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Comparison of health and economic impacts of $PM_{2.5}$ and ozone pollution in China

Yang Xie[a], Hancheng Dai[b,*], Yanxu Zhang[c,**], Yazhen Wu[b], Tatsuya Hanaoka[d], Toshihiko Masui[d]

[a] School of Economics and Management, Beihang University, Beijing 100191, China
[b] College of Environmental Sciences and Engineering, Peking University, Beijing 100871, China
[c] School of Atmospheric Sciences, Nanjing University, Nanjing, Jiangsu, China
[d] Center for Social and Environmental Systems Research, National Institute for Environmental Studies, 16-2 Onogawa, Tsukuba-City, Ibaraki 305-8506, Japan

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ABSTRACT

Many studies have reported associations between air pollution and health impacts, but few studies have explicitly differentiated the economic effects of $PM_{2.5}$ and ozone at China's regional level. This study compares the $PM_{2.5}$ and ozone pollution-related health impacts based on an integrated approach. The research framework combines an air pollutant emission projection model (GAINS), an air quality model (GEOS-Chem), a health model using the latest exposure-response functions, medical prices and value of statistical life (VSL), and a general equilibrium model (CGE). Results show that eastern provinces in China encounter severer loss from $PM_{2.5}$ and more benefit from mitigation policy, whereas the lower income western provinces encounter severer health impacts and economic burdens due to ozone pollution, and the impact in southern and central provinces is relatively lower. In 2030, without control policies, $PM_{2.5}$ pollution could lead to losses of 2.0% in Gross Domestic Production (GDP), 210 billion Chinese Yuan (CNY) in health expenditure and a life loss of around 10,000 billion, while ozone pollution could contribute to GDP loss by 0.09% (equivalent to 78 billion CNY), 310 billion CNY in health expenditure, and a life loss of 2300 billion CNY (equivalent to 2.7% of GDP). By contrast, with control policies, the GDP and VSLs loss in 2030 attributable to ambient air pollution could be reduced significantly. We also find that the health and economic impacts of ozone pollution are significantly lower than $PM_{2.5}$, but are much more difficult to mitigate. The Chinese government should promote air pollution control policies that could jointly reduce $PM_{2.5}$ and ozone pollution.

1. Introduction

Many studies have reported associations between outdoor air pollution and morbidity and mortality (Cakmak et al., 2016; Malley et al., 2017a, 2017b; Qin et al., 2019; Silva et al., 2013). Air pollution led to not only health damage but also economic losses in China and all over the world (Burnett et al., 2018; R. Xie et al., 2016; Y. Xie et al., 2016). The latest global study attributed 8.9 million [95% confidence interval (95% CI): 7.5–10.3] deaths to $PM_{2.5}$ pollution in 2015 (Burnett et al., 2018), which is much higher than the previous estimation from Global Burden Disease. A lot of studies show the $PM_{2.5}$ is the main air pollutant in China and causes significant health impacts and economic losses (Bai et al., 2018). One city level study in China shows the $PM_{2.5}$–related death was from 0.77 million to 1.258 million by using different exposure-response functions (Maji et al., 2018). A recent study in China showed that $PM_{2.5}$ has negative impacts on human cognitive performance (X. Zhang et al., 2018). Moreover, ozone pollution also deserves attention due to its association with a series of health endpoints such as respiratory-related hospital admissions, cardiovascular disease, lost school days, restricted activity days, asthma-related emergency department visits, and premature mortality (Anenberg et al., 2017; Hubbell et al., 2005; Orru et al., 2013; Rosenthal et al., 2013; WHO, 2013). Ozone exposure is also related to respiratory symptoms and the use of asthma medication for asthmatic school children using maintenance medication (Gent et al., 2003). McDonnell et al. (1999) found long-term exposure to ozone may cause the development of asthma in adult males. Another global study showed 1.04–1.23 million respiratory deaths in adults attributable to ozone exposures (Malley et al., 2017a, 2017b). Berman et al. (2012a) evaluated the health benefits from large-scale ozone reduction, and Fann et al. (2012) estimated 4700 ozone-related deaths resulting from 2005 air quality levels and 36,000 life years are lost from ozone exposure in the United States.

* Correspondence to: H. Dai, Room 426, Environmental Building, College of Environmental Sciences and Engineering, Peking University, Beijing 100871, China.
** Correspondence to: Y. Zhang, School of Atmospheric Sciences, Nanjing University, Nanjing, Jiangsu 210023, China.
E-mail addresses: dai.hancheng@pku.edu.cn (H. Dai), zhangyx@nju.edu.cn (Y. Zhang).

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Fann and Risley (2013) estimated that monitored reductions in $PM_{2.5}$ and ozone pollution avoided premature mortalities of 22,000–60,000 and 880–4100 from 2000 to 2007 in the United States, respectively. Ozone concentration in Chinese cities is between 74 and 201 μg/ m^3 and ozone pollution leads to 74.2 thousand of premature deaths and 7.6 billion US$ in 2016 (Maji et al., 2019).

Various studies at global and regional levels have attempted to quantify the economic impacts of air pollution. Y. Xie et al. (2016) found that without mitigation, $PM_{2.5}$ pollution will lead to about 2.0% GDP loss in China in 2030. Selin et al. (2009) assessed the human health and economic impacts of projected changes in ozone pollution between 2000 and 2050, and found that health costs by 2050 will be $580 billion and mortalities from acute exposure will exceed 2 million. Matus et al. (2012) found that by improving ozone and PM pollution, the GDP in China will have increased by about 5% in 2005. A report released by OECD estimated the health and economic impacts of global outdoor air pollution up to 2060 and found that the impacts are especially substantial in Asian countries (OECD, 2016). World Bank also investigated the cost of outdoor air pollution worldwide and called for actions to mitigate air pollution.

With fast economic development and increasing use of fossil fuels, China is facing serious air pollution accompanied by severe health problems. Most current studies about health impacts in China focused on PM_{10} and $PM_{2.5}$ pollution, or ozone pollution in a single city, single province or at the national level (Zhang et al., 2006). Few studies try to quantify the economic impacts of ozone pollution at the intra-national level and compare with the impacts of $PM_{2.5}$. In China, Environmental inequality is also a problem and provincial study is necessary for the policy implication (W. Zhang et al., 2018). In this study, we make a first attempt to simultaneously focus on the health and economic impacts of both $PM_{2.5}$ and Ozone pollution at the provincial level. The health-related damages are quantified using the annual average $PM_{2.5}$ and daily maximum 8-hour ozone concentration data provided by the GEOS-Chem model and the latest exposure-response functions (ERFs), and then monetized by integrating into a computable general equilibrium (CGE) model. In this way, a picture could be drawn on how changes in ambient air pollution will affect health expenditure, labor supply, the macroeconomy and the differences between ozone and $PM_{2.5}$ pollution all over China.

2. Methods and scenario

2.1. Research framework

This study develops an integrated assessment approach to evaluate the health and economic impacts of ambient air pollution in China (Fig. 1). The research framework combines the IMED/CGE (**I**ntegrated **M**odel of **E**nergy, Environment and economy for Sustainable **D**evelopment/Computable General Equilibrium) model, the Greenhouse Gas - Air Pollution Interactions and Synergies (GAINS)-China (base year 2005) model that projects future air pollutant emissions, an air quality model (GEOS-Chem: version v 10-01; present day: 2008), and the IMED/HEL (Health) model.

2.1.1. The IMED/CGE model

The IMED/CGE model is classified as a multi-sector, multi-region, recursive dynamic CGE model that covers 22 economic commodities and corresponding sectors. The base year is 2002. It includes 30 provinces in China and is solved by the Mathematical Programming System for General Equilibrium under General Algebraic Modeling System (GAMS/MPSGE) at a one-year time step (Dai et al., 2016). The IMED/CGE model provides energy consumption data by province and sector to the GAINS model; (2) quantifies the economic impacts of health damage. The GAINS-China model provides annual regional emissions data of primary air pollutants for 30 provinces in China. The IMED/CGE model and the GAINS model have been configured extensively to reflect the historical and future pathway of China in reference (Dong et al., 2015). For instance, we adjusted the model assumptions to match the historical statistics of population growth, GDP growth rate, energy (in Fig. A3), and air pollutant emissions (in Figs. A4–A8) in each province as much as possible. As for the future, China's GDP growth and demographic evolution follow the SSP2 (Shared Socio-economic Pathways) scenario (O'Neill et al., 2013), which is characterized by moderate economic growth, a fairly rapidly growing population and lessened inequalities between west, central and east China.

2.1.2. The GEOS-Chem model

An improvement from the previous study (Y. Xie et al., 2016) is that, instead of using the concentration results in the GAINS model, the GEOS-Chem model, which is an atmospheric transport and chemistry model and much better than the simple source-receptor matrix in the GAINS model, was used to calculate the daily-maximum-8-hour-average ozone concentration and daily average concentration of $PM_{2.5}$. GEOS-Chem model has been extensively evaluated and documented in over 100 refereed journal publications, including ozone air quality of China (Qin and Xie, 2011; Qin et al., 2018; Selin et al., 2009; Wang et al., 2009; Yan et al., 2016). The model has a horizontal resolution of 0.5-degree latitude and 0.67-degree longitude. This model domain is nested in a global model simulation with a resolution of 4-degree latitude and 0.67-degree longitude, which provides initial and boundary

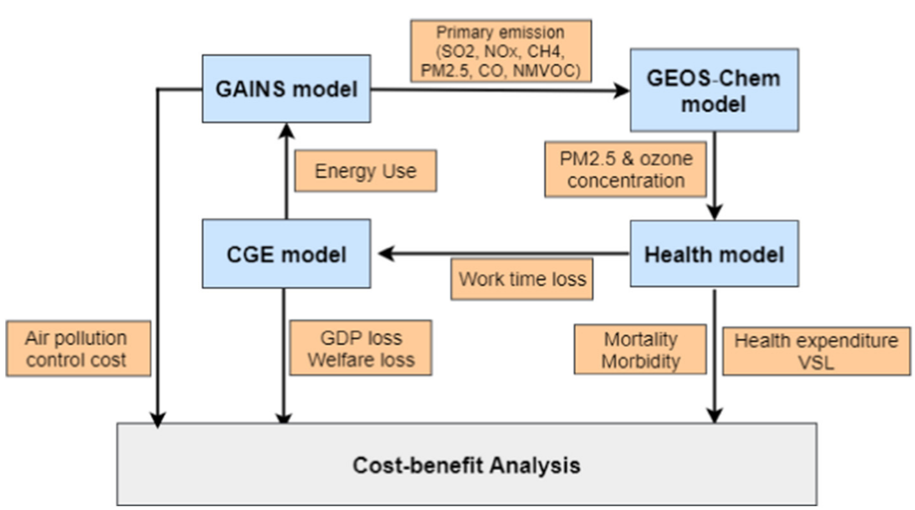

Fig. 1. Integrated research framework for assessing health and economic impacts of air pollution.

Table 1
Value of Statistical Life in 30 provinces (Unit: million CNY in 2002 price).

Region	2005	2010	2015	2020	2025	2030	2035	2040	2045	2050
Beijing	3.23	3.58	4.19	4.94	5.63	6.23	6.72	7.16	7.58	7.97
Tianjin	2.85	3.48	4.19	4.81	5.30	5.65	5.87	6.03	6.18	6.32
Hebei	1.78	2.19	2.60	3.06	3.48	3.83	4.13	4.39	4.65	4.89
Shanxi	1.65	1.96	2.36	2.78	3.17	3.50	3.77	4.02	4.25	4.46
InnerMong	1.85	2.69	3.27	3.90	4.48	4.96	5.36	5.73	6.09	6.42
Liaoning	2.17	2.78	3.41	4.14	4.82	5.44	6.00	6.55	7.11	7.68
Jilin	1.75	2.33	2.85	3.45	4.02	4.53	4.98	5.42	5.87	6.31
Heilongjiang	1.84	2.31	2.84	3.46	4.05	4.59	5.07	5.55	6.04	6.53
Shanghai	3.37	3.79	4.45	5.24	5.94	6.55	7.06	7.56	8.04	8.51
Jiangsu	2.31	2.98	3.43	3.91	4.33	4.70	5.01	5.31	5.60	5.86
Zhejiang	2.49	3.02	3.44	3.88	4.20	4.43	4.60	4.77	4.95	5.15
Anhui	1.43	1.87	2.33	2.86	3.35	3.77	4.13	4.45	4.75	5.04
Fujian	2.13	2.72	3.22	3.80	4.34	4.79	5.16	5.48	5.79	6.07
Jiangxi	1.45	1.84	2.33	2.91	3.46	3.95	4.37	4.77	5.16	5.52
Shandong	2.06	2.65	3.16	3.75	4.29	4.76	5.14	5.49	5.83	6.15
Henan	1.53	1.97	2.40	2.91	3.40	3.82	4.18	4.51	4.83	5.14
Hubei	1.61	2.09	2.58	3.15	3.70	4.19	4.62	5.02	5.41	5.78
Hunan	1.52	1.93	2.44	3.02	3.58	4.07	4.49	4.88	5.24	5.59
Guangdong	2.39	2.87	3.37	3.96	4.51	4.97	5.33	5.65	5.93	6.19
Guangxi	1.39	1.83	2.20	2.61	2.98	3.29	3.53	3.75	3.95	4.12
Hainan	1.63	2.01	2.38	2.78	3.13	3.41	3.64	3.83	4.01	4.17
Chongqing	1.60	2.07	2.59	3.15	3.68	4.14	4.54	4.92	5.28	5.63
Sichuan	1.45	1.89	2.35	2.87	3.38	3.83	4.23	4.60	4.97	5.33
Guizhou	1.08	1.42	1.74	2.12	2.49	2.83	3.13	3.41	4.33	4.78
Yunnan	1.34	1.63	1.98	2.43	2.81	3.15	3.42	3.67	3.93	4.17
Shaanxi	1.50	1.97	2.41	2.90	3.37	3.79	4.15	4.49	4.82	5.13
Gansu	1.31	1.62	1.98	2.41	2.82	3.20	4.03	4.36	4.67	4.98
Qinghai	1.52	1.90	2.31	2.75	3.17	3.53	3.83	4.11	4.40	4.67
Ningxia	1.51	1.86	2.21	2.57	2.86	3.08	3.26	3.43	3.61	3.78
Xinjiang	1.68	1.97	2.31	2.70	3.03	3.28	3.43	3.54	3.62	3.67

conditions. The size of the provinces varies drastically in China. The number of grid boxes ranges from ~600 in Xinjiang to ~10 in Beijing. We used the simple arithmetic average of the ozone concentration of all the grid boxes in a province for analysis. The model is driven by the meteorological data from the Goddard Earth Observing System (GEOS, version 5) of the NASA Global Modeling Assimilation Office (GMAO), and the meteorological data in 2008 are used for 2030 simulations (Silva et al., 2016; Silva et al., 2017). The same model setting is applied in an earlier study that focuses on China air quality, and the model performance was carefully evaluated against observation therein (Wang et al., 2017). Besides, we compared our results with earlier studies and found the results are quite consistent, even though the model setting up and used emission inventories slightly differ (Qin et al., 2017; Qin et al., 2018). In our study, we estimated the primary emissions of SO_2, NO_x, CH_4, $PM_{2.5}$ and NMVOC, and they were directly feed to the GEOS-Chem model simulations. As we mainly focus on Ozone and $PM_{2.5}$ air quality, PM_{10} was neglected because it has little impact on these two pollutants. Because BC and OC are co-emitted with $PM_{2.5}$, we scaled their emissions based on that of $PM_{2.5}$. We also included NH3 from agricultural sector using MIX emission inventory. It was included but kept constant as it was not affected by carbon mitigation. More details about this model are available in Section 6.3 of the Appendix.

2.1.3. The IMED/HEL model

The IMED/HEL model can estimate the health damage and the monetary values of $PM_{2.5}$ and ozone pollution. Exposure to incremental air pollutants leads to health impacts called health endpoints, including morbidity and mortality (all the mortality in this study means long-term exposure mortality) (Table A1 in Supplementary material). As showed in Eq. (1) in Appendix, the relative risk for the endpoint is believed to be in nonlinear relationship for $PM_{2.5}$ and linear relationship with the concentration level (Apte et al., 2015; Cakmak et al., 2016; Jerrett et al., 2009; Silva et al., 2013; Turner et al., 2016). For ozone, although some studies show ozone pollution also leads to health impacts even when the daily maximum 8-hour ozone concentration is below the widely accepted threshold value of 70 $\mu g/m^3$ (Berman et al., 2012b), there are no long-term exposure-response functions to quantify the health impacts. Therefore, we ignore the health impacts (Berman et al., 2012a; Turner et al., 2016). As showed in Tables A1 and A2, different exposure-response functions are used for ozone and $PM_{2.5}$. Our main purpose is to compare the health burden in different air pollution control policy. The morbidity, health expenditure and work loss days are the main indicators. So in this study, we didn't pay more attention on the cause-specific mortality. We use non-linear all-cause mortality in our study (Apte et al., 2015) for $PM_{2.5}$ and linear concentration-response functions for ozone (Burnett et al., 2018; Malley et al., 2017a, 2017b; Turner et al., 2016)

The method to calculate work loss time and health expenditure is described in Eqs. (2)–(7) in the Appendix. We adopt the concentration-response function from (Bickel et al., 2005). For $PM_{2.5}$, we used the relationship "Change of 207 work loss days (WLDs) (95% CI 176–238) per 10 $\mu g/m^3$ per year per 1000 people aged 15–64 in the general population". For the ozone pollution, there is no such concentration-response function, so we transformed the Minor restricted activity days (MRADs) "Increase in MRADs = 115 (95% CI 44, 186) per 10 $\mu g/m^3$ ozone (8-h daily average) per 1000 adults aged 18–64 per year" for ozone to the work loss days according the relationship for $PM_{2.5}$, which could instead the work loss from ozone pollution. The annual total medical expenditure and per capita work loss could be converted from the health impacts and used as a variation of the household expenditure and labor participation rate in the CGE model, which quantifies the macroeconomic impacts.

Furthermore, we monetize the non-market value of statistical life lost to reflect additional impacts from air pollution reduction based on the method developed by West et al. (2013), which could represent the majority of the benefit of air pollution control policy (Eq. (8) in the Appendix). In literature, the value of life ranges from 8.2 to 31.1 million USD (Matus et al., 2012), but here we adopt the latest value of statistical life of about $250,000 USD from empirical investigations using willingness to pay method in China (Jin, 2017; Xie, 2011), and VSLs in

Table 2
Scenario setting.

Scenario	Description
Reference	Ignore the health impacts of air pollution, health service cost, premature deaths and work loss days
woPol	The penetration rate of mitigation technology is fixed to the 2005 level
wPol	Various air-pollution-control technologies are used to reduce pollutant emissions and air pollutants concentration to levels below the woPol scenario
wPol2	Further reduce the emissions in 2030 of NO_x, VOC, CO by 50% and CH_4 by 20% from the wPol scenario

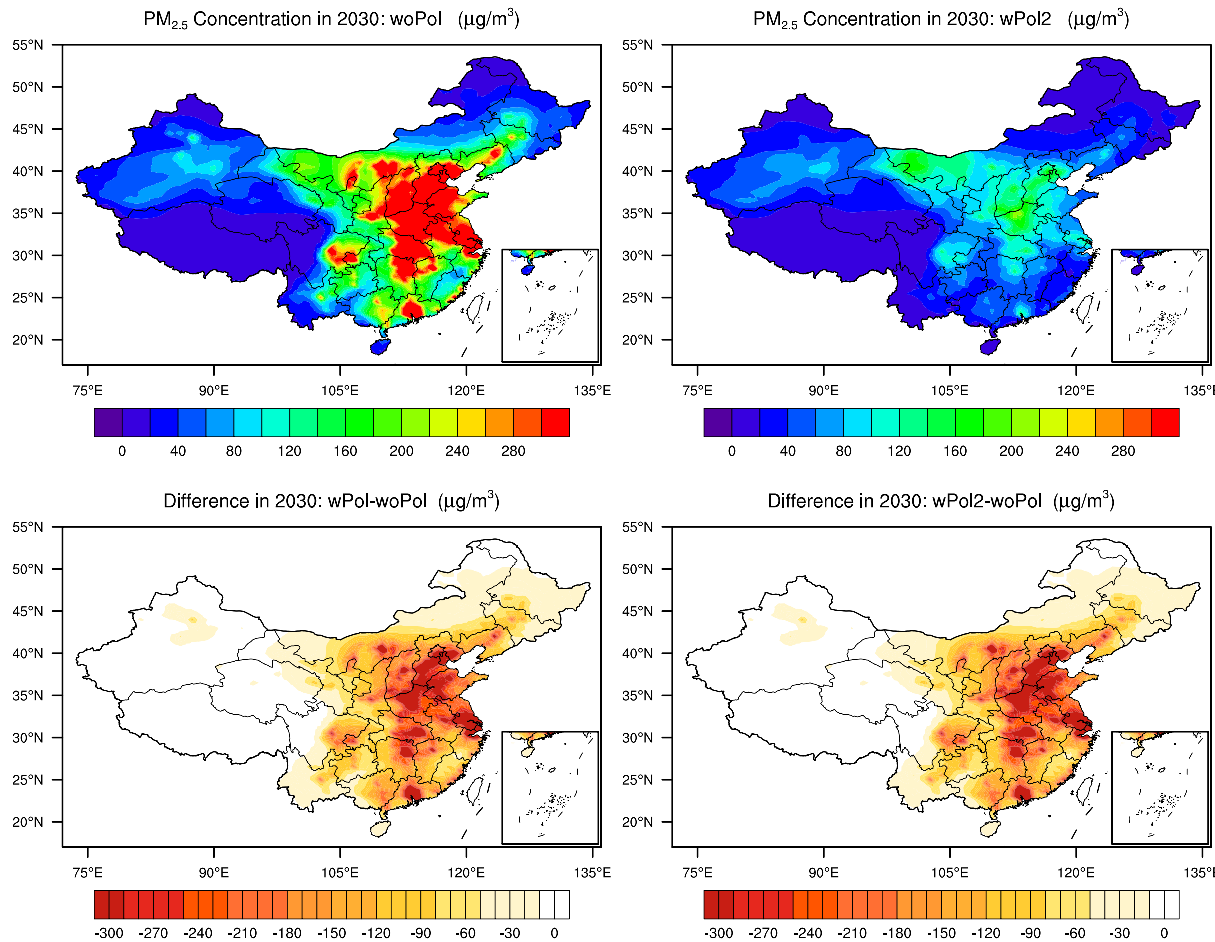

Fig. 2. Annual average $PM_{2.5}$ concentration in woPol and wPol scenarios (upper) and change from woPol to wPol and wPol2 scenarios (lower).

all provinces are adjusted using their GDP per capita values relative to the national average per capita GDP in 2010(Table 1) and an elasticity of 0.5 (Viscusi and Aldy, 2003).

A more detailed introduction to the IMED/HEL model, the CGE model, the GAINS-China model and GEOS-Chem model is provided in the Appendix. Furthermore, since the IMED models are continuously updated and documented, and the up-to-date introduction is available at http://scholar.pku.edu.cn/hanchengdai/imed_general.

2.2. Scenario

Four scenarios are established in this study (Table 2): reference, woPol, wPol and wPol2 scenarios. The reference scenario provides the economic results in the CGE model without coupling it with the IMED/HEL model, which means that $PM_{2.5}$ and ozone pollution-related health impacts are ignored such that air pollution will not cause additional health service cost, premature deaths, or work loss days. This scenario is an ideal situation that does not exist. However, its role is to compare with the other scenarios and evaluate the negative impacts of pollution and the benefits of pollution control.

The remaining three scenarios couple the IMED/HEL model with the IMED/CGE model to capture the macroeconomic impacts of the health effects. The woPol scenario assumes that the penetration rate of mitigation technology is fixed to the 2005 level, implying that the emissions from additional energy combustion will be uncontrolled in the future. It is meant to show the impacts of pollution control policies rather than represents reality.

The wPol scenario takes China's current air pollution policies into account. Furthermore, the sectoral and provincial differences in emission limit values and time of their introduction are considered as well. Therefore, various air-pollution-control technologies are used to reduce pollutant emissions and air pollutants concentration to levels below the woPol scenario. More details of the technology settings are in the Appendix.

We also set up the wPol2 scenario, in which more aggressive air pollutant control technologies are adopted to further reduce the emissions in 2030 of NO_x, VOC, CO by 50% and CH_4 by 20% from the wPol scenario. This scenario is meant to explore the additional potential of mitigation effects, especially with regard to ozone pollution.

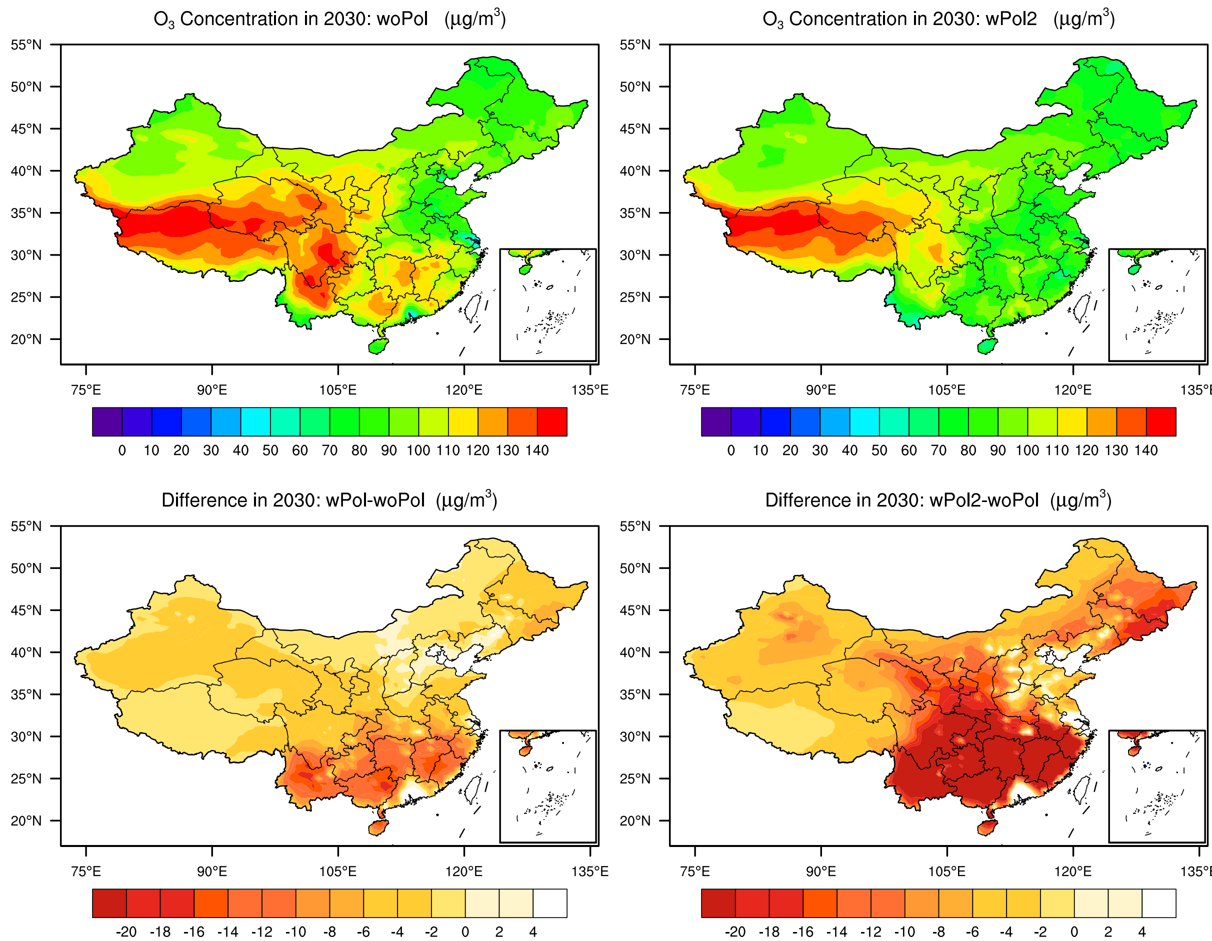

Fig. 3. Daily maximum 8-hour ozone concentration in woPol and wPol scenarios (upper) and change from woPol to wPol and wPol2 scenarios (lower).

3. Results

3.1. Air pollutants emissions and concentration

Results show that, unsurprisingly, air pollutants emissions in the wPol scenario are much lower than those in the woPol scenario in 2030, and the wPol2 scenario exhibits significant further reduction (Fig. A4 in Appendix). For instance, NO_x emissions will rise to 32 million ton in 2030 without control in woPol scenario, while with control in the wPol and wPol2 scenarios, they will be reduced to 24 and 12 million ton in 2030, respectively. VOC emissions will increase from 16 million ton in 2000 to 30 million ton in 2030 in the woPol scenario, and decline to 20 and 10 million ton in wPol and wPol2 scenario in 2030, respectively. Using these emission pathways as inputs for the GEOS-Chem model, $PM_{2.5}$ annual average concentration and the daily maximum 8-hour mean ozone concentration is simulated in 30 provinces of China in 2030 (Figs. 2 and 3).

$PM_{2.5}$ concentration demonstrates a similar trend as the emissions. It shows that $PM_{2.5}$ concentration is higher in the east part of China in woPol scenario in 2030, especially in Tianjin (420 μg/m^3), Beijing (380 μg/m^3), Hebei (350 μg/m^3) and Henan (360 μg/m^3). Figs. 2 and 3 (lower two panels) also show that the $PM_{2.5}$ concentration could be reduced significantly under policy scenarios. In the heavy polluted provinces such as Hebei, Tianjin, Henan and Shandong, $PM_{2.5}$ concentration will reduce by about 75% in 2030.

In contrast to geographical distribution of $PM_{2.5}$ pollution, the daily maximum 8-hour ozone concentration in 2030 is higher in southwest China and lower in east China with the highest level in the southwest provinces such as Sichuan (130 μg/m^3), Qinghai (130 μg/m^3), and Gansu (120 μg/m^3) provinces in the WoPol scenario. In the populous regions like Beijing (96 μg/m^3), Tianjin (78 μg/m^3), and Jiangsu (75 μg/m^3), ozone concentration is high enough to cause various health impacts, and only Hainan (66 μg/m^3) and Shanghai (66 μg/m^3) could meet the national standard of 70 μg/m^3. Even under the intensive air pollution control strategy, the ozone concentration could only decline slightly in China.

Different from the formation of secondary $PM_{2.5}$, the relationship between reduction in ozone precursors emissions and concentration is not linear. In the wPol scenario, although air pollutants emission reduction is over 50%, daily maximum 8-hour ozone concentration will not decrease at a similar magnitude. Provinces such as Hunan, Anhui will witness the most significant reduction, albeit only falling by < 10%. Moreover, there is no significant reduction in Hebei, Shanxi or Inner Mongolia. Conversely, daily maximum 8-hour ozone concentration will increase in Beijing, Shanghai, and parts of Guangdong in the wPol scenario. Note that we are using the same meteorological data in 2008 and 2030 simulations. Therefore, all the changes are caused by changes in anthropogenic emissions. These patterns, especially the different signs of ozone concentration changes responding to anthropogenic emissions changes, are resulted from the different ozone formation regimes these provinces are located. The great metropolitan regions such as Beijing, Shanghai, and Guangzhou City in Guangdong Province are generally VOC-controlled, and the declining NOx emissions in wPol and wPol2 scenarios will reduce the ozone destruction rate by reacting with NOx and thus increase ozone concentration (Chou et al., 2011; Xue et al., 2014).

Considering that ozone concentration is much higher in the daytime, we also evaluated the impact of anthropogenic emission changes in the 24-hour average ozone concentration (Fig. A10). The response of 24-hour average ozone concentration is significantly different from the daily maximum 8-hour in that the former has percentage changes toward the positive axis. In regions with increased concentration, the changes in concentration are more prominent when using a 24-hour average metric rather than the daily maximum 8-hour. However, in regions with decreased concentration (such as Beijing, Shanghai, and Guangzhou), the magnitude of changes becomes less significant. This pattern is largely associated with the diurnal cycles of ozone formation and removal. While the daily maximum 8-hour concentration mainly represents the daytime when active ozone production is occurring, the 24-hour average is also influenced by the nighttime condition when photochemical ozone formation ceased, and anthropogenic NOx emissions efficiently destruct ozone. Therefore, decreasing anthropogenic emissions largely increase nighttime ozone concentration (Zhang et al., 2004).

3.2. Health impacts

3.2.1. Health impacts attributable to $PM_{2.5}$ pollution

Mortality attributable to $PM_{2.5}$ is 9.2 million and 2.3 million in woPol and wPol scenario in 2030 in China, respectively, which means the air pollution mitigation policy could reduce 6.9 million premature deaths in China in 2030. At the provincial level, provinces with higher population density will suffer more mortality. The mortality in the top five populous provinces of Henan, Shandong, Jiangsu, Hebei and Guangdong will be 1500, 960, 1100, 830 and 690 thousand people in woPol scenario, respectively. However, provinces with severer air pollution and higher population density will have more benefit from air quality improvement. In wPol scenario, the mortality is 270, 200, 170, 140 and 220 thousand people in Henan, Shandong, Jiangsu, Hebei and Guangdong provinces in 2030, respectively. In province with good air quality such as Hainan, mortality is only 16 thousand in woPol scenario and 3.4 thousand in wPol scenario in 2030. In 2005, total morbidity from $PM_{2.5}$ pollution was about 140 million cases in WoPol scenario. It will increase to 230 million cases in 2030 in WoPol scenario and 70 million in wPol scenario.

Premature deaths among labor force cohort aged between 15 and 65 years old will reduce labor supply and total work time. For $PM_{2.5}$, the national average per capita work time loss in 2030 will reach 56 h (2.7% of annual total annual work hours) in the woPol scenario. The $PM_{2.5}$ reduction in the wPol scenario proves to be quite effective in reducing work time loss, to 15 h (0.71% of annual work hours) in 2030. The provincial disparity in the per capita work time loss is consistent with the provincial disparity in $PM_{2.5}$ concentration. In the woPol scenario, Tianjin (98 h, or 4.7%), Henan (83 h, or 4.0%), Shanghai (99 h or 4.7%), Hebei (82 h or 4.0%), and Beijing (88 h or 4.2%) have the highest annual per capita work time loss in 2030. By contrast, the provinces with the highest work time loss in the wPol scenario in 2030 switch to Chongqing (24 h, 1.1%), Henan (23 h, 1.1%), Sichuan (22 h, 1.1%), Hunan (18 h, 0.88%), and Hubei (18 h, 0.89%).

3.2.2. Health impacts attributable to ozone pollution

As Fig. 4 shows, the mortality of ozone pollution is much lower than $PM_{2.5}$, while morbidity is much higher than $PM_{2.5}$. In 2030, the national total number of mortality is about 583 (95% CI: 190–980) thousand persons in woPol scenario. At the provincial level, Sichuan (74 thousand people per year), Gansu (18), Shaanxi (23) and Hunan (47) will encounter most of the ozone-related mortality in the woPol scenario. Nevertheless, mitigation benefits from air pollution control policies are significant (Fig. 4 right column). Mortality reaches 491 and 340 thousand persons in the wPol and wPol2 scenarios, respectively, and air pollution control policy will lead to a decrease in mortality by 92 thousand persons in the wPol scenario and 240 thousand persons in the wPol2 scenario. Meanwhile, ozone-related morbidity consists of coughs, asthma, bronchodilator usage, lower respiratory symptoms, and respiratory-related hospital admissions (Table A1). West and central provinces such as Sichuan, Qinghai, Jiangxi, Hunan and Chongqing encounter higher morbidity. People in these provinces have annual risk rates of about 4–5% in suffering from health effects such as asthma attacks, respiratory hospital admission, allergic rhinitis, acute respiratory symptoms and coughs from ozone exposure. In contrast, provinces in the eastern parts of China, for instance, Tianjin, Jiangsu, Beijing and Shandong where ozone concentration levels are lower, are at a lower risk (about 1–2%) of suffering from such adverse health effects caused by ozone exposure.

With regard to work loss days, however, there is no concentration-response function about work loss days for ozone exposure in the literature. Therefore, in this study, we converted minor restricted activity days of ozone into work loss days based on the relationship of $PM_{2.5}$, e.g., minor restricted activity days are 2.78 times work loss days. Fig. 5 shows the per capita work loss hours due to morbidity and cumulative mortality. In 2030, the national average per capita work loss is 2.8, 2.4 and 2.0 h in the woPol, wPol and wPol2 scenarios in China, respectively. At the provincial level, Qinghai, Sichuan, Gansu and Xinjiang will encounter more work loss hours in the woPol scenario, about 5.7, 5.5, 4.3 and 3.3 h, respectively. The recovered work loss in the wPol scenario ranges from 0.8 h in Jiangxi Province to −0.3 (increase) hour in Beijing.

3.3. Economic impacts of ambient air pollution

Fig. 4 (the bottom two rows) and Fig. 6 show the economic losses due to ambient air pollution-related health impacts, including health expenditure, the value of life lost, GDP loss and welfare loss.

3.3.1. Medical expenditure

In 2005, China paid an additional 37 billion CNY (2002 constant price) on the $PM_{2.5}$ pollution-related health problem in woPol scenario, and in 2030 it will increase to 210 billion CNY. While in the wPol scenario, the additional health expenditure is reduced to about 15 billion CNY in 2005 and 53 billion CNY in 2030. In 2005, Jiangsu, Shandong, Henan, Guangdong, Hebei and Beijing encountered higher total expenditure, which was 4.4, 3.5, 2.6, 2.6, 2.5 and 2.3 billion CNY in woPol scenario, respectively. In 2030, Sichuan, Shandong, Henan, Hebei, Anhui and Jiangsu will encounter higher total expenditure, amounting to 24, 18, 17, 14, 13 and 13 billion CNY in 2005 in woPol scenario and slipping to 7.6, 4.0, 4.5, 2.9, 3.7 and 2.9 billion CNY in wPol scenario, respectively.

It worth noticing that although $PM_{2.5}$ pollution seemingly poses more serious health damage and burden than ozone when looking at indicators such as mortality, the health expenditure on ozone exposure-related health problem is not that low. In 2030, the expenditure is estimated to be 100, 87 and 58 billion CNY in the woPol, wPol and wPol2 scenarios in China, respectively, which is comparable to $PM_{2.5}$ related medical expenditure. The top five provinces account for the majority of the health expenditure in the woPol scenario, including Sichuan (20 billion CNY), Hunan (11 billion CNY), Jiangxi (6.9 billion CNY), Gansu (4.8 billion CNY), and Hubei (4.9 billion CNY), all of which are relatively less developed provinces in China. This implies that ozone pollution will become a non-ignorable economic burden to residents living in the low- and mid-income provinces. Moreover, reduction rates of total expenditure in these five provinces in the wPol scenario are as follows: −8.6% (Sichuan), −26% (Hunan), −7.1% (Gansu), −21% (Hubei), −3.5% (Qinghai).

3.3.2. GDP loss and welfare loss

Both labor supply loss and medical expenditure increase will affect the macroeconomic indicators in terms of GDP and residential welfare. It turns out that ozone pollution will lead to much lower macroeconomic impacts than that of $PM_{2.5}$ pollution. Moreover, the GDP loss due to both $PM_{2.5}$ and ozone pollution in the woPol scenario in our

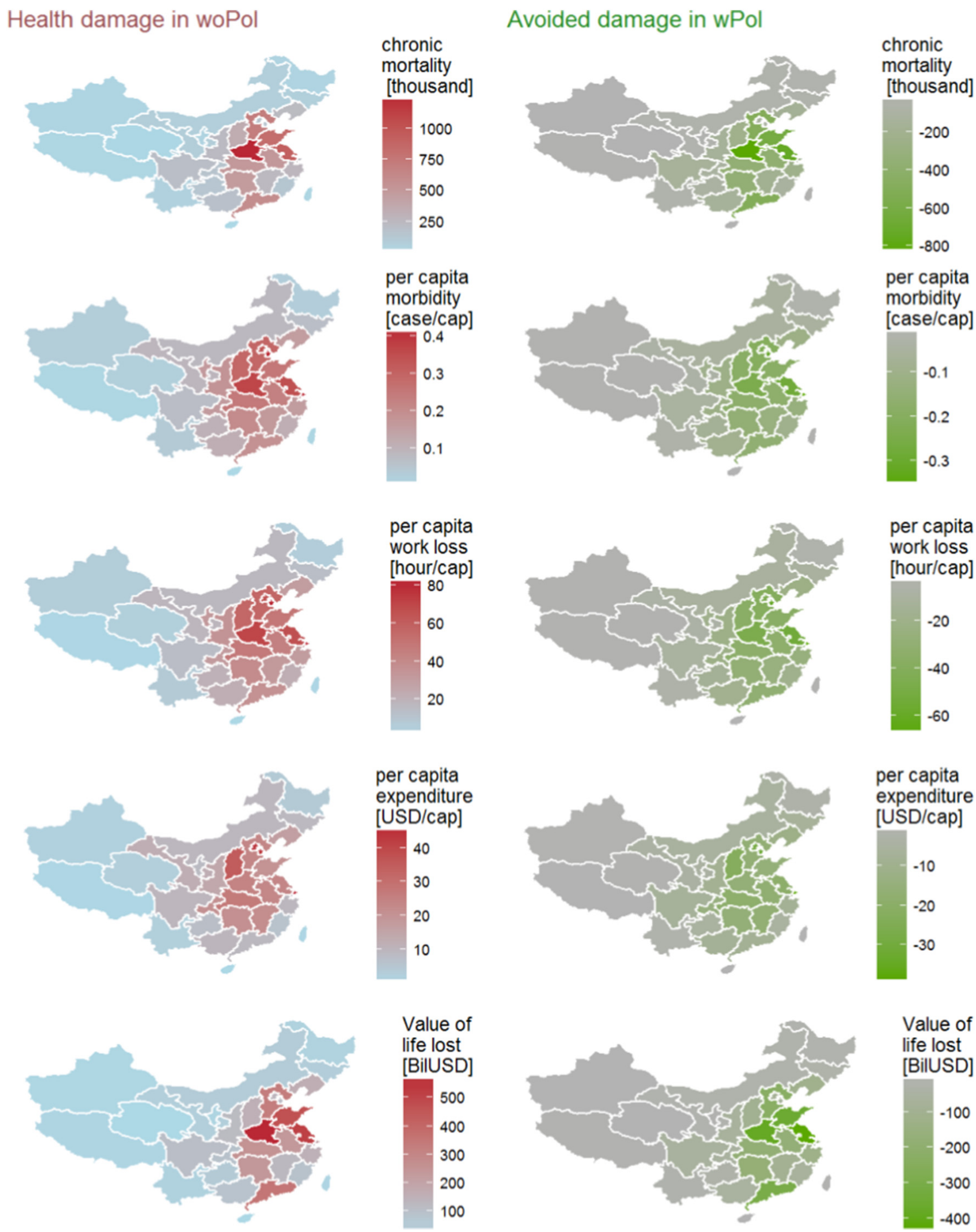

Fig. 4. Health damage due to PM2.5 pollution (left) and benefit of mitigation (right).

study is comparable to that reported by the OECD[14] (2.6% in 2060). Matus et al. (2008) used a CGE model to estimate the benefits of air pollution control in the USA, and found that the benefits rose steadily from 1975 to 2000 from 50 billion USD to 400 billion USD (from 2.1% to 7.6% of market consumption).

In the woPol scenario, $PM_{2.5}$ pollution will cost China GDP loss of 2.0% and the health expenditure of 210 billion CNY in 2030. By contrast, with control policy in the wPol scenario, a control cost of 830 billion CNY (0.79% of GDP) is contrasted by a projected gain of 1.2% of GDP in China from improving air pollution. As for the welfare loss, which is defined as total consumption change measured by Hicks' equivalent variation (Fujimori et al., 2015), China experiences 2.7% and 0.63% welfare loss in woPol and wPol scenarios in 2030, respectively, higher than GDP loss.

As indicated in Fig. 6, in 2030, China will experience a GDP loss of about 0.09% in the woPol scenario, 0.08% in the wPol scenario and 0.07% in the wPol2 scenario due to ozone pollution. At the provincial level, provinces in the west and southwest will experience higher GDP losses, for example, Qinghai (0.23%, 0.22% and 0.21% in the woPol, wPol and wPol2 scenarios, respectively), Sichuan (0.22%, 0.29% and 0.18%), Gansu (0.17%, 0.16% and 0.15%), Ningxia (0.15%, 0.14% and 0.13%), and Hunan (0.15%, 0.13% and 0.11%). By contrast, Hainan

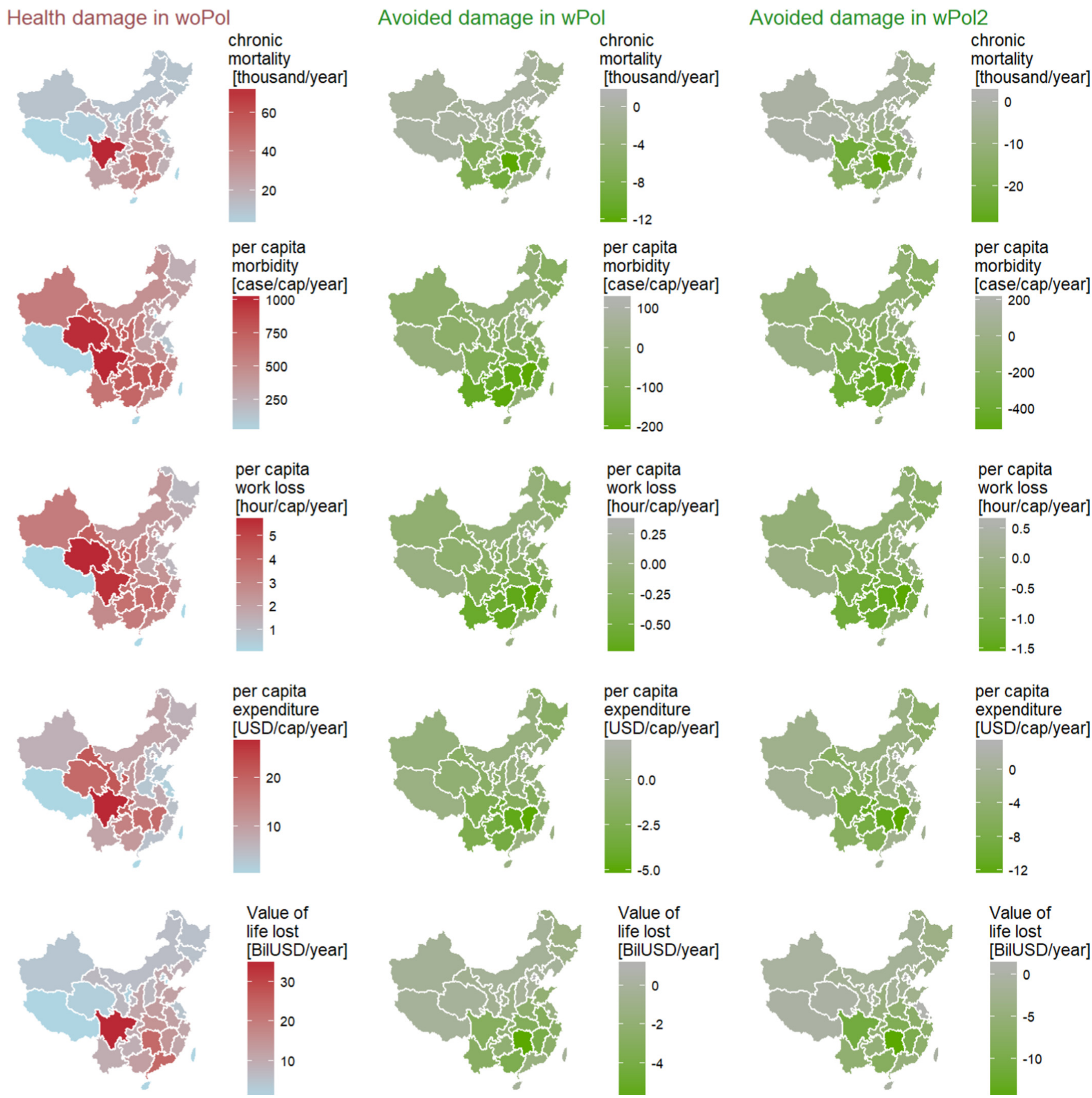

Fig. 5. Health damage due to ozone pollution (left/red) and the benefit of mitigation in 2030 (right/green). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

will experience almost ignorable GDP loss from ozone pollution, about 0.002% in the woPol (0.002% in the wPol scenario). GDP loss is moderate in other regions: 0.09% (0.10%) in Beijing, 0.04% (0.05%) in Tianjin, and 0.05% (0.04%) in Jiangsu. Furthermore, welfare loss from ozone-related health impacts in 2030 is about 0.15%, 0.13% and 0.12% in the woPol, wPol and wPol2 scenarios, respectively. Welfare loss is higher in provinces such as Qinghai (0.46%, 0.45% and 0.43%), Ningxia (0.34%, 0.32% and 0.28%), Sichuan (0.31%, 0.30% and 0.27%) in the woPol, wPol and wPol2 scenarios in 2030, respectively. These provinces are in the west of China, where ozone from natural sources is quite high. The difference between the two scenarios is not significant.

3.3.3. *Value of statistical life lost*

The benefits of avoided air pollution mortality are monetized using the value of statistical life (VSLs). In China, the VSL from $PM_{2.5}$ pollution is 38,000, 10,000 and 9300 billion CNY in woPol, wPol and wPol2 scenario, respectively, which is about 38%, 10% and 9.3% of GDP in 2030. At the provincial level, VSL is higher in Shandong, Jiangsu, Hebei, Guangdong and Sichuan province, which is about 4200, 3600, 2700, 2700 and 2600 billion CNY in woPol scenario in 2030, respectively.

For ozone, the national VSL is about 2300 and 2000 billion CNY respectively in the woPol and wPol scenarios, which is about 2.7% and 2.3% of GDP. In wPol2 scenario, the VSL is quite similar to wPol scenario. At the provincial level, Sichuan has the highest mortality and

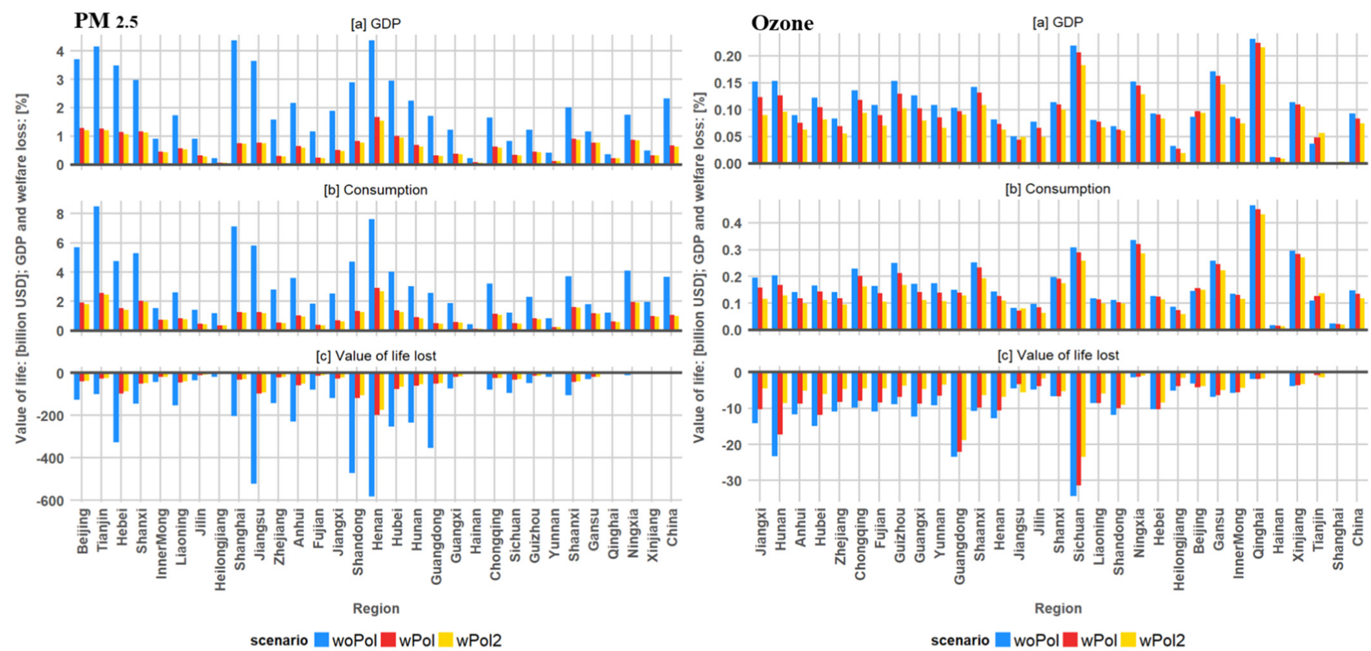

Fig. 6. GDP loss, welfare loss and value of statistical life lost due to $PM_{2.5}$ (left) and ozone (right) pollution in 30 provinces in 2030.

moderate per capita GDP and the VSL is the highest (280 billion CNY, or 8.2% of GDP in woPol), followed by Hunan (190 billion CNY, or 7.8%), Jiangxi (120 billion CNY, or 6.0%), the western provinces of Gansu (57 billion CNY, or 4.7%), Qinghai (17 billion CNY, or 6.1%).

4. Discussion

4.1. Comparative impacts of $PM_{2.5}$ versus ozone

The national impacts of ozone and $PM_{2.5}$ pollution are summarized in Fig. 7. It is found that health and economic impacts from ozone are much smaller than $PM_{2.5}$ pollution except for per capita morbidity and expenditure. Taking the wPol scenario in 2030 for example, the total mortality is 2.4 million attributable to $PM_{2.5}$ pollution, while the total mortality due to ozone is 0.49 million. Moreover, per capita work loss is only 2.4 h from ozone while 18 h due to $PM_{2.5}$. Conversely, upper respiratory symptoms dominate PM-related endpoints while the overwhelming endpoints related to ozone are bronchodilator usage and weaker respiratory symptoms. As a result, the per capita morbidity caused by ozone (4.2% per capita per year) is nearly 10 times that of $PM_{2.5}$ (0.5% per capita per year) mainly due to bronchodilator usage. Consequently, per capita expenditure due to ozone pollution is 87 CNY, which is much higher than that caused by $PM_{2.5}$ (40 CNY). Furthermore, ozone causes less GDP loss (0.08%) than $PM_{2.5}$ (0.6% in the wPol scenario and 2.3% in the woPol scenario). One city level study in China shows the $PM_{2.5}$-related death was from 0.77 million to 1.258 million by using different exposure-response functions in 2016. Our result is comparable with this estimation in 2015 in the wPol scenario (Maji et al., 2018). Another study at the Chinese cities shows ozone pollution leads to 74.2 thousand of premature deaths and 7.6 billion US$ economic losses in 2016 in China. Our result is comparable with their study (Maji et al., 2019).

In terms of spatial and temporal features, $PM_{2.5}$ and ozone concentration also vary by region and by season. $PM_{2.5}$ concentration are much higher in densely populated areas, while daily maximum 8-hour ozone concentration is higher in relatively low populated western provinces. Furthermore, ozone concentration are higher in summer (due to the higher active reaction of photochemical production) but lower in winter in most provinces and cities, dominated by zero-out ozone in some provinces while by anthropogenic sources elsewhere (Fig. A9). In accordance with the features of ozone concentration distribution, ozone-related health impacts are more severe in the western provinces with higher daily maximum 8-hour ozone concentration and moderate population density. The provinces of Qinghai, Sichuan, Gansu and Jiangxi suffer from higher per capita morbidity, more work hour loss and higher economic impacts. In contrast, health impacts are lower in east China, where the population density is much higher than in the west. The provinces in the southwest and northwest experience higher GDP loss and welfare loss due to ozone pollution. At the same time, these provinces are relatively less developed and have less motivation to control ozone pollution.

4.2. Policy implications

China is suffering from severe ambient air pollution. Ideally, air pollution control policy aimed at reducing primary emissions such as NO_x, SO_x and VOC should improve $PM_{2.5}$ and ozone pollution simultaneously. However, we find that it is more difficult to reduce daily maximum 8-hour ozone concentration compared with $PM_{2.5}$ (Fig. A4) because the ozone generation process is not in a linear relationship with precursor emissions. Although ozone precursor emissions have been reduced a lot in the wPol scenario (Fig. A4), the daily maximum 8-hour ozone concentration reduction is quite limited (< 10%). Even more aggressive reduction efforts are made in the wPol2 scenario, in contrast to $PM_{2.5}$ whose daily concentration will be reduced by over 70% in almost all provinces, reduction rates of daily maximum 8-hour ozone concentration are merely around 20% in most provinces. Conversely, it even increases in urban areas around Beijing, Shanghai and Guangzhou. This implies that in the longer term, ozone pollution will be a more persistent air pollution problem.

A similar phenomenon has been reported in previous studies in China. For instance, Chou et al. (2011) found that the mixing ratio of ozone increased with the increasing NO_2/NO ratio, whereas the NO_z mixing ratio leveled off when $NO_2/NO > 8$. Consequently, the ratio of ozone to NO_z increased to above 10, indicating the shift from a VOC-sensitive regime to a NO_x-sensitive regime. Xue et al. (2014) found varying and considerable impacts of ozone generation processes in different areas of China depending on the atmospheric abundances of

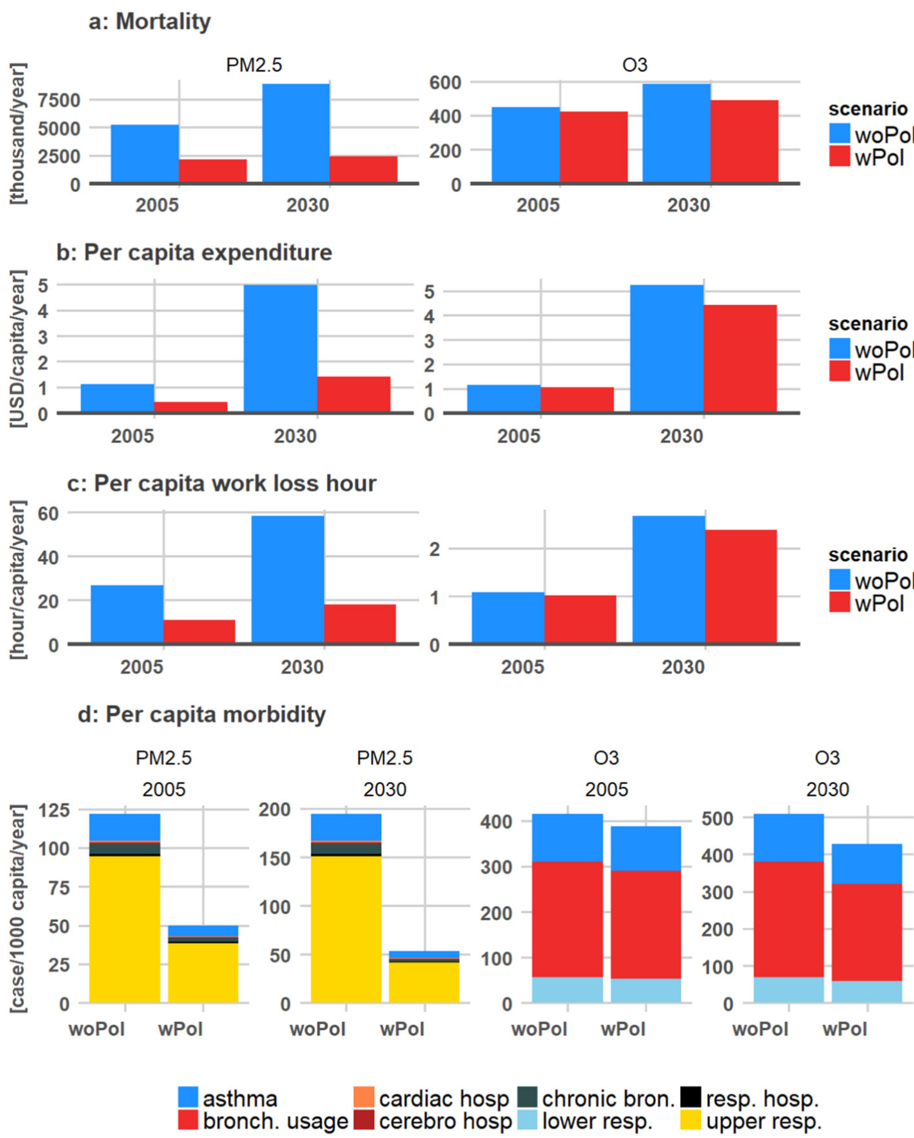

Fig. 7. Comparing national health impacts between $PM_{2.5}$ and ozone.

aerosol and NO_x. This is partly because most of $PM_{2.5}$ is from anthropogenic activities like industry and transportation sectors. As the emissions of sulfur dioxide decline significantly as a result of China's strict control policies (e.g., in the power generation sector), the share of NO_x emissions from the transportation sector will rise in the future, which will enhance the ozone pollution in the urban regions. Moreover, a significant source of ozone is natural emissions that are beyond the control of human activity, especially in the western provinces.

Such unintended features of ozone have implications for residential behavior in order to avoid adverse health impacts. One study from WHO shows human exposure to ozone during the winter is reduced because more time is spent indoors. Moreover, building structures and slow rates of ventilation will reduce ozone penetration indoors even during the summer (Amann, 2008). Therefore, the government and residents should act interactively. For instance, the government should provide daily public information about air quality, and the public should adjust their lifestyles according to the air quality information.

4.3. Limitation, uncertainty and future work

Despite the efforts of quantifying the health and economic impacts of ambient ozone and $PM_{2.5}$ pollution in this study, there are some limitations and uncertainties, which need further investigation. Uncertainty within our framework is classified into three sources. The first source is the uncertainty of future economic development and energy consumption in the CGE model. CGE model can capture the economic impact of air pollution-related health impacts. It could reflect market impact and give more detail on the output change, labor force price and so on. However, comparing with other approaches, such as Willingness to Pay, CGE might underestimate the adverse impact of air pollution, because so far, the CGE model can only quantify the impact of labor supply reduction and cannot include the adverse impact on suffering from air pollution-related health problem, such as stress, uncomfortable feeling and so on. The second source is the estimation of future air pollutant emissions and ozone concentration, which is related to both technology selection and the behavior of the GEOS-Chem model. The last source is related to ERFs used in the IMED/HEL model. In terms of uncertainty of ERFs, the numbers in the parenthesis show 95% CI of ERFs. Besides these uncertainties, climate change also has impacts on future ozone air quality and could have intersection effect on ozone precursor emissions. But in this study, we do not quantify the magnitude of the impact without a detailed model analysis.

Furthermore, many epidemiological studies show exposure to higher ozone concentration not only leads to health problem, but also reductions in labor productivity (Brauer et al., 1996; Korrick et al., 1998). However, the effects on productivity cannot be quantified in this study, implying that the current economic impact of ozone pollution could be underestimated. Besides, our results may be underestimated because we neglect mortality among those younger than 30, including effects on children and neonatal effects (West et al., 2013). In our study, we didn't simulate the contribution from ammonia emissions reduction from agricultural sector. As the Giannadaki et al. pointed the reducing agricultural emissions by 50% lead to economic benefit of many billions US$ (Giannadaki et al., 2018; Lelieveld et al., 2015). However, in our simulation we didn't consider this part. Last but not least, as noted in the supplementary information, there are no ERFs for work loss days for ozone, and as the second-best approach we converted it from the restricted activity day, which leads to uncertainties concerning the quantifying of the market economic impacts in the CGE model. We expect future epidemic studies could fill this gap.

5. Conclusion

Air pollution could affect human health and economic welfare significantly, and pollution control could bring substantial benefits. This study provides a comparative assessment of such impacts resulted from $PM_{2.5}$ and ozone pollution. We find that $PM_{2.5}$ pollution causes much higher health and economic impacts than ozone, especially in east China. However, it is more difficult to combat ozone pollution in the long term. For instance, in 2030, $PM_{2.5}$ annual average concentration could decline by around 75% under the strictest mitigation scenario, whereas ozone concentration reduction is merely around 20%. As a result, mortality could be reduced by as much as 6.9 million, most of which are from $PM_{2.5}$. Mitigation could also increase work time by 41 and 0.1 h, and save health expenditure by 157 billion CNY and 50 billion CNY from $PM_{2.5}$ and ozone control, respectively.

Air pollution control is found to be cost-effective at the national level, although the situation could be different among provinces. At the national level, the benefit is much higher than air pollution control cost. In contrast to control cost of 830 billion CNY, the national net benefit is about 1.3% of GDP in 2030. Nonetheless, a closer look at the provincial situation reveals a mixed picture. The eastern provinces of China with higher population density and heavy pollution will receive higher benefit from air quality mitigation policy, such as Tianjin, Hebei, Shandong. By contrast, southern and western provinces such as Gansu, Qinghai and Yunnan will have lower benefit than cost.

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Appendix A. Supplementary data

The Supporting Information shows additional results of primary emissions and seasonal daily maximum 8-hour and 24-hour ozone concentration. It also introduces the IMED/HEL model, CGE model, GEOS-Chem model, and the GAINS-China model. Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2019.05.075.

References

Amann, M., 2008. Health Risks of Ozone From Long-range Transboundary Air Pollution. WHO Regional Office Europe.
Anenberg, S.C., Miller, J., Minjares, R., Du, L., Henze, D.K., Lacey, F., et al., 2017. Impacts and mitigation of excess diesel-related NOx emissions in 11 major vehicle markets. Nature 545, 467–471.
Apte, J.S., Marshall, J.D., Cohen, A.J., Brauer, M., 2015. Addressing global mortality from ambient PM2.5. Environ. Sci. Technol. 49, 8057–8066.
Bai, R., Lam, J.C.K., Li, V.O.K., 2018. A review on health cost accounting of air pollution in China. Environ. Int. 120, 279–294.
Berman, J.D., Fann, N., Hollingsworth, J.W., Pinkelton, K.E., Rom, W.N., Szema, A.M., et al., 2012a. Health benefits from large-scale ozone reduction in the United States. Environ. Health Perspect. 120, 1404.
Berman, J.D., Fann, N., Hollingsworth, J.W., Pinkerton, K.E., Rom, W.N., Szema, A.M., et al., 2012b. Health benefits from large-scale ozone reduction in the United States. Environ. Health Perspect. 120, 1404–1410.
Bickel, P., Friedrich, R., Droste-Franke, B., Bachmann, T., Gressmann, A., Rabl, A., et al., 2005. Externe Externalities of Energy Methodology 2005 Update.
Brauer, M., Blair, J., Vedal, S., 1996. Effect of ambient ozone exposure on lung function in farm workers. Am. J. Respir. Crit. Care Med. 154, 981–987.
Burnett, R., Chen, H., Szyszkowicz, M., Fann, N., Hubbell, B., Pope, C.A., et al., 2018. Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. Proc. Natl. Acad. Sci. 115, 9592–9597.
Cakmak, S., Hebbern, C., Vanos, J., Crouse, D.L., Burnett, R., 2016. Ozone exposure and cardiovascular-related mortality in the Canadian census health and environment cohort (CANCHEC) by spatial synoptic classification zone. Environ. Pollut. 214, 589–599.
Chou, C.-K., Tsai, C.-Y., Chang, C.-C., Lin, P.-H., Liu, S., Zhu, T., 2011. Photochemical production of ozone in Beijing during the 2008 olympic games. Atmos. Chem. Phys. 11, 9825–9837.
Dai, H., Mischke, P., Xie, X., Xie, Y., Masui, T., 2016. Closing the gap? Top-down versus bottom-up projections of China's regional energy use and CO_2 emissions. Appl. Energy 162, 1355–1373.
Dong, H., Dai, H., Liang, D., Fujita, T., Geng, Y., Klimont, Z., et al., 2015. Pursuing air pollutant co-benefits of CO_2 mitigation in China: a provincial leveled analysis. Appl. Energy 144, 165–174.
Fann, N., Risley, D., 2013. The public health context for PM2.5 and ozone air quality trends. Air Qual. Atmos. Health 6, 1–11.
Fann, N., Lamson, A.D., Anenberg, S.C., Wesson, K., Risley, D., Hubbell, B.J., 2012. Estimating the national public health burden associated with exposure to ambient PM2.5 and ozone. Risk Anal. 32, 81–95.
Fujimori, S., Masui, T., Matsuoka, Y., 2015. Gains from emission trading under multiple stabilization targets and technological constraints. Energy Econ. 48, 306–315.
Gent, J.F., Triche, E.W., Holford, T.R., Belanger, K., Bracken, M.B., Beckett, W.S., et al., 2003. Association of low-level ozone and fine particles with respiratory symptoms in children with asthma. Jama 290, 1859–1867.
Giannadaki, D., Giannakis, E., Pozzer, A., Lelieveld, J., 2018. Estimating health and economic benefits of reductions in air pollution from agriculture. Sci. Total Environ. 622, 1304–1316.
Hubbell, B.J., Hallberg, A., McCubbin, D.R., Post, E., 2005. Health-related benefits of attaining the 8-hr ozone standard. Environ. Health Perspect. 73–82.
Jerrett, M., Burnett, R.T., Pope III, C.A., Ito, K., Thurston, G., Krewski, D., et al., 2009. Long-term ozone exposure and mortality. N. Engl. J. Med. 360, 1085–1095.
Jin, Y., 2017. Valuation of Health Risks With Discrete Choice Experiment and Cost-benefit Analysis of Air Pollution Control Strategies. Peking University (PhD thesis).
Korrick, S.A., Neas, L.M., Dockery, D.W., Gold, D.R., Allen, G.A., Hill, L.B., et al., 1998. Effects of ozone and other pollutants on the pulmonary function of adult hikers. Environ. Health Perspect. 106, 93.
Lelieveld, J., Evans, J., Fnais, M., Giannadaki, D., Pozzer, A., 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525, 367–371.
Maji, K.J., Ye, W.-F., Arora, M., Nagendra, S.S., 2018. PM2.5-related health and economic loss assessment for 338 Chinese cities. Environ. Int. 121, 392–403.
Maji, K.J., Ye, W.-F., Arora, M., Nagendra, S.S., 2019. Ozone pollution in Chinese cities: assessment of seasonal variation, health effects and economic burden. Environ. Pollut. 247, 792–801.
Malley, C.S., Henze, D.K., Kuylenstierna, J.C., Vallack, H.W., Davila, Y., Anenberg, S.C., et al., 2017a. Updated global estimates of respiratory mortality in adults ≥30 years of age attributable to long-term ozone exposure. Environ. Health Perspect. 125, 087021.
Malley, Christopher S., H, D.K., Kuylenstierna, Johan C.I., Vallack, Harry W., Davila, Yanko, Anenberg, Susan C., Turner, Michelle C., Ashmore, Mike R., 2017b. Updated global estimates of respiratory mortality in adults ≥30 years of age attributable to long-term ozone exposure. Environ. Health Perspect. 125, 9.
Matus, K., Yang, T., Paltsev, S., Reilly, J., Nam, K.-M., 2008. Toward integrated assessment of environmental change: air pollution health effects in the USA. Clim. Chang. 88, 59–92.
Matus, K., Nam, K.-M., Selin, N.E., Lamsal, L.N., Reilly, J.M., Paltsev, S., 2012. Health damages from air pollution in China. Glob. Environ. Chang. 22, 55–66.
McDonnell, W.F., Abbey, D.E., Nishino, N., Lebowitz, M.D., 1999. Long-term ambient ozone concentration and the incidence of asthma in nonsmoking adults: the AHSMOG study. Environ. Res. 80, 110–121.
OECD, 2016. The Economic Consequences of Outdoor Air Pollution. OECD Publishing, Paris.

O'Neill, B.C., Kriegler, E., Riahi, K., Ebi, K.L., Hallegatte, S., Carter, T.R., et al., 2013. A new scenario framework for climate change research: the concept of shared socio-economic pathways. Clim. Chang. 1–14.
Orru, H., Andersson, C., Ebi, K.L., Langner, J., Åström, C., Forsberg, B., 2013. Impact of climate change on ozone-related mortality and morbidity in Europe. Eur. Respir. J. 41, 285–294.
Qin, Y., Xie, S., 2011. A multi-factor designation method for mapping particulate-pollution control zones in China. Sci. Total Environ. 409, 3603–3612.
Qin, Y., Wagner, F., Scovronick, N., Peng, W., Yang, J., Zhu, T., et al., 2017. Air quality, health, and climate implications of China's synthetic natural gas development. Proc. Natl. Acad. Sci. 114, 4887–4892.
Qin, Y., Höglund-Isaksson, L., Byers, E., Feng, K., Wagner, F., Peng, W., et al., 2018. Air quality-carbon-water synergies and trade-offs in China's natural gas industry. Nat. Sustain. 1, 505.
Qin, Y., Fang, Y., Li, X., Naik, V., Horowitz, L.W., Liu, J., et al., 2019. Source attribution of black carbon affecting regional air quality, premature mortality and glacial deposition in 2000. Atmos. Environ. 206, 144–155.
Rosenthal, F.S., Kuisma, M., Lanki, T., Hussein, T., Boyd, J., Halonen, J.I., et al., 2013. Association of ozone and particulate air pollution with out-of-hospital cardiac arrest in Helsinki, Finland: evidence for two different etiologies. J. Expo. Sci. Environ. Epidemiol. 23, 281–288.
Selin, N.E., Wu, S., Nam, K.-M., Reilly, J.M., Paltsev, S., Prinn, R.G., et al., 2009. Global health and economic impacts of future ozone pollution. Environ. Res. Lett. 4, 044014.
Silva, R.A., West, J.J., Zhang, Y., Anenberg, S.C., Lamarque, J.-F., Shindell, D.T., et al., 2013. Global premature mortality due to anthropogenic outdoor air pollution and the contribution of past climate change. Environ. Res. Lett. 8, 034005.
Silva, R.A., West, J.J., Lamarque, J.-F., Shindell, D.T., Collins, W.J., Dalsoren, S., et al., 2016. The effect of future ambient air pollution on human premature mortality to 2100 using output from the ACCMIP model ensemble. Atmos. Chem. Phys. 16, 9847–9862.
Silva, R.A., West, J.J., Lamarque, J.-F., Shindell, D.T., Collins, W.J., Faluvegi, G., et al., 2017. Future Global Mortality From Changes in Air Pollution Attributable to Climate Change.
Turner, M.C., Jerrett, M., Pope III, C.A., Krewski, D., Gapstur, S.M., Diver, W.R., et al., 2016. Long-term ozone exposure and mortality in a large prospective study. Am. J. Respir. Crit. Care Med. 193, 1134–1142.
Viscusi, W.K., Aldy, J.E., 2003. The value of a statistical life: a critical review of market estimates throughout the world. J. Risk Uncertain. 27, 5–76.
Wang, H.G., Montoliu-Munoz, M., Gueye, N.F.D., 2009. Preparing to Manage Natural Hazards and Climate Change Risks in Dakar, Senegal: A Spatial and Institutional Approach.
Wang, H., Zhang, Y., Zhao, H., Lu, X., Zhang, Y., Zhu, W., et al., 2017. Trade-driven relocation of air pollution and health impacts in China. Nat. Commun. 8.
West, J.J., Smith, S.J., Silva, R.A., Naik, V., Zhang, Y., Adelman, Z., et al., 2013. Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health. Nat. Clim. Chang. 3, 885–889.
WHO, 2013. Review of Evidence on Health Aspects of Air Pollution-REVIHAAP Project. World Health Organization, Copenhagen, Denmark.
Xie, X., 2011. The Value of Health: Applications of Choice Experiment Approach and Urban Air Pollution Control Strategy. Ph.D. Thesis. Peking university.
Xie, Y., Dai, H., Dong, H., Hanaoka, T., Masui, T., 2016. Economic impacts from PM2.5 pollution-related health effects in China: a provincial-level analysis. Environ. Sci. Technol. 50, 4836–4843.
Xie, R., Sabel, C.E., Lu, X., Zhu, W., Kan, H., Nielsen, C.P., et al., 2016. Long-term trend and spatial pattern of PM2.5 induced premature mortality in China. Environ. Int. 97, 180–186.
Xue, L., Wang, T., Gao, J., Ding, A., Zhou, X., Blake, D., et al., 2014. Ground-level ozone in four Chinese cities: precursors, regional transport and heterogeneous processes. Atmos. Chem. Phys. 14, 13175–13188.
Yan, Y., Lin, J., Chen, J., Hu, L., 2016. Improved simulation of tropospheric ozone by a global-multi-regional two-way coupling model system. Atmos. Chem. Phys. 16, 2381–2400.
Zhang, R., Lei, W., Tie, X., Hess, P., 2004. Industrial emissions cause extreme urban ozone diurnal variability. Proc. Natl. Acad. Sci. U. S. A. 101, 6346–6350.
Zhang, Y., Huang, W., London, S.J., Song, G., Chen, G., Jiang, L., et al., 2006. Ozone and daily mortality in Shanghai, China. Environ. Health Perspect. 1227–1232.
Zhang, X., Chen, X., Zhang, X., 2018. The impact of exposure to air pollution on cognitive performance. Proc. Natl. Acad. Sci. 115, 9193–9197.
Zhang, W., Liu, Y., Feng, K., Hubacek, K., Wang, J., Liu, M., et al., 2018. Revealing environmental inequality hidden in China's inter-regional trade. Environ. Sci. Technol. 52, 7171–7181.