Manuscript prepared for Atmos. Chem. Phys. with version 2015/04/24 7.83 Copernicus papers of the LATEX class copernicus.cls. Date: 30 March 2017

Comparison of Emissions Inventories of Anthropogenic Air Pollutants in China

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Abstract. Anthropogenic air pollutant emissions have been increasing rapidly in China, leading to worsening air quality. Modelers use emissions inventories to represent the temporal and spatial distribution of these emissions needed to estimate their impacts on regional and global air quality. However, large uncertainties exist in emissions estimates. Thus, assessing differences in these in-

- 5 ventories is essential for better understanding of air pollution over China. We compare five different emissions inventories estimating emissions of carbon dioxide (CO₂), carbon monoxide (CO), nitrogen oxides (NO_x), sulfur dioxide (SO₂), and particulate matter with an aerodynamic diameter of 10 μ m or less (PM₁₀) from China. The emissions inventories analyzed in this paper include Regional Emissions inventory in ASia v2.1 (REAS); Multi-resolution Emission Inventory for China
- 10 (MEIC); Emission Database for Global Atmospheric Research v4.2 (EDGAR); the inventory by Yu Zhao (ZHAO); and the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS). We focus on the period between 2000 and 2008 during which Chinese economic activities have more than doubled. In addition to national totals, we also analyzed emissions from four source sectors (industry, transport, power, and residential) and within seven regions in China (East, North, Northeast,
- 15 Central, Southwest, Northwest, and South) and found that large disagreements exist among the five inventories at disaggregated levels. These discrepancies lead to differences of $67\mu g/m^3$, 15ppbv, and 470ppbv for monthly mean PM₁₀, O₃, and CO, respectively, in modelled regional concentrations in China. We also find that all the inventory emissions estimates create a VOC-limited environment and MEIC emissions lead to much lower O₃ mixing ratio in East and Central China compared to the
- 20 simulations using REAS and EDGAR estimates, due to its low VOC emissions. Our results illustrate

that a better understanding of Chinese emissions at more disaggregated levels is essential for finding effective mitigation measures for reducing national and regional air pollution in China.

1 Introduction

- Obtaining accurate emissions estimates for air pollutant species is important in Asia, where five of
 the worst air quality countries in the world are located (Hsu et al., 2014). Emissions of ozone precursors, including nitrogen oxides (NO_x ≡ NO + NO₂) and carbon monoxide (CO), affect tropospheric ozone (O₃) mixing ratio at local, regional, and inter-continental scales (Fiore et al., 2009; West et al., 2009). In addition to the emissions of primary particulate matter (PM), those of sulfur dioxide (SO₂) and NO_x also affect PM concentrations on local and regional scales. Both surface O₃ and PM are
 linked to adverse health impacts (Dockery et al., 1993; Levy et al., 2001; Pope III et al., 2002), and
- O₃ also affects agricultural crop yields (Heck et al., 1983; Krupa and Manning, 1988; Avnery et al., 2011).

One key country in need of accurate emissions estimates is China, the largest emitter and the biggest contributor to the uncertainty in the source and the magnitude of many of the air pollu-

- tant species. The difference in global CO, SO₂, and NO_x emissions estimates among inventories is 28%, 42%, and 17% in 2000, respectively (Granier et al., 2011). China's uncertainty is much larger for CO and NO_x and 90% of global CO₂ emissions uncertainty stems from China (Andres et al., 2014). Energy consumption has been steadily increasing in China but at the same time, the implementation of emissions control measures, including the flue-gas desulphurization (FGD) in
- 40 coal-fired power plants, has led to rapid changes in emission factors in recent decades (Xu, 2011; Zhang et al., 2012; Kurokawa et al., 2013). Several emissions inventories have been developed in the past, either specifically for China or for Asia (Streets and Waldhoff, 2000; Streets et al., 2003; Zhao et al., 2008; Klimont et al., 2009; Lu et al., 2010; European Commission, Joint Research Centre (JRC)/Netherlands Environmental Assessment Agency (PBL), 2011; Lei et al., 2011; Lu et al.,
- 45 2011; Smith et al., 2011; Zhao et al., 2012; Kurokawa et al., 2013; Klimont et al., 2013) but none have assessed or compared emissions from different source sectors at more disaggregated scales than the national level.

The purpose of this study is to analyze the differences among the existing emissions inventory estimates for China's anthropogenic gaseous and aerosol emissions and how they affect air quality

- 50 simulations. We analyze the emissions of carbon dioxide (CO₂), carbon monoxide (CO), sulfur dioxide (SO₂), nitrogen oxides (NO_x), non-methane volatile organic compounds (NMVOCs), and particulate matter with an aerodynamic diameter less than 10 μ m (PM₁₀). We first evaluate the differences among inventories at the national level between years 2000 and 2008 for CO₂, CO, SO₂, NO_x, and PM₁₀ and produce composite emissions estimates, using Monte Carlo samplings. Second,
- 55 we focus on four source sectors (industry, transport, power, and residential) in seven regions of

China (the East, North, Northeast, Central, Southwest, Northwest and South) for CO, SO₂, NO_x, and PM₁₀. Next, we analyze emissions estimates in the transport sector in more detail. By disaggregating emissions into these source sectors and regions, we aim to understand where the differences occur and how we can better constrain emissions. We also use a chemical transport model, the Weather

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Research and Forecasting model coupled with Chemistry (WRF-Chem), to assess how the different emissions estimates affect air quality modeling results.

The paper is organized as follows. Section 2 explains the emissions inventories that we have compared. Section 3 analyzes the differences in emissions inventories first at the national level and then in seven regions within China. Section 4 compares transport sector emissions in depth. Section

5 describes the impact of the emissions inventories on air quality simulations. Section 6 presents a summary of results and suggested future research.

2 Emissions Inventories

In this study, we compare five existing emissions inventories at the national, regional, and source sector levels between years 2000 and 2008 (Table 1). The Regional Emission inventory in ASia
version 2.1 (REAS) is a regional emissions inventory for most of the Asian countries including the

- East, Southeast, South, and Central Asia and the Asian part of Russia (Kurokawa et al., 2013). The Emission Database for Global Atmospheric Research version 4.2 (EDGAR) is a global emissions inventory and includes major air pollutants from combustion and non-combustion sources (European Commission, Joint Research Centre (JRC)/Netherlands Environmental Assessment Agency (PBL),
- 75 2011). Multi-resolution Emission Inventory for China (MEIC, http://meicmodel.org/) is an inventory developed at Tsinghua University, Beijing, China, and provides national emissions estimates for 2008 and 2010. A national emissions inventory for the 2000-2008 period was developed at Nanjing University (Zhao et al., 2008) and includes disaggregated information at the source sector and provincial levels for the year 2007. The Greenhouse Gas and Air Pollution Interactions and Synergies
- 80 (GAINS, http://gains.iiasa.ac.at/models/index.html) model is a framework for analyzing co-benefits of reduction strategies for air pollution and greenhouse gas sources globally, which provides estimates of emissions, including province-level emissions from China (Amann et al., 2011). These five emissions inventories were developed using a similar methodology, where emissions were calculated as the product of activity data, such as fuel consumption or industrial production, emission factors
- 85 of combustion or production technology, and penetration rate and emission reduction efficiency of emission controls (Zhao et al., 2014). Table 2 shows how emissions in each of the inventories are aggregated to the four primary source sectors (industry, transport, power, and residential) that we analyze in this paper. They were grouped in this way to be able to compare at the four source sector levels among the inventories, as this is how some of the inventories (i.e., MEIC) are structured. Here
- 90 we explain each of the emissions inventories in more detail.

2.1 REAS

REAS was developed collaboratively between the National Institute for Environmental Studies and Asia Center for Air Pollution Research, Japan (Kurokawa et al., 2013). The inventory comprises emissions data from 30 Asian countries and regions, including China divided into 33 sub-regions

- 95 (22 provinces, five autonomous regions, four municipalities, and two special administrative regions), between years 2000 and 2008 at a 0.25° longitude x 0.25° latitude horizontal resolution. A previous version of REAS spanned a longer time period and included projections of emissions (Ohara et al., 2007) but v2.1 is based on updated activity data and parameters. The emissions sources provided are power plants, combustible and non-combustible sources in industry, on-road and off-road sources in
- 100 transport, and residential and others such as agricultural activities and evaporative sources. Important proxies for gridding include rural, urban, and total populations, as well as road networks.

2.2 EDGAR

EDGAR was developed by the Joint Research Centre of the European Commission, in collaboration with the Netherlands Environmental Assessment Agency (European Commission, Joint Research

- 105 Centre (JRC)/Netherlands Environmental Assessment Agency (PBL), 2011). This database incorporated experiences of the dataset EDGAR v3.2 FT2000 from Olivier et al. (2001). EDGAR is a gridded emissions inventory of greenhouse gases, air pollutants and aerosols that spans 1970 2008 at a 0.1° longitude x 0.1° latitude horizontal resolution. The source sectors provided are energy, industrial processes, product use, agriculture, waste, and other anthropogenic sources. Country
- 110 emissions are compiled based on the International Energy Agency (IEA) energy statistics. Emission factors are taken from the EMEP/EEA air pollutant emission inventory guidebook (European Environment Agency, 2013) and other scientific literature. Gridding of national total emissions is done using several types of proxy data (population, road, power plants, animals, crop) as described in Janssens-Maenhout et al. (2013).

115 **2.3 MEIC**

MEIC is an inventory developed at Tsinghua University, Beijing, China, and provides source sector information for the 31 Chinese sub-regions (all those included in the REAS, except the two special administrative regions: Hong Kong and Macau) for 2008 and 2010 (Li et al., 2014; Zheng et al., 2014; Li et al., 2015; Liu et al., 2015). The MEIC model has a flexible spatial and sectoral resolution

120 and allows for gridding of the emission product into a user-specific grid including 0.25° longitude x 0.25° latitude horizontal resolution, as well as coarser grids. The emissions source sectors provided are power plants, industry, transport, residential and agricultural sources. Important proxy data for gridding of emissions includes population, roads, and power plants.

2.4 ZHAO

- 125 The inventory made at Nanjing University is a national inventory that estimates source sector emissions from all the 31 Chinese sub-regions, the same as MEIC (Zhao et al., 2013b, 2015; Cui et al., 2015; Xia et al., 2016). The inventory includes the national-level data for 2000-2008 and we use the available disaggregated emissions estimates for 2007 for comparison. The sectors provided are industry (including cement, iron & steel, other industrial combustion, and other industrial processes),
- 130 power, transport (including on-road and off-road), and residential. This inventory does not provide gridded emissions.

2.5 GAINS

The GAINS model was developed at the International Institute for Applied Systems Analysis and estimates global emissions, including those for the 31 sub-regions in China, as in MEIC and ZHAO,

- 135 as well as Hong Kong and Macau, as in REAS (Amann et al., 2008; Klimont et al., 2009). The GAINS model calculates emissions estimates in five-year intervals from 1990 to 2050, with the projection starting in year 2015. It has a large number of source sectors including energy, domestic, industrial combustion and processes, road and non-road transport, and agriculture, for which activities originate from international and national statistics. It provides output in various formats and
- 140 spatial resolution, including 0.5° latitude x 0.5° longitude horizontal grid. For this study, we use estimates from energy, domestic, transport, and industry sectors for the years 2000 and 2005, using the global dataset developed within the European Union project ECLIPSE (version V5a, http://www.iiasa.ac.at/web/home/research/research/rograms/air/Global_emissions.html) (Klimont et al., 2016). Sectoral proxies used in Representative Concentration Pathways (RCP) and Global Energy
- 145 Assessment (GEA), as well as population and selected industrial plant locations are used as important proxies for gridding.

3 National and Regional Comparisons

To better understand the differences among anthropogenic emissions estimates of four air pollutant species, we first analyzed differences in national total emissions estimates between years 2000 and 2008. For each of the species, we further compared these estimates in seven different regions (Fig. 1) for four source sectors separately. In the following sections, we first describe the differences at the national level, and then at the regional level for each species.

3.1 National Level Comparisons

Fig. 2 illustrates China's national total emissions for the four air pollutant species of our interest (CO,
SO₂, NO_x, and PM₁₀) as well as CO₂ estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS, between 2000 and 2008, along with other published study estimates. We also used one million Monte

Carlo samples from all emissions inventories, sector by sector, to create a composite emissions estimate for each species. For the inventories that provided a standard deviation or uncertainty, we used the information and assumed either a normal or log-normal distribution based on the information

160 provided. If such information was not available, we used the relative uncertainty percentage provided by REAS for a sector for each species to estimate standard deviation and assumed normal distribution.

We find the largest difference, ranging from 65 - 94 Tg/year (87 - 106%), between REAS and EDGAR emissions estimates for total CO in China with REAS exceeding EDGAR throughout the

- 165 2000 2008 time period (Fig. 2). We further find that the major sectors leading to the differences are industry and transport (Fig. 3). Indeed, between REAS and EDGAR, 39% of the difference in national total CO emissions stems from the industry sector in 2000. By 2008, the emissions difference in the industry sector contributes 51% of the total emissions difference for CO emissions in China.
- 170 What is the cause of this large difference within the industry sector? Coal combustion plays a large role in CO emissions from this sector in the REAS estimate and 98.6% of the combustible industrial emissions are due to coal in 2008. The comparison of fuel use statistics among REAS, EDGAR, and GAINS for 2000 (Fig. 4) and net emission factors per sector among REAS, EDGAR, GAINS, and MEIC (Fig. 5) are useful in understanding the reason behind the differences. Coal use in industry
- 175 between REAS and EDGAR shows similar values but there is a large difference in emission factors for industrial CO between REAS (2.2 ton CO/TJ) and EDGAR (1.1 ton CO/TJ). Because emission factors are related to each technology type, penetration of the technology, uncontrolled emission factor and the emission reduction efficiency of each technology type, these factors all contribute to differences. Obtaining estimates for CO is particularly difficult because of the many technology
- 180 types that exist for emissions reduction. For the transport sector, estimated emissions by EDGAR are still lower than those of REAS (Fig. 3), even with its higher fuel use and emission factor, most likely because the modeling of superemitters has been omitted in EDGAR.

The smallest CO source sector is power and it has the smallest difference among the inventories. Power emissions only contribute to 1.9, 3.1, 1.1, and 0.8% of the national emissions in REAS,

- 185 EDGAR, MEIC, and ZHAO, respectively, in 2008 for the former three and in 2007 for ZHAO. GAINS estimates 1.0% of its national emissions comes from power in 2005. REAS estimates a 2.3 Tg (159%) increase in CO emissions from the power sector between 2000 and 2008, while EDGAR only estimates a 0.43 Tg (15%) increase in the same time period. At the national level, the difference in CO emissions from the power sector between REAS and EDGAR decreased from 50% to 13%
- 190 between the same period (2000-2008).

The difference for PM_{10} between REAS and EDGAR is also not insignificant and ranges between 2.7-7.8 Tg yr⁻¹ (25 and 59%) over time (Fig. 2). Similar to CO, REAS estimates the highest and EDGAR estimates the lowest national PM_{10} emissions. As shown in Fig. 3, the major differences

arise mainly from the industry sector, where EDGAR emissions show significantly lower estimates

compared to those of REAS and all the others. Opposite is the case for power sector emissions and 195 EDGAR emissions are double those of REAS and others. For PM₁₀, EDGAR estimates lower fuel use for coal and oil in industry than REAS and higher fuel use for coal and gas in power sector than REAS (Fig. 4). The net emission factor for PM_{10} in industry is also lower for EDGAR than REAS and the opposite is the case for power (Fig. 5). EDGAR thus estimates lower emissions for industry, while estimating higher emissions than REAS for the power sector (Fig. 3). The large 200 difference in industrial PM₁₀ emissions may also be due to differences in removal efficiency of a

certain technology embedded in emission calculations among inventories.

The power emissions for NO_x dominate the national total for REAS, EDGAR, and Zhang et al. (2009) (Fig. 3). 10.9 Tg yr⁻¹ (46%) and 10.2 Tg yr⁻¹ (51%) of the national NO_x emissions are

- estimated to come from the power sector in REAS and EDGAR, respectively, in 2008. 9.2 Tg yr⁻¹ 205 (44%) are estimated to come from the power sector in 2006 in the INTEX-B inventory by (Zhang et al., 2009). Streets et al. (2003) estimated power to be the dominant source sector, contributing 4.4 Tg yr⁻¹ (39% of NO_x emissions) in 2000, followed by 2.8 Tg yr⁻¹ each (equal 25% contribution) from industry and transport. The national emissions inventories, however, do not show power sector
- emissions dominating for NO_x . For MEIC, industrial emissions are estimated to be slightly higher 210 than those from the power sector. For ZHAO, the two sources are similar in magnitude. 33% (36%) and 35% (35%) of the total emissions equalling 8.6 Tg yr⁻¹ (9.4 Tg yr⁻¹) and 8.3 Tg yr⁻¹ (7.9 Tg yr⁻¹) are estimated to come from the power (industry) sector in these two national inventories MEIC in 2008 and ZHAO in 2007, respectively. One of the possible reasons for this is due to the
- difference in the net emission factors among emission inventories (Fig. 5). MEIC estimates much 215 higher emission factors for NO_x emissions from the industry sector than from power, unlike other inventories that estimate the opposite (REAS and GAINS) or fairly close to each other (EDGAR).

The differences for the other species are smaller, although it is clear that Lamarque et al. (2010) estimates much lower emissions for both NO_x and SO_2 , compared to others (Fig. 2). The range of

- the absolute difference between REAS and EDGAR for CO_2 and SO_2 are 4.25 553 Tg yr⁻¹ and 220 0.75 - 7.9 Gg yr⁻¹, respectively, between 2000 and 2008. MEIC and ZHAO emissions estimates fall between the REAS and EDGAR estimates most of the time, although they are closer to the REAS estimates, which are higher than those of EDGAR, for most species. GAINS estimates sometimes do not fall between the REAS and EDGAR estimates but the differences are still small. The timing 225 of the SO₂ emissions reduction in 2007 in REAS coincides with what is reported in Zhang et al.

3.2 Regional Level Comparisons

(2009), Klimont et al. (2009), and Lu et al. (2011).

When we compare emissions in the seven regions within China (East, North, Northeast, Central, Southwest, Northwest, and South, as shown in Fig. 1), we find larger differences than at the national 230 level for almost all species (Figs. 6 - 9). We compare in detail the differences among emissions inventories for each species per region and for each source sector below.

3.2.1 Carbon monoxide, CO

Atmospheric CO is mainly a result of incomplete combustion of fossil fuels and biofuels and exposure to ambient CO is harmful to human health (Aronow and Isbell, 1973; Stern et al., 1988; Allred
et al., 1989; Morris et al., 1995). CO emissions are also important precursors to the formation of tropospheric O₃, which also has harmful human health impacts, including increased asthma exacerbations, decreased pulmonary function, and increased mortality (Schwartz et al., 1994; Mudway and Kelly, 2000; Levy et al., 2005). Because of the existence of diverse emissions sources with various emissions control technologies in China, it has been a challenge to estimate CO emissions accurately,

240 using a bottom-up methodology with emission factors and activity levels (Streets et al., 2006). This explains why we see the largest difference in CO emissions estimates at the national level compared in Fig. 2 to all other species.

Fig. 6 shows the seven regional CO emissions estimates from each source sector. For CO emissions, industry is the only source sector that shows a steep increase over time in all regions for REAS

- and EDGAR estimates, especially between 2002 and 2008. GAINS also shows an increase between 2000 and 2005. Due to the rapid increase in its emissions, by 2008, industry is the largest source sector for CO in the two largest source regions East and North regardless of which inventory. REAS CO emissions estimates are consistently higher than those of EDGAR across all regions except for the Northeast for industry emissions, and MEIC, ZHAO, and GAINS CO emissions estimates for
- 250 this sector generally fall between the estimates of REAS and EDGAR. The two regions where this does not apply are Central and Northwest, and their industrial CO emissions estimates by MEIC, ZHAO, and GAINS are higher than the estimates by the other two emissions inventories. Analysis at the source sector level reveals that the majority of the differences in CO emissions among the inventories stem from the industry sector and that they are, in many regions, increasing over time.
- 255 The second largest CO source is the residential sector and the estimates by the national inventories MEIC and ZHAO are always higher in all regions than the regional inventory REAS and the global inventory EDGAR estimates. GAINS estimates the residential sector to be the largest source sector and their estimates are also usually higher than REAS and EDGAR in almost all regions, except in the Southwest and the South in 2005, where the REAS and GAINS estimates are close to each
- 260 other. EDGAR estimates for residential sector emissions are the lowest among the inventories analyzed here, because it does not include provincial but rather uses the national statistics-based IEA estimates for coal use in residential sector, leading to lower activity level (Fig. 4). On the other hand, GAINS emissions for this sector are the highest because it is unique in considering factors which are technology specific, rather than using one factor per fuel for the whole residential sector. For

265 example, there are significant differences in emissions for different types of stoves and boilers in the residential sector and these technology-specific data are incorporated into the GAINS model.

The third largest CO source and the source sector with the second largest difference after industry is transport, contributing 45.6% (34.4%) of the total difference in 2000 (2008). Emissions from North and East regions contribute to these large differences. Both REAS and EDGAR emissions

- 270 inventories show decrease at the national level, although at the regional scale, the change is variable, ranging from -0.59 Tg (-1.5 Tg) for EDGAR to -1.8 Tg (1.4 Tg) for REAS between 2000 and 2008 in the North (East). This difference might be due to a couple of reasons. First, emission factors and reduction measures assumed can be different. For example, EDGAR may be estimating much larger emissions reduction in newer vehicles with more stringent emission standards. Second, the number
- 275 of vehicles assumed in different vehicle types is different among the inventories (Fig. 10), even if the total number may be similar. For REAS, the number of vehicles of each type (passenger cars, buses, light and heavy duty trucks, and motorcycles) in 2000 was taken from Borken et al. (2008) and extrapolated to 2008, using trends from National Bureau of Statistics (2001-2009) (Kurokawa et al., 2013). Emission factors due to control strategies and policies in REAS stem from estimates
- in Borken et al. (2008) and Wu et al. (2011), as explained in Saikawa et al. (2011). For EDGAR, the fleet distribution is based on the international statistics from the International Road Federation (IRF, 1990, 2005, 2007) which were analyzed in the framework of the EU 'Quantify' project (Borken et al., 2008). Zhang et al. (2009) estimated an 11% decrease in CO from the transport sector between 2001 and 2006 due to emissions control technologies, despite the doubling of the number of vehicles in the same period. We will analyze the transport emissions in more detail in Section 3.3 as we have

some more disaggregated data for this sector available for comparison.

At the regional level, the ranking of source sectors does not always hold and also changes over time. For Northwest, emissions from the residential sector are estimated to be the largest in all years in all inventories. In Southwest, REAS estimates slightly higher industrial emissions (6.6 Tg yr⁻¹ in 2000 and 12.4 Tg yr⁻¹ in 2008) than residential emissions (6.3 Tg yr⁻¹ in 2000 and 9.9 Tg yr⁻¹ in 290 2008) but EDGAR estimates higher transport emissions (2.5 Tg yr⁻¹) than industrial (2.0 Tg yr⁻¹) in 2000. Similarly, in the South, REAS estimates industry to be the largest source sector (6.4 Tg yr^{-1}) followed by residential (5.3 Tg yr^{-1}) and transport (4.5 Tg yr^{-1}) in 2008, whereas EDGAR

 295 yr^{-1}) and transport (0.73 Tg yr⁻¹) with much lower emissions than the other two in the same year.

This clearly illustrates the importance of constraining emissions at the disaggregated levels.

The East, encompassing the Pearl-River-Delta and the industrial coast, is the largest source region of CO. In 2008, 32, 27, and 26% of the national total CO emissions from REAS, EDGAR, and MEIC estimates, respectively, were emitted from this region. Similarly, ZHAO (GAINS) estimates 30% (29%) of the national total CO emissions is from the East in 2007 (2005). CO emissions from

estimates residential to be the largest (3.7 Tg yr^{-1}) , followed by industry in a close second (3.4 Tg)

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the industry sector in the East, in particular, show large differences, and the absolute difference more

than doubles from 2000 to 2008. In 2008, there is a 22.4 Tg yr⁻¹ difference in CO emissions within the industry sector between REAS and EDGAR, which constitutes a 64% difference between the two emissions estimates in the East in that year. This difference makes up 25% of the difference between

- 305 the two national total CO emissions estimates. The difference between the REAS and EDGAR emissions estimates for the transport sector for this region is also increasing and is 10.1 Tg yr⁻¹ in 2008, equivalent to 29% of the regional total CO difference and 11% of the national CO difference. One thing to note about this region is that EDGAR CO estimates for the transport sector are decreasing over time, whereas those of REAS indicate the opposite.
- The North is the second largest source region of CO, and it contributes 21, 14, and 21% of the 310 national total CO emissions for REAS, EDGAR, and MEIC estimates, respectively, in 2008. ZHAO (GAINS) estimates 18.5% (18.1%) of the national total CO emissions come from this region in 2007 (2005). Combined with the East emissions, the two regions contribute 53, 42, 47, 48, and 47% of the emissions in REAS, EDGAR, MEIC, ZHAO, and GAINS, respectively, in 2008 for the
- former three, 2007 for ZHAO, and 2005 for GAINS. The pattern shown for East and North, the 315 more developed regions in China, is similar, and the only difference is that EDGAR estimates larger residential emissions compared to transport emissions in the East, whereas the opposite is the case for the North in the early 2000s.

3.2.2 Sulfur dioxide, SO₂

- 320 SO₂ leads to acid rain through sulfuric acid deposition, destroying buildings by corroding metals and deteriorating paint and stone. Furthermore, it harms aquatic and terrestrial ecosystems. SO₂ is also a precursor of sulfate aerosols that scatter radiation, leading to direct cooling of the atmosphere. Sulfate aerosols also act as condensation nuclei, making clouds more reflective and prolonging the lifetime of clouds, enhancing the cooling impact (Haywood and Boucher, 2000; Ramanathan et al., 2001). 325

Fig. 7 shows the seven regional SO_2 emissions estimates for each source sector. For SO_2 emissions, the power sector is the largest source sector in most years for both REAS and EDGAR, and 38 - 54% (52 - 61%) of national total SO₂ emissions are from the power sector in REAS (EDGAR) between 2000 and 2008. Contrary to CO emissions, we find a large divergence between REAS and

- 330 EDGAR power sector emissions estimates during 2000 - 2008 across all regions. While EDGAR SO₂ power emissions estimates continue to increase over time, those of REAS peak in that time range, although the specific year is not uniform across the regions. Up to the peak in the REAS estimates, REAS and EDGAR follow similar trajectories. However, REAS SO2 emissions in the Central and the Northwest start to decrease in 2004, in 2005 in the South, East, and North, and in 2006 in
- the Northeast and the Southwest. 335

The large difference in SO₂ emissions from the power sector between REAS and EDGAR is due to the difference in the assumed timing of the installation of FGD in coal-fired power plants. Newly designed policy incentives and an increase in policy inspection have led to an increase in the installation of FGD in China and the percentage of plants with FGD increased from 10% to

- 340 71% between 2006 and 2009 (Xu, 2011). The number of power plants is listed in Table 3. While EDGAR assumed a delayed penetration of FGD (1%), electrostatic precipitators (6%) and flue-gas recirculation (4%) leaving 90% of power plants still fully-uncontrolled in 2008, REAS estimated a more optimistic installation scenario, especially for large power plants and referred to Lu et al. (2011) in deciding implementation rates of FGD to power plants in China. For example in 2007, Lu
- et al. (2011) used the range of 51.4 95%, with the mean of 73.2%, based on the Chinese Ministry of Environmental Protection (MEP) official data (2009) reporting of SO₂ removal efficiency of FGD and applying the triangular distribution with the ideal removal efficiency of 95% (Zhao et al., 2011). This explains why REAS emissions estimates from the power sector are closer to the emissions estimates by MEIC, and those by Lu et al. (2011), as seen in Figs. 3 and 7. The largest emissions
- 350 decrease from the power sector are seen in the East and North regions, where there were 250 and 206 power plants, respectively, reinforcing that this difference is due to the FGD implementation assumption in power plants.

The second largest source sector for China's SO₂ emissions is industry. Nationally, it contributes 13 (53%), 17 (33%), 17 (53%), 14 (44%), and 9.3 (27%) Tg yr⁻¹ of total SO₂ emissions in REAS,

EDGAR, and MEIC for 2008, ZHAO for 2007, and GAINS for 2005, respectively. In some regions, such as the Northeast, there is very little difference among inventories. On the other hand, we see a much larger difference in the Southwest. While EDGAR estimates industry to be the second largest source sector in this region, constituting 31 - 37% of regional emissions, all other emissions inventories estimate industry to be the largest source sector in the region, constituting 46 - 60% of the
regional total. Similar to its estimates for CO emissions, REAS tends to estimate higher emissions

from the industry sector in most of the regions.

 SO_2 emissions discrepancies in the two other sectors remain relatively small and constant across all regions, with the residential sector emissions in the Southwest as the only exception. The residential sector emissions difference in the Southwest between EDGAR and REAS estimates have decreased from 354 Gg yr⁻¹ in 2000 to 215 Gg yr⁻¹ in 2008.

3.2.3 Nitrogen oxides, NO_x

 NO_x plays an important role in the formation of tropospheric O_3 and nitrate aerosols. The NO_x emissions trend in Asia, and especially in China, has been an important topic, due to the rapid changes that have been observed in the past two decades (Richter et al., 2005; Gu et al., 2014). Fig. 8

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shows the seven regional NO_x emissions estimated for each source sector. The difference in this sector is the largest in the East and the Northeast regions, both with 470 Gg yr⁻¹ in 2008.

The large differences among the emissions inventories stem from the transport sector in the East, North, South, and Northwest. For the transport sector, the East has an increasing difference over time, changing from 0.40 Tg in 2000 to 1.3 Tg in 2008. While transport contributes 27 - 30% of the

- 375 regional total emissions for REAS in the East, it only contributes 15 19% for EDGAR. MEIC estimates the transport sector in the East to contribute 25% of the regional total NO_x emissions. In the North, South, and Northwest, the difference in the transport sector emissions among the inventories can also be as high as 450, 355, and 326 Gg yr⁻¹, respectively. The key reasons why the differences are large and they are growing are two-folds. First, as we explain later in Section 4, the differences
- 380 in the allocation of fuel (gasoline and diesel) and the differences in vehicle categories play a role. Second, the pace of the implementation of measures assumed among different inventories is different.

Little to no emissions control technologies for NO_x has been developed and promoted in China for the power and industrial combustion sectors and this is the main reason why we see a large increase for NO_x emissions. China only used low- NO_x combustion technology and started to install selective

reduction methods after 2005 (Zhao et al., 2013a). The only other NO_x mitigation strategy for China was emissions standards for reducing tail pipe emissions from vehicles (Zhao et al., 2013a). For example, there is no national NO_x emissions standard for coal-fired industrial boilers, as opposed to the vehicle emission standards that have been tightened over the years.

390 3.2.4 Coarse particulate matter, PM_{10}

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China's PM_{10} emissions have been increasing rapidly and they contribute approximately 21.6 (15.2) Tg yr⁻¹ of the 38.3 (39.3) Tg yr⁻¹ total PM_{10} emissions from 22 Asian countries, including Afghanistan, Bangladesh, Bhutan, Nepal, Sri Lanka, India, Maldives, Pakistan, South Korea, North Korea, China, Japan, Singapore, Taiwan, Laos, Cambodia, Brunei, Myanmar, Philippines, Thailand, Vietnam, and

395 Indonesia, in the REAS (EDGAR) estimate. These differences between REAS and EDGAR estimates indicate the large differences in China, as well as in other parts of Asia. Here, we only discuss primary emissions of PM₁₀, emitted directly from anthropogenic sources.

Fig. 9 shows the seven regional PM₁₀ emissions estimates for each source sector. The largest source sector, as well as the largest emissions difference, stems from the industry sector. Industrial emissions contribute 64, 19, and 78% of the total PM₁₀ emissions in REAS, EDGAR, and MEIC for 2008, respectively, and 65% (50%) for ZHAO (GAINS) for 2007 (2005). As illustrated in the low industrial contribution in 2008 in EDGAR its industrial emissions increased by 1.3 Tg from 2000 to 2008, while those of REAS increased by 5.8 Tg in the same period. The reason for this large increase in industrial PM₁₀ emissions is due to the fast growth of industry and limited stringency of air

405 quality legislation and its poor enforcement (Zhao et al., 2013a). In addition, uncertainty accounting for fugitive emissions due to leaks or other unintentional releases adds to the difference among the inventories. For industrial PM_{10} emissions, REAS estimates are always consistently higher than those of EDGAR in all regions, and the difference between the two inventories is four to five-fold, constituting 61 - 74% of the total differences.

- 410 We see relatively little change in differences among the inventories between 2000 and 2008 for transport and residential sectors. It is also important to point out that the spatial distribution of emissions in some of the inventories, especially the global ones, are often more static than the national ones due to the limited local information, although this static nature over time of the global inventories is not only for PM_{10} but also applies to other species as well. There are, however, some
- 415 interesting sector-dependent differences. First, GAINS estimates higher residential emissions than REAS and EDGAR in all regions in both 2000 and 2005 except in the South in 2005. Second, REAS estimates are not always higher than those of EDGAR for the residential sector emissions. In the Northeast, REAS PM₁₀ emissions estimates are higher than those of EDGAR. For the Southwest and the North, REAS emissions estimates are higher than EDGAR estimates only for the period
 420 2002 2005.

4 Road Transport Sector Comparison

Rapid growth in the number of vehicles has created a significant air quality challenge in China. Many have researched the importance of on-road transport emissions on Beijing's (Hao et al., 2001; Westerdahl et al., 2009) and China's air quality (Fu et al., 2001; Walsh, 2007; Saikawa et al., 2011).

- We found significant differences in CO and NO_x emissions in the transport sector and here we analyze the differences for these emissions in more depth by focusing on both on-road and off-road transport emissions. Here, we first compare the contribution of different vehicle categories to the total vehicles in REAS, EDGAR, and GAINS. Then, we compare on-road and off-road emissions estimates of CO, NO_x , SO_2 , and PM_{10} at the national level, as well as for each region.
- 430 Comparing the contribution of various gasoline (Fig. 10a) vehicles among the three inventories, EDGAR is very different from the other two. The similar comparison for diesel vehicles (Fig. 10b) reveals even a larger difference. As stated earlier for the industrial sector, it is likely that emission factors and/or the technology levels estimated within each of the vehicle types are causing the differences. EDGAR emission factors specifically for on-road vehicles is not available but comparing the
- 435 net transport-sector emission factors between EDGAR and GAINS (Fig. 5), GAINS has 5.6 times higher value per unit of fuel than EDGAR. The lack of modeling superemitters in EDGAR is also contributing significantly to the differences. It is also possible that something more fundamental, such as the definition of vehicle types, is causing the differences.

In the following section, we compare national on-road and off-road transport emissions first among REAS, EDGAR, ZHAO, and GAINS, and then in the seven regions within China (East, North, Northeast, Central, Southwest, Northwest, and South, as shown in Fig. 1), for REAS, EDGAR,

and ZHAO. We compare in detail the differences among emissions inventories for each species per region and for each source sector below.

4.1 Carbon monoxide, CO

- Fig. 11 shows the national and seven regional CO transport emissions estimated in REAS, EDGAR, ZHAO, and GAINS (national estimate only), separated into on-road and off-road emissions. The figure clearly shows that the difference in this sector stems from on-road emissions. 99% of the difference between REAS and EDGAR CO transport emissions are from on-road at the national level, and in the East, we see up to a difference of 99.4% at the regional level. Indeed, at the national
- 450 and all regional levels, there is more than an order of magnitude of difference in emissions between REAS and EDGAR on-road emissions. ZHAO on-road emissions estimates are always in between REAS and EDGAR estimates and ZHAO off-road estimates are always higher than both REAS and EDGAR.

4.2 Nitrogen Oxides, NO_x

Fig. 12 shows the national and seven regional NO_x transport emissions estimated in REAS, EDGAR, and ZHAO, separated into on-road and off-road emissions. Contrary to the CO emissions, there are many regional differences in these emissions estimates. At the national level, REAS (ZHAO) estimates 42-56% (49%) higher for on-road emissions compared to EDGAR. Off-road emissions are much more constrained among the three emissions inventories and REAS and EDGAR give similar estimates between 2005 and 2007.

The East is estimated to contribute 28-38, 6.3-6.8, and 37% of the total transport emissions in REAS, EDGAR, and ZHAO, respectively. REAS (ZHAO) emissions estimates are 5.6-7.4 (6.2) times larger than EDGAR on-road emissions, and 2.6-9.5 (6.7) times larger than off-road emissions. For NO_x emissions, although on-road emissions are still larger in most of the regions, off-road emissions.

- 465 sions are also important and are mostly increasing in both REAS and EDGAR. For the East, REAS estimates an increase from 307 Gg yr⁻¹ in 2000 to 1100 Gg yr⁻¹ in 2008 in off-road emissions. For the Northwest, EDGAR estimates larger emissions from off-road compared to on-road for NO_x, which we do not see in either REAS or ZHAO. REAS estimates a higher growth rate for off-road emissions and their emissions estimates increase from 28.4 Gg yr⁻¹ in 2000 to 75.1 Gg yr⁻¹ in
- 470 2008, while EDGAR off-road emissions estimates only increase from 98.5 Gg yr⁻¹ to 110 Gg yr⁻¹ over the same time period. The large emissions differences in the region are most likely due to much greater railway emissions by coal and diesel locomotives assumed in the EDGAR inventory, compared to REAS.

4.3 Coarse Particulate Matter PM₁₀ and Sulfur dioxide SO₂

475 Fig. 13 shows the national PM_{10} and SO_2 on-road and off-road emissions estimated in REAS, EDGAR, and GAINS. PM_{10} shows a good agreement for on-road emissions between REAS and GAINS, although EDGAR on-road is much lower. The low emissions estimates for EDGAR for PM_{10} is most likely due to the lack of superemitters in EDGAR, since those are the primary emitters. On-road emissions for SO₂ also shows a good agreement, especially between EDGAR and GAINS, although REAS values show an increase in the late 2000s that we do not find in the other

- 480 GAINS, although REAS values show an increase in the late 2000s that we do not find in the other two inventories. SO₂ is calculated differently than for the other species in REAS, based on gasoline/diesel consumption instead of vehicle category. This might also be the reason for the difference among the inventories.
- Off-road emissions are in especially good agreement for PM_{10} among the three inventories. However, they diverge quite significantly for SO₂ emissions. GAINS, in particular, has low emissions estimates for off-road SO₂ emissions, although it estimates high emissions for CO and PM_{10} . Based on Fig. 5, it is most likely due to the high emission factors GAINS have for these off-road vehicles in the transport sector.

5 Impacts on air quality

490 5.1 Model description

To assess how these differences in emissions inputs affect air quality simulation results, we used the Weather Research and Forecasting model coupled with Chemistry (WRF-Chem) version 3.5 (Grell et al., 2005). The model domain covers much of the Asian region, with a horizontal resolution of 20×20 km with 31 vertical levels and China at its center (Fig. 15). The initial and lateral chemical

- 495 boundary conditions are taken from a present-day simulation of the NOAA Geophysical Fluid Dynamics Laboratory (GFDL) global chemistry-climate model AM3 (Naik et al., 2013), driven by the global gridded emissions from the inventory of Lamarque et al. (2010). The meteorological data are obtained from the National Center for Environmental Prediction (NCEP) Global Forecast System final gridded analysis datasets. We used Carbon-Bond Mechanism version Z (CBMZ) (Zaveri and
- 500 Peters, 1999) for gas-phase chemistry and the Model for Simulating Aerosol Interactions and Chemistry (MOSAIC) (Zaveri et al., 2008) for aerosol chemistry. The rest of the model setup (aerosol dry deposition, wet deposition, photolysis, radiation, and microphysics) is the same as applied in our previous study (Zhong et al., 2015).
- We chose the three emissions inventories that provided gridded emissions and are targeted at dif-505 ferent scales: EDGAR at global, REAS at regional, and MEIC at national. In addition, EDGAR estimates the lowest emissions for most species, whereas REAS estimates the highest and thus provides a range of air quality simulations as a result of varying emissions. We then performed model simulations for January and July for 2008, using each of these inventories. Because MEIC only covers China, we applied REAS emissions outside of China for the simulation with MEIC. For
- 510 biomass burning emissions, we used the Fire INventory from NCAR (FINN) (Wiedinmyer et al., 2011) and for biogenic emissions, we used the Model of Emissions of Gases and Aerosols from Nature (MEGAN) interactively within WRF-Chem (Guenther et al., 2012). For aircraft emissions,

we used emissions developed for the Hemispheric Transport of Air Pollution (HTAP) for the year 2008 (Janssens-Maenhout et al., 2015). In order to focus on differences in air quality due to differing

- anthropogenic emissions estimates of gaseous pollutants and PM, we did not include dust simu-515 lation in this study. However, sea salt is calculated online (Gong, 2003). Before the beginning of each monthly simulation, the model was spun-up for ten days to ventilate the regional domain. The model simulation including dust has been validated with existing measurements for the year 2007 in (Zhong et al., 2015) and here we focus on differences in air quality simulation due to differing 520 gridded anthropogenic emissions inputs.

5.2 Simulated results and discussion

Fig. 14a illustrates the spatial distribution of January emissions for CO, NO_x , SO_2 , PM_{10} , and NMVOC that we used as inputs for the WRF-Chem simulations. As mentioned earlier, CO and PM₁₀ show high variations and the emissions are especially concentrated in the eastern part of China. Al-

though the difference in national SO_2 emissions was not as large as those of the other two species, 525 Fig. 14a clearly illustrates that REAS estimates much larger emissions compared to the other two inventories.

Fig.15a compares the simulated monthly mean PM₁₀ concentrations, as well as that of CO, NO₂, SO_2 , and O_3 mixing ratios in January 2008, using the three inventory estimates as emissions in-530 puts. These differences in simulated concentrations or mixing ratios of pollutants are solely due to the emissions used as model inputs. Overall, the simulated monthly means show similar spatial distributions. All three simulations show high levels of CO, NO₂, SO₂, and PM₁₀ in the Beijing-Tianjin-Hebei area in the North, Shanxi province in the North, and Sichuan basin in the Southwest. In contrast, the mixing ratios of O_3 are relatively low over the same regions. Despite the similar 535 spatial distributions, concentrations of the simulated monthly means differ substantially.

For CO, both simulations using REAS and MEIC result in higher mixing ratios than when using EDGAR. We quantified the regional monthly mean of each simulation by averaging all grid cells in each region, as illustrated in Table 4. The REAS and MEIC regional monthly means are 270 - 470 ppbv (169 - 194 ppbv) higher in the polluted area in the Central (the East) region, than the EDGAR

- simulation. For NO₂, the largest differences in regional monthly mean occur between simulations 540 using EDGAR and MEIC emissions, mainly in the Central (8.1 ppbv), followed by the East (7.2 ppbv) and the Northeast (3.3 ppbv). These regions are where the differences in emissions are the largest as well. For SO₂, both simulations using REAS and MEIC show differences in monthly mean less than 30% in most regions compared to those with EDGAR emissions, except in the Southwest,
- 545 where REAS and MEIC estimates are 1.5 and 1.7 ppbv higher, respectively, than EDGAR estimates. For PM₁₀, EDGAR simulation is 20 - 60 μ g m^{-3} lower than the other two in most regions. For example, MEIC simulation estimates 15 μ g m^{-3} (103%) higher monthly mean in the Northeast and 20 μ g m^{-3} (85%) higher in the Southwest than EDGAR. REAS simulation estimates more than

55% higher monthly mean PM10 concentrations than EDGAR in most regions, with the highest dif-

- ference (76%) occurring in the Northeast. The largest absolute difference of 67 µg m⁻³ in a regional monthly mean between MEIC and EDGAR simulations is found in Central. Based on the observations from nine stations in Wuhan within the Central region, the monthly mean PM₁₀ concentrations in January were 130 µg m⁻³ (Feng et al., 2011) and this is closer to the simulated values using the MEIC (REAS) emissions inventory of 47.4 (50.6) µg m⁻³, compared to the value using the EDGAR emissions inventory of 32.3 µg m⁻³, although the model simulations are largely underestimated.
- For O_3 , simulations using REAS and EDGAR inputs show only a slight difference in monthly mean of 1-5 ppbv in January. However, O_3 mixing ratios using MEIC emissions are much lower than those using EDGAR emissions in the Central (31%) and the East (25%). MEIC's low anthropogenic VOC emissions in combination with high NO_x emissions in these regions (see 14a) bring much
- 560 higher NO_x titration and produce a VOC-limited environment, as illustrated in Figure 16a. For these two regions, despite the REAS and MEIC having similar NO_x emissions, their VOC emissions differ by more than 10 times. EDGAR emissions are the lowest for NO_x for both the Central and the East but their estimates are the largest for VOCs in the Central and the second largest in the East among the three inventories. In both cases, simulations using EDGAR inventory lead to the largest
- O_3 mixing ratios, due to the limited titration of NO_x during the night time. The NO_x mixing ratio in these two regions estimated in EDGAR is much lower compared to that in REAS and MEIC, as seen in Fig. 15a. This result illustrates the importance of constraining NO_x and VOC emissions in the East and Central regions in understanding the way to mitigate O_3 pollution for the future.
- We also analyzed the differences of three simulations in July 2008 (Fig. 15b). We find a difference of more than 50% for CO, NO₂, SO₂, and PM₁₀ in one or more regions. The Central and the East again showed the largest differences, as found in January. There was a 34 μ g m^{-3} difference in PM₁₀ in Central China between REAS and EDGAR and a 129 ppbv difference in the East for CO between REAS and MEIC. Again, Wuhan mean for July of 70 μ g m^{-3} of PM₁₀ was better captured by MEIC (REAS) of 52.0 (53.5) μ g m^{-3} , compared to that by EDGAR of 36.0 μ g m^{-3} . The difference we
- 575 find for O_3 in East, North, and Central are also important, due to the high mixing ratio estimated in REAS is close to the 8-hr WHO guideline of $100\mu g m^{-3}$. From Fig. 16b, it is clear that the difference of O_3 mixing ratio in these three regions is due almost solely to the VOC emissions between REAS and MEIC.

6 Conclusions

580 In this study, we compared five emissions inventories of anthropogenic CO_2 and air pollutant emissions in China at national and regional levels from four source sectors. The REAS and EDGAR inventories have been developed and maintained for years and have been extensively used for air quality modeling over the Asian continent, while the two national emissions inventories (MEIC and ZHAO) were recently developed, and few air quality modeling studies have been published using

the data from these inventories at this time. GAINS has its roots in the RAINS-Asia model dating back to early 90's project covering primarily SO₂ and later on developed to include more pollutants. The GAINS dataset used here originates from a global project and has been used in several air quality and climate modeling exercises. This analysis reveals large differences in emissions estimates among the existing inventories. Furthermore, analysis of regional and sector specific emissions, as
opposed to total national emissions reveals differences in emissions from certain sectors that would not have been noticed by only analyzing the national total emissions.

We find that there is a significant need to better constrain emissions at the source sector and regional levels. Transparency in what inputs are used to create different emissions inventories is critical for a more thorough comparison. CO emissions differ the most, and those from the transport 595 sector, especially the on-road transport emissions, need to be better constrained. Industrial emissions also tend to have a large difference among inventories and SO₂ emissions from the power sector also need to be assessed, especially for recent years. The East and the North are the two largest emitting regions and more efforts are needed to understand emissions from these areas.

- Emissions inputs have a large impact on air quality simulation results in China nationally, and 600 more prominently within the regions. Different emissions inputs lead to 67 μ g m⁻³ (34 μ g m⁻³) monthly mean difference in PM₁₀ concentrations in Central China in January (July). Similarly, we found 470 ppbv difference in January in Central and 129 ppbv difference in July in the East for CO. We also found that all the three inventory emissions estimates create a VOC-limited environment in the Central and the East that produces much lower O₃ mixing ratio estimates, compared to the
- 605 simulations using REAS and EDGAR estimates in January. The difference in emissions inputs leads to 15 ppbv difference in O_3 in Central China in January. In July, we find 8.5 ppbv difference in North, where REAS simulations lead to a monthly-mean of 63 ppbv O_3 . Our results illustrate that a better understanding of Chinese emissions at more disaggregated levels is essential for finding effective mitigation measures for reducing national and regional air pollution in China.
- 610 Acknowledgements. We thank the editor and the two anonymous reviewers for constructive comments that improved the manuscript. We also thank Geoffrey Martin and Raquel Soat for their assistance in this project. The project was partly supported by the National Science Foundation (grant number AGS-1350021). REAS is supported by the Global Environmental Research Fund of the Ministry of the Environment Japan (S-7 and S-12). The NCEP GFS data used for this study are from the Research Data Archive (RDA) which is maintained
- 615 by the Computational and Information Systems Laboratory (CISL) at the National Center for Atmospheric Research (NCAR). The data is available at http://rda.ucar.edu/datasets/ds083.2/. We would like to acknowledge high-performance computing support from Yellowstone (ark:/85065/d7wd3xhc) provided by NCAR's Computational and Information Systems Laboratory, sponsored by the National Science Foundation.

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Figure 1. Seven regions in China used for analysis in this paper



Figure 2. National total emissions estimates for CO_2 , CO, SO_2 , NO_x , and PM_{10} estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS between 2000 and 2008, along with other existing emissions inventory estimates.



Figure 3. National emissions estimates for CO₂, CO, SO₂, NO_x, and PM₁₀ by source sector estimated by REAS, EDGAR, MEIC, ZHAO, GAINS, and ZHANG between 2000 and 2008.



Figure 4. Total fuel use estimates by REAS, EDGAR, and GAINS for the four source sectors in 2000.



Figure 5. Net emission factors for CO, SO₂, NO_x, and PM₁₀ used among REAS and EDGAR for 2008, MEIC for 2010, and GAINS for 2005, for the four source sectors.





Figure 6. National and regional total emissions for CO for four different source sectors (industry, transportation, power, and residential) estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS between 2000 and 2008.



Figure 7. National and regional total emissions for SO_2 for four different source sectors (industry, transportation, power, and residential) estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS between 2000 and 2008.



Figure 8. National and regional total emissions for NO_x for four different source sectors (industry, transportation, power, and residential) estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS between 2000 and 2008.



Figure 9. National and regional total emissions for PM_{10} for four different source sectors (industry, transportation, power, and residential) estimated by REAS, EDGAR, MEIC, ZHAO, and GAINS between 2000 and 2008.



Figure 10a. Contribution of each of the vehicle categories to the number of gasoline vehicles



Figure 10b. Contribution of each of the vehicle categories to the number of diesel vehicles



Figure 11. National and regional on-road and off-road transport sector emissions of CO estimated by REAS, EDGAR, ZHAO, and GAINS between 2000 and 2008.



Figure 12. National and regional on-road and off-road transport sector emissions of NO_x estimated by REAS, EDGAR, ZHAO, and GAINS between 2000 and 2008.



Figure 13. National and regional on-road and off-road transport sector emissions of CO estimated by REAS, EDGAR, ZHAO, and GAINS between 2000 and 2008.



Figure 14a. Emissions of five pollutants (PM₁₀, CO, NMVOC, NO_x, and SO₂) in kg km⁻² month⁻¹ in 2008 January of the three emissions inventories.



Figure 14b. Emissions of five pollutants (PM₁₀, CO, NMVOC, NO_x, and SO₂) in kg km⁻² month⁻¹ in 2008 July of the three emissions inventories.



Figure 15a. Mixing ratios and concentrations of five pollutants in January using three emissions inventories.



Figure 15b. Mixing ratios and concentrations of five pollutants in July using three emissions inventories.



Figure 16a. Emissions of NO_x and VOCs as well as O_3 mixing ratio in each region in January using three emissions inventories.



Figure 16b. Emissions of NO_x and VOCs as well as O_3 mixing ratio in each region in July using three emissions inventories.

	Years	Source Sectors	Species	Horizontal Resolution	Coverage	Reference	
REAS	2000-2008	power plants, combustible and non- combustible sources in industry, on-road and off-road sources in transportation, residential, agricultural, and other anthropogenic sources	CO ₂ , SO ₂ , CO, PM ₁₀ , PM _{2.5} , BC, OC, NO _x , NH ₃ , NMVOC, CH ₄ , N ₂ O	0.25° x 0.25°	East, Southeast, South & Central Asia. Asian part of Russia	Kurokawa et al., 2013	
EDGAR	1970-2008	energy, industrial processes, product use, agriculture, large scale biomass burning, and other anthropogenic sources	CO_2 , SO_2 , CO , PM_{10} , NO_x , NH_3 , $NMVOC$, CH_4 , N_2O $HFCs$, SF_6 , NF_3	0.1° x 0.1°	Global	EC-JRC/PBL, 2011	
MEIC	2008, 2010	power, industry, transport ation , residential and agricultural sources	CO_2 , SO_2 , CO , PM_{10} , NO_x , $NMVOC$	0.1° x 0.1°	China	www.meicmodel.org	
ZHAO	2000-2014	power, combustible and non- combustible sources in industry, on-road and off-road sources in transporta tion and residential	CO ₂ , SO ₂ , CO, TSP, PM ₁₀ , PM _{2.5} , BC, OC, NO _x , Hg	N/A	China	Zhao et al., 2013b Zhao et al., 2015 Cui et al., 2015 Xia et al., 2016	
GAINS	1990-2030 (5-yr increment, projection starting in 2015)	energy, domestic, industrial combustion and processes, road and non-road transport ation and agriculture	CO ₂ , SO ₂ , CO, TSP, PM ₁₀ , PM _{2.5} , PM ₁ , BC, OC, NO _x , NH ₃ , VOC, CH ₄ , N ₂ O, F-gases	0.5° x 0.5°	Global	Amann et al., 2011 Klimont et al., in review Klimont et al., in preparation	

Table 1. Description of emissions inventories used for this study

Table 2. Source categorizations

	EDGAR	REAS	ZHAO	MEIC	GAINS		
	Manufacturing industries and construction Production of minerals Production of chemicals Production of metals Production of pulp/paper/food/drink Production of halocarbons and SF ₆	Iron and steel Chemical and petrochemica Combustible C	1		Iron and steel Pulp and Paper Combustible Non-ferrous metals Non-metallic minerals Other		
Industry	Refrigeration and air conditioning Foam blowing Fire extinguishers Aerosols F-gas as solvent Semicondutor/electronics manufacturing Electrical equipment Other F-gas use Solvent and other product use	Pig iron Crude steel Iron steel others Aluminum & Aluminu Copper Zinc Lead Cement Bricks Lime Coke orens Oil refinery Other transformation Sulphuric acid Others	Industry	Industry	Pig iron Coke ovens Agglomeration plants Steel Rolling mills Cast iron Non-ferrous metals Cement & Lime Sulfuric acid Aluminium Aluminium Glass production Fertilizer production Brick manufacturing Pulp and paper Refineries Others		
Transportation	Domestic aviation Road transportation Rail transportation Domestic navigation Other transportation Other transportation Fugitive emissions from solid fuels Fugitive emissions from oil and gas Public electricity and heat production Other energy industries Non-energy use of lubricants/waxes (CO ₂)	Cars Buses Light trucks Heavy trucks Motorcycles Other vehicles Domestic navigation Railway & etc. Power plants	Light duty vehicles Rural vehicles Small gasoline engines Heavy duty vehicles Motorcycles Machines Inland shipping Railway Power	Transportation	Cars Buses Light duty vehicles Heavy duty vehicles Motorcycles Domestic navigation Railway & etc. Power plants Diesel generators Briquette production Extraction and distribution		
Residential	Fossil fuel fires Residential and other sectors Waste incineration	Residential and other sectors	Residential and other sectors	Residential and other sectors	of liquid & gaseous fuels Cooking and heating Kerosene lighting Waste (trash) burning		

Table 3. Number of power plants in each region within China

Region	Number of coal power plants
East	250
North	206
Central	86
South	78
Northeast	76
Southwest	66
Northwest	43

Source: Carbon Monitoring for Action

 Table 4. Regional monthly mean concentrations of MEIC, REAS, and EDGAR and largest differences found within a region in WRF-Chem simulation in 2008

(a) January																				
Regions	PM ₁₀ (μg/m ³)				O ₃ (ppbv)			SO ₂ (ppbv)			NO ₂ (ppbv)				CO (ppbv)					
	MEIC	REAS	EDGAR	diff	MEIC	REAS	EDGAR	diff	MEIC	REAS	EDGAR	diff	MEIC	REAS	EDGAR	diff	MEIC	REAS	EDGAR	diff
Central	163	155	96	67	31	41	46	15	26	23	21	5.3	15	12	6.9	8.1	852	632	382	470
East	123	129	82	48	32	39	43	11	15	16	18	3.1	15	13	8.0	7.2	623	598	329	294
North	27	27	19	8.5	45	46	47	2.0	8.1	7.1	9.3	2.2	5.6	5.1	3.9	1.7	255	214	147	108
Northeast	29	25	14	15	41	41	46	4.5	4.2	5.1	5.9	1.7	6.6	6.5	3.4	3.3	259	242	165	94
Northwest	19	19	14	4.9	55	56	56	1.0	3.9	3.4	4.0	0.59	1.5	1.4	1.2	0.3	166	154	119	47
South	127	128	82	46	40	47	44	6.8	6.3	7.7	8.6	2.3	4.5	4.0	3.4	1.1	534	548	321	228
Southwest	42	37	23	19	59	60	59	1.2	5.1	4.9	3.4	1.7	1.5	1.4	1.3	0.13	284	242	156	128

 $\mathrm{PM}_{10}~(\mu\mathrm{g/m^3})$ $O_3 (ppbv)$ SO_2 (ppbv) NO2 (ppbv) CO (ppbv) Regions MEIC REAS EDGAR diff 72 55 7.3 7.1 4.0 2.7 224 77 Central 64 37 34 62 56 7.0 0.27 5.4 6.7 263 300 6.9 28 55 63 55 2.5 7.8 5.0 3.7 247 321 192 129 East 56 64 36 8.1 6.9 8.4 9.4 8.7 39 33 21 13 58 63 8.5 4.2 4.5 5.6 1.3 2.6 3.0 2.0 1.0 178 212 130 82 North 54 3.1 39 33 21 12 51 55 47 8.2 1.5 2.2 3.3 1.8 2.7 1.9 1.2 172 199 153 Northeast 46 Northwest 8.4 8.8 6.3 2.5 55 58 53 4.8 1.4 1.3 1.6 0.22 0.57 0.78 0.61 0.21 94 95 90 5.3 19 17 39 44 40 2.4 5.2 2.9 2.9 4.0 170 185 156 South 21 3.8 5.0 4.3 3.1 1.1 29 50 Southwest 11 12 7.9 4.1 50 53 3.6 1.8 2.3 1.5 0.75 0.93 1.3 1.1 0.34 116 125 104 21

(b) July