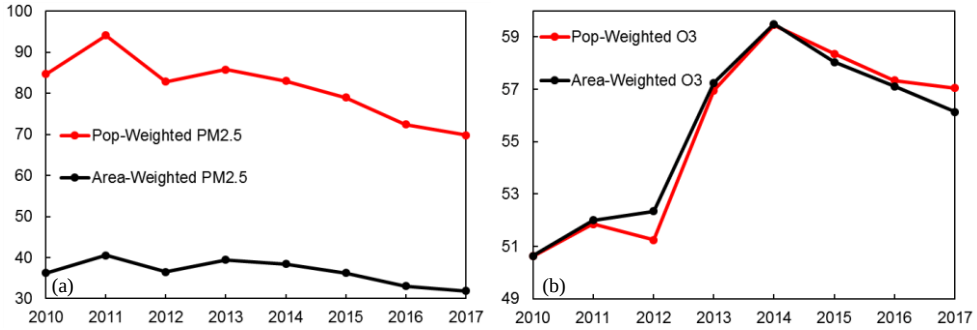

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Figure 2: Annual $PM_{2.5}$ changes (unit of μg m^{-3}) from 2011 to 2017 due to anthropogenic emission changes in China only. The results are calculated as the differences between CEDS_MEIC and CEDS_MEIC_ChinaFix for each year.

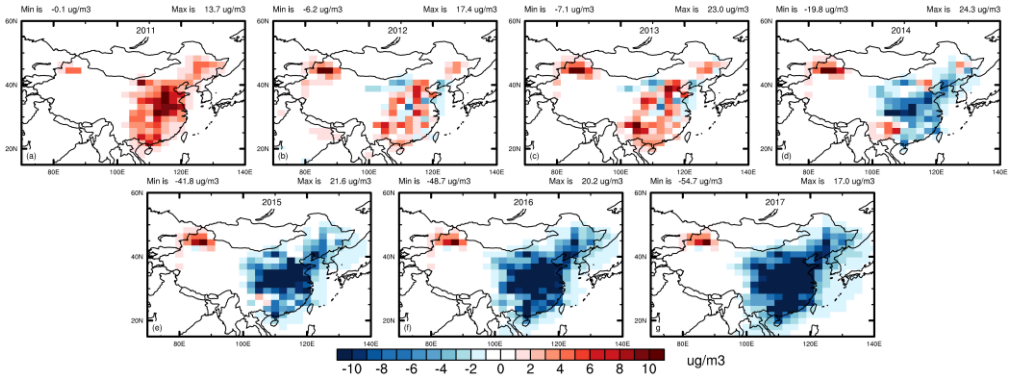

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Figure 3: Same as Fig. 2 but for annual MDA8 O_3 changes.

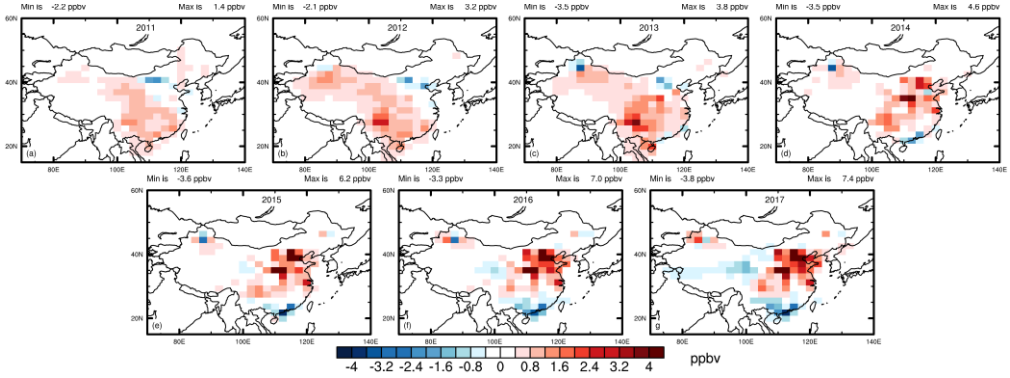

Figure 4: The annual $PM_{2.5}$ and MDA8 O_3 changes in China due to emissions and meteorological changes from 2010 to 2017.

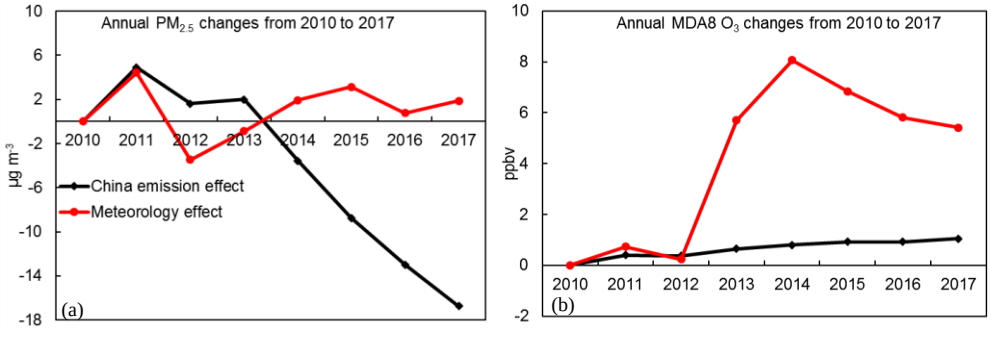

Figure 5. Spatial distribution for global tropospheric ozone burden changes from 2010 to 2017, as a result of emission changes in China.

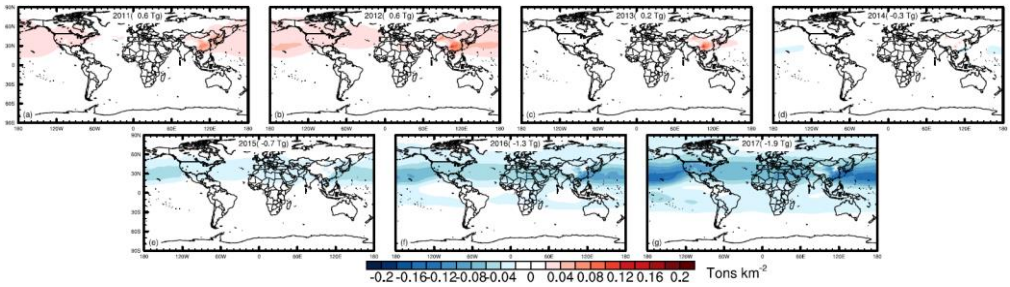

Figure 6: The annual population-weighted $PM_{2.5}$ and MDA8 O_3 changes in South Korea, Japan, and U.S. from 2010 to 2017 on caused by Chinese emission changes.