China's international trade and air pollution in the United States

Jintai Lin[a,1,2], Da Pan[a,1], Steven J. Davis[b], Qiang Zhang[c,2], Kebin He[d,e,2], Can Wang[c,d], David G. Streets[f], Donald J. Wuebbles[g], and Dabo Guan[c,h]

[a]Laboratory for Climate and Ocean-Atmosphere Studies, Department of Atmospheric and Oceanic Sciences, School of Physics, Peking University, Beijing 100871, China; [b]Department of Earth System Science, University of California, Irvine, CA 92697; [c]Ministry of Education Key Laboratory for Earth System Modeling, Center for Earth System Science, Tsinghua University, Beijing 100084, China; [d]State Key Joint Laboratory of Environmental Simulation and Pollution Control, School of Environment, Tsinghua University, Beijing 100084, China; [e]Collaborative Innovation Center for Regional Environmental Quality, Beijing 100084, China; [f]Argonne National Laboratory, Lemont, IL 60439; [g]Department of Atmospheric Sciences, School of Earth, Society, and Environment, University of Illinois at Urbana–Champaign, Urbana, IL 61801; and [h]water@leeds, School of Earth and Environment, University of Leeds, Leeds, LS2 9JT, United Kingdom

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China is the world's largest emitter of anthropogenic air pollutants, and measurable amounts of Chinese pollution are transported via the atmosphere to other countries, including the United States. However, a large fraction of Chinese emissions is due to manufacture of goods for foreign consumption. Here, we analyze the impacts of trade-related Chinese air pollutant emissions on the global atmospheric environment, linking an economic-emission analysis and atmospheric chemical transport modeling. We find that in 2006, 36% of anthropogenic sulfur dioxide, 27% of nitrogen oxides, 22% of carbon monoxide, and 17% of black carbon emitted in China were associated with production of goods for export. For each of these pollutants, about 21% of export-related Chinese emissions were attributed to China-to-US export. Atmospheric modeling shows that transport of the export-related Chinese pollution contributed 3–10% of annual mean surface sulfate concentrations and 0.5–1.5% of ozone over the western United States in 2006. This Chinese pollution also resulted in one extra day or more of noncompliance with the US ozone standard in 2006 over the Los Angeles area and many regions in the eastern United States. On a daily basis, the export-related Chinese pollution contributed, at a maximum, 12–24% of sulfate concentrations over the western United States. As the United States outsourced manufacturing to China, sulfate pollution in 2006 increased in the western United States but decreased in the eastern United States, reflecting the competing effect between enhanced transport of Chinese pollution and reduced US emissions. Our findings are relevant to international efforts to reduce transboundary air pollution.

input–output analysis | emission control | international collaboration

A key driver of the rapid economic growth in China over the past decade is the great expansion in the production of goods for export (1). Although growth has slowed since the global financial crisis, between 2000 and 2007 the volume of Chinese exports grew by 390% (2). As the Chinese economy has grown, the economic structure has also changed, transitioning from a net importer to a large net exporter of energy-intensive industrial products (2). The energy needed to support this economic growth and transformation has come from combustion of fossil fuels, primarily coal, which has contributed to a global increase in emissions of carbon dioxide (CO_2) (3, 4). At the same time, increased combustion of fossil fuels, relatively low combustion efficiency, and weak emission control measures have also led to drastic increases in air pollutants such as sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), black carbon (BC), and primary organic carbon (OC) (5–8). Indeed, fossil-fuel–intensive manufacturing, large manufacturing volume, and relatively weak emission controls have meant that China emits far more pollutants per unit of gross domestic product (GDP) than countries with more advanced industrial and emission control technologies (*SI Appendix*, Table S1). Per unit of GDP in 2006, China emitted 6–33 times as much air pollutants as the United States (Fig. 1 *E–H*). For these reasons, air quality has recently become a major focus of environmental policy in China (8).

In this study, the terms "export," "import," and "trade" all refer to transaction of goods between countries. The pollutants emitted in China due to its production of goods for foreign consumption are regarded as emissions embodied in export (EEE) of China (9, 10). The EEE is unique in that the associated goods are consumed outside of China, raising a question about the extent to which China and its export partners should be accountable for the emissions (10–12). The attribution depends on whether the emission accounting is based on production or on consumption. Production-based accounting considers all emissions physically produced in China to be Chinese emissions, including the EEE. Such accounting is used as default in current emission inventories such as the Emission Database for Global Atmospheric Research (13). By comparison, consumption-based accounting views all emissions associated with production of goods consumed by China to be China's responsibility, no matter whether the production occurs in China or in other countries (9, 10). Thus, the consumption-based Chinese emissions exclude the EEE but include the emissions embodied in import of China

Significance

International trade affects global air pollution and transport by redistributing emissions related to production of goods and services and by potentially altering the total amount of global emissions. Here we analyze the trade influences by combining an economic-emission analysis on China's bilateral trade and atmospheric chemical transport modeling. Our focused analysis on US air quality shows that Chinese air pollution related to production for exports contributes, at a maximum on a daily basis, 12–24% of sulfate pollution over the western United States. The US outsourcing of manufacturing to China might have reduced air quality in the western United States with an improvement in the east, due to the combined effects of changes in emissions and atmospheric transport.

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[1]J.L. and D.P. contributed equally to this work.

[2]To whom correspondence may be addressed. E-mail: linjt@pku.edu.cn, qiangzhang@tsinghua.edu.cn, or hekb@tsinghua.edu.cn.

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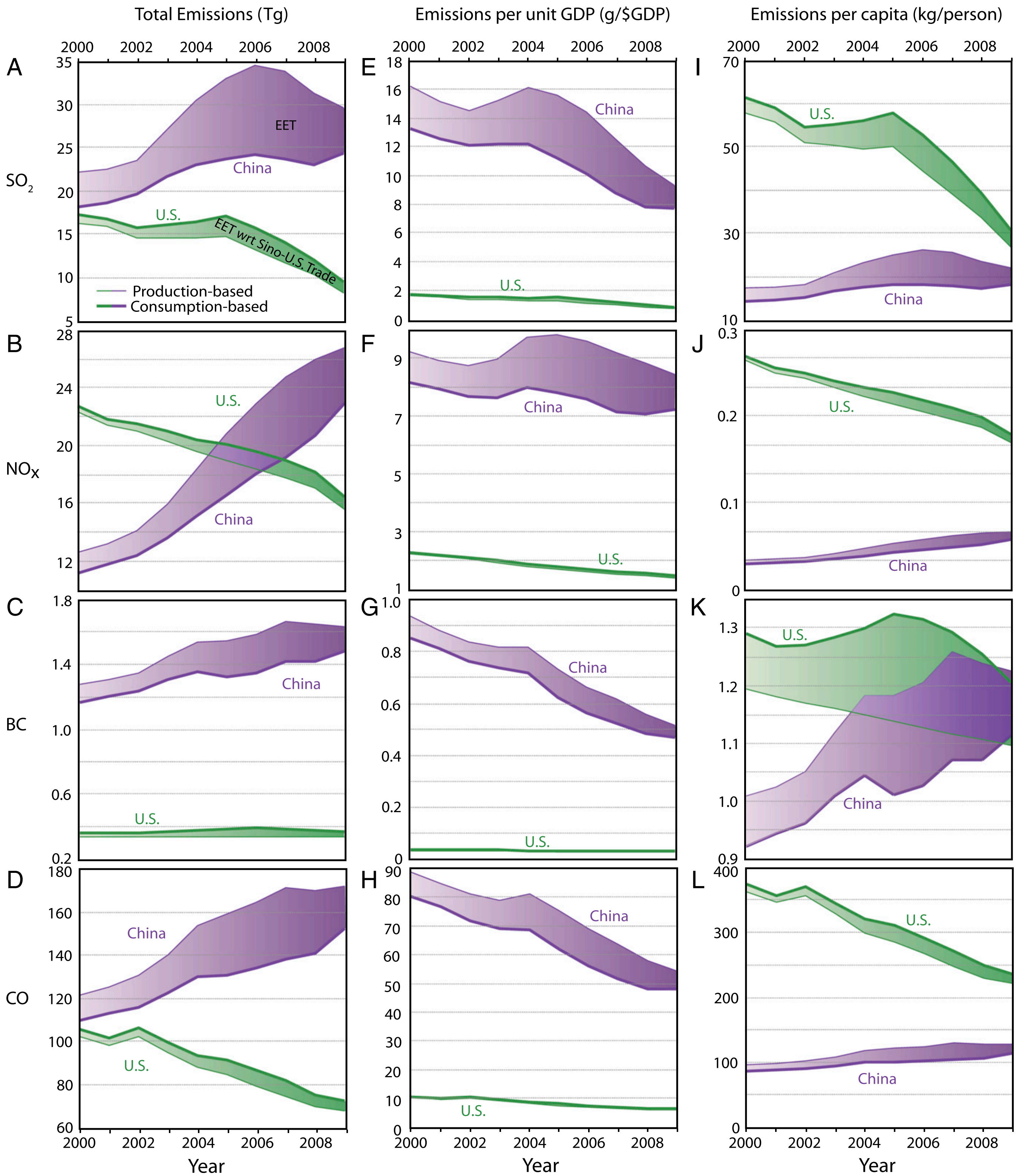

Fig. 1. Air pollutants embodied in Chinese trade between 2000 and 2009. (*A*–*D*) Production-based emissions (thin lines), consumption-based emissions (thick lines), and their differences (i.e., Chinese EET associated with its trade with the rest of the world in purple shading, and EET associated with Sino-US trade alone in green shading). All Chinese emissions are calculated here, the US production-based emissions are taken from the National Emissions Inventory, and the US consumption-based emissions are derived based on production-based emissions and Sino-US trade-related emissions. Although China's production-based emissions are growing rapidly, its EET are equivalent to substantial fractions of the production-based emissions. Similarly, the EET due to Sino-US trade are equivalent to large proportions of the production-based US emissions since 2006. (*E*–*H*) Emissions per GDP. Although China's production-based emissions per unit GDP have been decreasing, its consumption-based emissions per unit GDP have decreased less significantly or have increased since 2008. (*I*–*L*) Emissions per capita. Per capita emissions are very different between the United States and China, and this disparity is increased when the consumption-based emissions are considered. For data sources, see *SI Appendix*, Table S1, footnote.

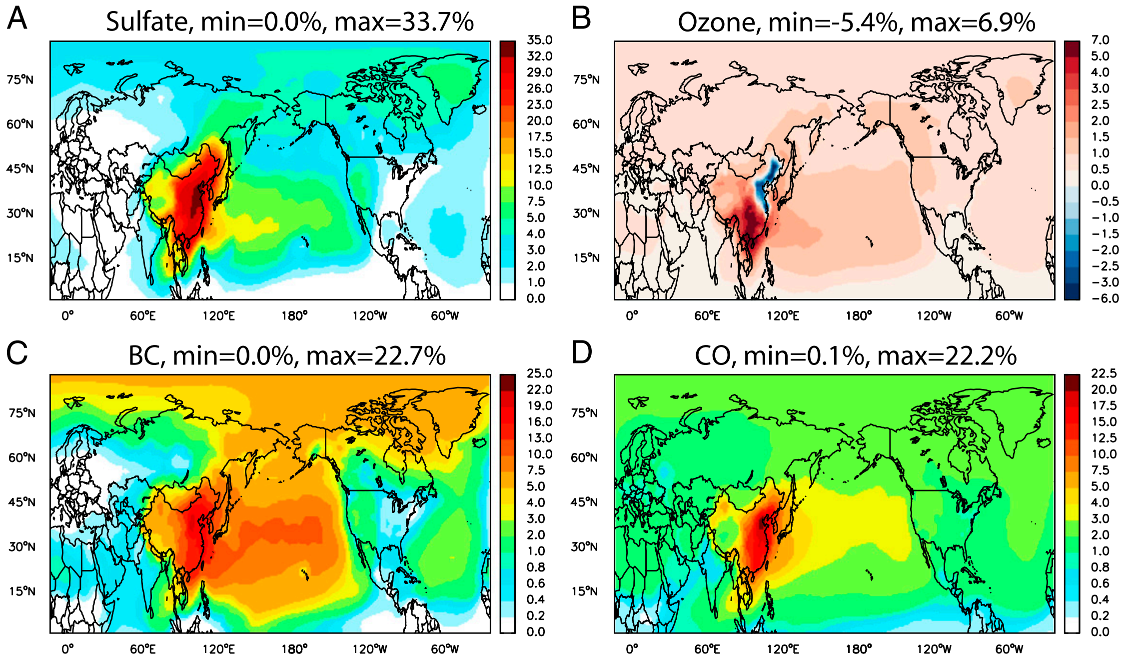

Fig. 2. Simulated percentage contribution of surface air pollution in 2006 from Chinese EEE for (*A*) sulfate, (*B*) ozone, (*C*) BC, and (*D*) CO. Results are shown for annual mean concentrations in the lowest model layer (0–130 m), presented as (simulation 1 – simulation 2)/simulation 1 in the *SI Appendix*, section 6. The color scale is nonlinear to better present the wide range of impacts over different regions. The Chinese EEE affect pollutant concentrations most significantly over China, but they also affect the rest of East Asia, the Arctic, western North America, and other regions downwind of China. The negative impacts on ozone concentrations over parts of the northern Chinese provinces are primarily because the EEE-related NO_x emissions increase the ozone sink in the nighttime overcompensating for the effect of enhanced ozone production in the daytime.

(EEI, i.e., emissions in other countries due to production of goods for Chinese consumption). The numerical difference between production- and consumption-based emissions of China is the EEE less the EEI, the result of which is regarded as the emissions embodied in net trade (EET) of China (10). Similar emission analyses are applicable to other countries.

Previous studies have quantified the substantial CO_2 emissions embodied in Chinese trade (10, 11). Thus, far, however, relatively little attention has been paid to trade-related emissions of short-lived air pollutants and especially the resulting impacts on the global atmospheric environment, except for an analysis done for local air quality of the Pearl River Delta (14). This is true despite the direct harm these pollutants do to human health (15–18), agriculture (19), ecosystems (20), and global climate (21, 22). And as scientific evidence of transport of Chinese air pollution across the Pacific Ocean has grown since the late 1990s (23–29), the United States and Canada have a special interest in reducing Chinese air pollution. In the case of CO_2, consumption-based accounting of emissions has been motivated by the argument—often made by developing countries—that consumers who benefit from a process should bear some responsibility for associated environmental damage (30). A similar accounting for emissions of air pollutants and consequent impacts on the global atmospheric environment may therefore be necessary to facilitate discussion of international collaborations on transboundary air pollution control (31).

We quantify the emissions of SO_2, NO_x, CO, BC, and OC embodied in Chinese exports and imports between 2000 and 2009 using an economic input–output model constructed from economic and emission data. The model resolves trade between China and four countries/regions [the United States, the European Union (EU), Japan, and an aggregated region of all other countries] and 42 industry sectors, and allocates pollutant emissions to countries and industry sectors according to where goods are consumed. As part of our analysis, we also quantify the uncertainties in emission derivation using a Monte Carlo approach. We then simulate the effects of export-related Chinese emissions on air pollution in China and downwind regions, using the GEOS-Chem global chemical transport model. See *SI Appendix* for details of our analytic approach, data sources, and model simulations.

Results

Fig. 1 *A–D* shows the trends over 2000–2009 in the EET of China related to its trade with the rest of the world (purple shading) and in the EET with respect to Sino-US trade alone (green shading), together with the production- and consumption-based emission accounting for China and the United States. For China, although the production-based emissions of SO_2 and BC have declined since 2007 due to the global financial crisis and sulfur emission control, the consumption-based emissions of all pollutants have continued to rise, reflecting a net decrease in the EET. Nonetheless, the EET were equivalent to a large fraction of production-based Chinese emissions, and this fraction expanded between 2000 and 2006. For example, the EET of SO_2 grew from 4.0 teragrams (Tg) (equivalent to 18% of production-based Chinese emissions) in 2000 to 10.3 Tg (30%) in 2006 (Fig. 1*A*). The fraction of the EET grew similarly for NO_x, CO, and BC (Fig. 1 *B–D*). Meanwhile, although the EET for Sino-US trade were equivalent only to 2–8% of production-based US emissions in 2000, the proportion grew by a factor of 2–3 to reach 6–19% in 2006 (Fig. 1*A–D*). This trend reflects the decline of production-based emissions of the United States and its continuous outsourcing (32).

Although the EET represent the difference between the EEE and the EEI, the EET of China were numerically close to its EEE over 2000–2009. This is because the EEE of China are larger than the EEI by a factor of 4–6 during these years, reflecting China's trade imbalance with the rest of the world, the types of goods being traded, and the differences in emission intensity between China and its trading partners (*SI Appendix*, section 5.2). In 2006, the Chinese EEE contributed 36% of its production-based emissions for SO_2, 27% for NO_x, 22% for CO, and 17% for BC. And for all these pollutants, about 21% of the Chinese EEE in 2006 were attributed to China-to-US export of goods.

Fig. 1 *E*–*H* shows that Chinese emissions per unit of GDP have mostly decreased between 2000 and 2009. However, the production-based emissions per unit GDP have recently decreased at a faster rate than have the consumption-based emissions per unit GDP. In the case of NO_x, the consumption-based Chinese emissions per unit GDP have actually increased since 2008 (Fig. 1*F*). Meanwhile, emissions per unit GDP have also declined in the United States, regardless of whether or not the emissions embodied in Sino-US trade are included (Fig. 1 *E*–*H*). The emissions per unit GDP for China are much greater than those for the United States, based on both production- and consumption-based accounting. In 2009, the production-based emissions per unit GDP for China were about 6–17 times greater than the United States. The difference in consumption-based emissions per unit GDP was somewhat less: 5–14 times greater in China than the United States.

Finally, Fig. 1 *I*–*L* illustrates the large gap in emissions per capita between the United States and China. Over 2000–2009, the EET per capita for China related to its trade with the rest of the world (purple shading) were close to the EET per capita for the United States related to Sino-US trade alone (green shading). For China, although the production-based emissions per capita have fallen or flattened since 2007, the consumption-based emissions per capita have increased (Fig. 1 *I*–*L*). This again suggests that the global financial crisis affected Chinese exports but did not stem domestic growth. The trends contrast to the reductions in both production- and consumption-based emissions per capita for the United States.

Using the GEOS-Chem chemical transport model, we simulated the impacts of the EEE-related Chinese pollution on the global atmospheric environment in 2006 (*SI Appendix*, section 6 for descriptions of various model simulations). Fig. 2 shows the modeled percentage of annual mean surface pollutant concentrations in the Northern Hemisphere in 2006 attributable to the atmospheric transport and transformation of the EEE-related Chinese air pollution. The EEE-related Chinese pollution accounted for 23–34% of sulfate concentrations, 10–23% of BC, and 12–23% of CO over East China (east of 100°E). This pollution resulted in ozone reductions over the North China Plain and Northeast China with ozone enhancements over the southern provinces. The mixed impacts reflect the nonlinear chemical processes that govern the ozone level: The additional NO_x due to Chinese EEE enhanced the nighttime ozone loss, compensating for the effect of enhanced daytime ozone production (33).

Fig. 2 shows that, through the atmospheric transport and transformation, parts of the EEE-related Chinese pollution in 2006 affected the surface air pollutant levels over the rest of East Asia, the North Pacific, western North America, Arctic, and other regions downwind of China. In particular, the EEE-related Chinese pollution contributed about 3–10% of the annual mean surface sulfate concentrations, 1–3% of BC, 2–3% of CO, and 0.5–1.5% of ozone over the western contiguous United States (west of 100°W). On a monthly basis, the trans-Pacific transport of Chinese air pollution was enhanced in spring (*SI Appendix*, Fig. S7) due to active cyclonic activities and strong westerly winds (24, 34).

The trans-Pacific transport is largely episodic (28, 35), such that the influence of Chinese pollution on US air quality varies significantly from one day to another. Fig. 3 shows the maximum contribution of EEE-related Chinese air pollution to daily mean surface air pollutant concentrations over the United States in 2006. On a day-to-day basis, the transport of EEE-related Chinese pollution contributed, at a maximum, 12–24% of sulfate concentrations, 2–5% of ozone, 4–6% of CO, and up to 11% of BC over the western United States, and it also contributed up to 8% of daily mean ozone over parts of the Great Lakes region. Furthermore, the trans-Pacific transport increased the number of days in 2006 when the daily maximum 8-h average ozone concentration exceeded the current US standard (75 ppb). For the 217 model gridcells constituting the contiguous United States, there are 38 gridcells that had one extra day or more of ozone exceedance in 2006 because of the transport of the EEE-related Chinese air pollution, including the gridcells covering the Los Angeles area and many regions in the eastern United States (*SI Appendix*, Fig. S8).

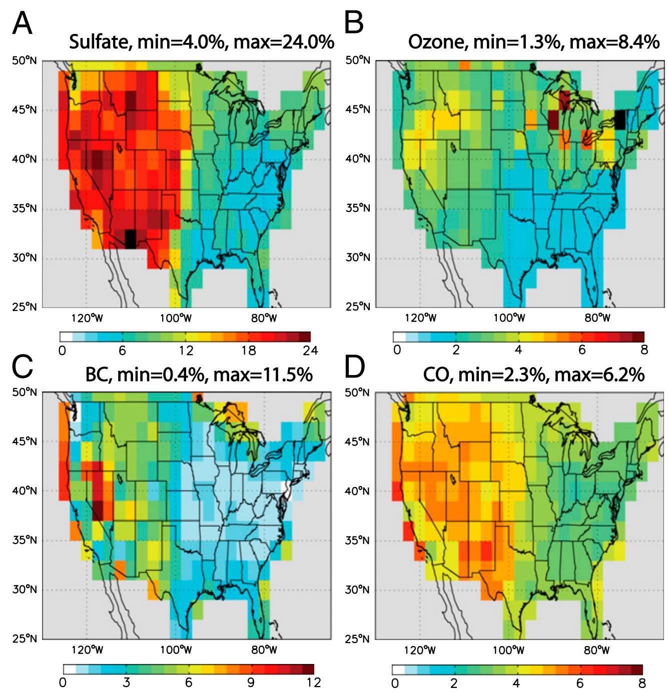

Fig. 3. Simulated maximum percentage contribution of Chinese EEE to daily mean US surface air pollution in 2006 for (*A*) sulfate, (*B*) ozone, (*C*) BC, and (*D*) CO. Results are presented as the maximum value across the 365 d in 2006 of (simulation 1 − simulation 2)/simulation 1 in *SI Appendix*, section 6. Results outside of the contiguous United States are colored in gray. The maximum contribution of EEE-related Chinese air pollution to US pollutant levels on a day-to-day basis is much greater than the annual mean influence.

In 2006, China-to-US export of goods resulted in about 7.4% of the production-based Chinese emissions for SO_2, 5.7% for NO_x, 3.6% for BC, and 4.6% for CO. Meanwhile, outsourcing manufacture to China also led to a reduction in production-based US emissions. Had the United States produced all of the goods that are actually imported from China under a hypothetical scenario, we estimate that the production-based US emissions in 2006 would be higher by 1.7% for SO_2, 1.3% for NO_x, 0.8% for BC, and 1.1% for CO, after accounting for the difference in emission intensity between the two countries (*SI Appendix*, Table S1). No China-to-US exports would also mean less production-based Chinese emissions. We used GEOS-Chem to simulate the changes in 2006 surface air quality in China and the United States due to emission changes in both countries with versus without China-to-US exports, assuming the spatial variability of emissions to be unaffected. The modeling results in Fig. 4 show that about 3–7% of annual mean surface sulfate concentrations, 2–5% of BC, and 2–5% of CO over East China in 2006 were caused by the Chinese EEE due to its production of goods for US consumption. Over the eastern United States (east of 100°W), annual mean surface concentrations in 2006 were

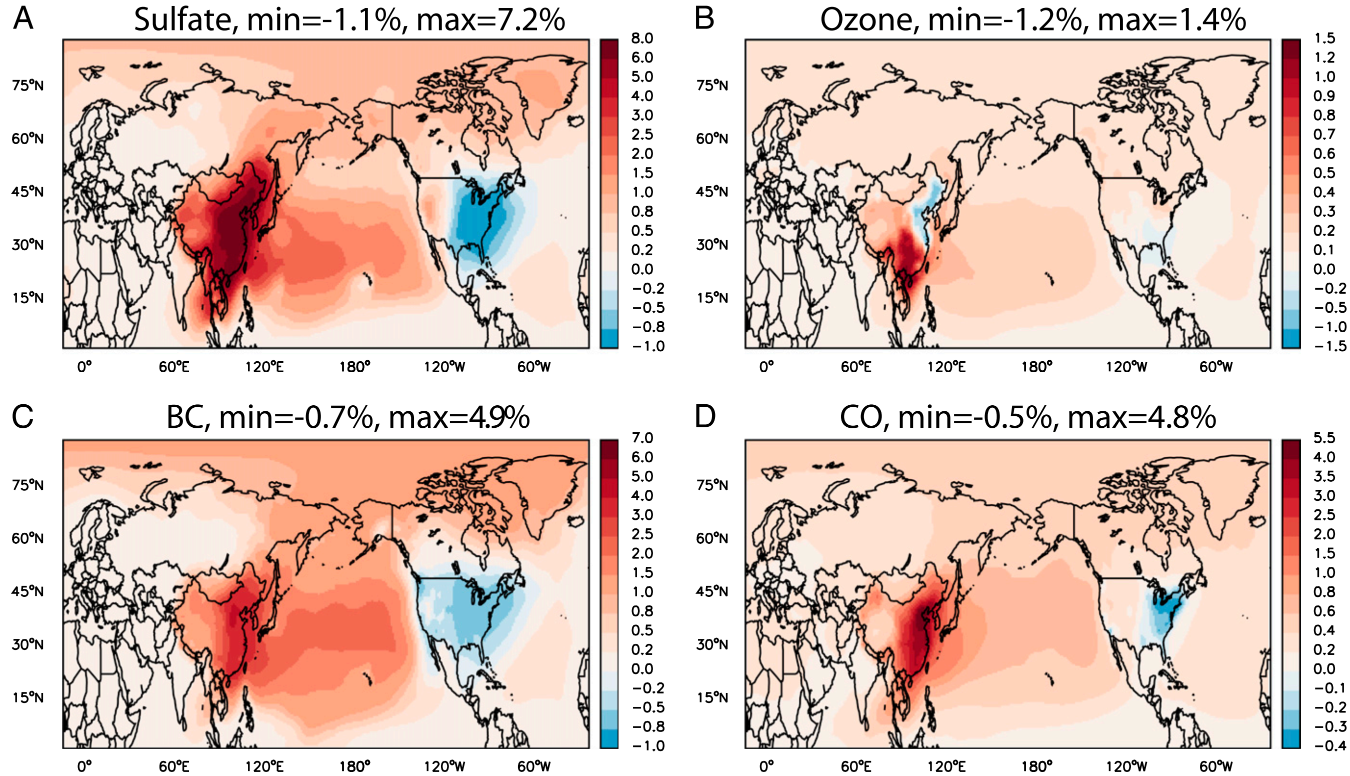

Fig. 4. Simulated percentage change in 2006 surface air pollution due to Chinese export of goods to the United States versus producing the same goods in the United States for (*A*) sulfate, (*B*) ozone, (*C*) BC, and (*D*) CO. Results are shown for annual mean concentrations in the lowest model layer (0–130 m), presented as (simulation 1 – simulation 6)/simulation 1 in the *SI Appendix*, section 6. The color scale is nonlinear to better present the wide range of impacts over different regions. The China-to-US export of goods results in enhanced (production-based) emissions of China with a reduction in US emissions. Air quality in China worsens as a result of these additional emissions. Over the western United States, concentrations of sulfate, ozone, and CO also increase because the elevated transport of Chinese pollution overcompensates for the effect of reduced US emissions. Meanwhile, concentrations of sulfate, BC, and CO decrease over the eastern United States, a beneficial effect particularly given its high population density.

reduced by 0.5–1.1% for sulfate, 0.5–0.8% for BC, and 0–0.5% for CO as a result of US emission reductions due to outsourcing manufacture to China. Over the western United States, however, sulfate concentrations were enhanced by 0–2% and ozone and CO levels were also increased slightly. These increases occurred because the transport of EEE-related Chinese pollution overcompensated for the effect of reduced US emissions. Given the much higher population density in the eastern United States (http://sedac.ciesin.columbia.edu/data/collection/gpw-v3; for 2005), outsourcing manufacture to China resulted in an overall beneficial effect for the US public health. In particular, population-weighted average sulfate, BC, and CO concentrations decreased by 0.3–0.9% over the United States (125°W–70°W, 33°N–49°N) with a negligible increase of 0.1% in ozone. This benefit, however, was at the expense of air quality deterioration over the western United States and the populous Chinese regions.

Our emission and atmospheric model results are subject to uncertainties from a variety of sources. The emission calculations are affected by errors in emission factors, economic statistics, and input–output tables. A detailed error analysis for total emissions, EEE, and EEI is presented in the *SI Appendix*, sections 3 and 5.3, based on Monte Carlo simulations. For EEE, the overall uncertainties (95% confidence intervals around the central estimates) are about −17% to 17% for SO_2, −27% to 27% for NO_x, −45% to 45% for CO, and −35% to 51% for BC. The uncertainties for EEI are larger for these pollutants reflecting our simplified treatment of EEI (*SI Appendix*, sections 1.1.1 and 5.3). The uncertainties for EET are close to those for EEE. The atmospheric model simulations are subject to errors in emission inputs as well as errors in the model representations of tropospheric chemical and meteorological processes. The chemistry- and meteorology-related model uncertainties are difficult to quantify and are likely on the order of 30% (36). For the model results in Figs. 2–4 presented as percentage contribution, the uncertainties may be reduced substantially because the presented values are the normalized differences between pollutant concentrations from various model simulations (*SI Appendix*, section 6) whose uncertainties may largely offset each other. Our modeling results are for 2006, and the results may be different for other years.

Discussion

Rising emissions produced in China are a key reason global emissions of air pollutants have remained at a high level during 2000–2009 even as emissions produced in the United States, Europe, and Japan have decreased. However, our results indicate that about 36% of SO_2 and 27% of NO_x emitted in China in 2006 (19–24% in 2009) were related to goods exported for consumption outside of China. If all of the emissions were reallocated according to where goods are consumed (i.e., based on consumption-based accounting), emissions of many of China's trade partners would be much higher. For example, the US emissions for SO_2, NO_x, CO, and BC would be 6–19% higher in 2006 if the emissions embodied in its trade with China were included (Fig. 1 *A*–*D*; thick green versus thin green lines). And as we have also shown, outsourcing production to China does not always relieve consumers in the United States—or for that matter many countries in the Northern Hemisphere—from the environmental impacts of air pollution. Sulfate air quality in the western United States is poorer because of transport of Chinese pollution associated with production of goods for US consumption, although air quality in the eastern United States is improved.

The thin purple lines in Fig. 1 *E–H* show the significant progress China has made since 2000 in reducing the (production-based) emissions per unit GDP through technological improvements and changes in economic structure (7, 37). In particular, SO_2 emissions per unit GDP are decreasing rapidly since 2004 (38) (Fig. 1*E*). However, the emissions per unit GDP for all pollutants remain much higher than those of the United States (Fig. 1 *E–H*), and further improvements in technology and economic structure could reduce emissions of pollutants much more. Differences in the ratio of pollutant to CO_2 emissions between the United States and China (*SI Appendix*, section 7 and Table S11) indicate that production-based Chinese emissions could be reduced by 58–62% for SO_2, 47–54% for CO, and up to 22% for NO_x over 2000–2009 if China were to enhance energy efficiency and deploy emission control technologies as effective as those used in the United States. Even if such improvements were made to only those facilities involved in producing goods for export, the reduction in emissions would significantly improve the air quality in China and in downwind regions. For instance, the annual mean surface sulfate concentrations in 2006 would have been about 10–19% lower in China and 1–5% lower in the western United States based on the simulation of GEOS-Chem.

Consideration of international cooperation to reduce transboundary transport of air pollution (31) must confront the question of who is responsible for emissions in one country during production of goods to support consumption in another. Polluting industries in China and other emerging economies supply a large proportion of global consumption through international trade. Sustaining the current trading system while minimizing transboundary air pollution—and other environmental impacts—will likely require international agreements informed by consumption-based accounting of emissions of air pollutants as well as atmospheric transport modeling of air pollution.

Materials and Methods

Calculation of EEE and EEI is based on an input–output analysis of the economic processes required to produce a particular good or service, multiplied by sector-specific emission intensities. See *SI Appendix*, Fig. S1 for the flowchart. Emissions from ocean shipping vessels are not accounted for. Sectoral emission intensities are calculated as total production-based Chinese emissions (which are estimated with a technology-based, bottom-up approach) divided by total monetary outputs from the respective sectors. The estimated production-based total emissions are consistent with the literature (*SI Appendix*, Fig. S3). A Monte Carlo method is used to quantify uncertainty associated with errors in emission factors, economic statistics, and the input–output analysis itself. Emissions of CO_2 are calculated with a similar approach, and the resulting emissions embodied in trade are consistent with previous studies (*SI Appendix*, Fig. S4). The global GEOS-Chem chemical transport model (version 8–03-02; on the 2.5° long × 2° lat grid) is used to simulate the impacts of EEE-related Chinese air pollution on the global atmospheric environment. We do not distinguish the EEE of volatile organic compounds that would otherwise enhance the modeled ozone production efficiency of NO_x; a sensitivity simulation shows that the effect is mostly confined in the North China Plain (*SI Appendix*, section 6 and Fig. S6). Detailed descriptions of our analytic approach, data sources, and model simulations are presented in *SI Appendix*.

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