LETTER

National- to port-level inventories of shipping emissions in China

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Abstract

Shipping in China plays a global role, and has led worldwide maritime transportation for the last decade. However, without taking national or local port boundaries into account, it is impossible to determine the responsibility that each local authority has on emission controls, nor compare them with land-based emissions to determine the priority for controlling these emissions. In this study, we provide national- to port-level inventories for China. The results show that in 2013, the total emissions of CO, non-methane volatile organic compounds (NMVOCs), nitrogen oxides (NOₓ), particulate matter (PM), SO₂ and CO₂ were 0.0741 ± 0.0004 Tg·yr⁻¹, 0.0691 ± 0.0004 Tg·yr⁻¹, 1.91 ± 0.01 Tg·yr⁻¹, 0.164 ± 0.001 Tg·yr⁻¹, 1.30 ± 0.01 Tg·yr⁻¹ and 86.3 ± 0.3 Tg·yr⁻¹ in China, respectively. By providing high-resolution spatial distribution maps of these emissions, we identify three hotspots, centered on the Bohai Rim Area, the Yangtze River Delta and Pearl River Delta. These three hotspots account for 8% of the ocean area evaluated in this study, but contribute around 37% of total shipping emissions. Compared with on-road mobile source emissions, NOₓ and PM emissions from ships are equivalent to about 34% and 29% of the total mobile vehicle emissions in China. Moreover, this study provides detailed emission inventories for 24 ports in the country, which also greatly contributes to our understanding of global shipping emissions, given that eight of these ports rank within the top twenty of the port league table. Several ports in China suffer emissions 12–147 times higher than those at Los Angeles port. The ports of Ningbo-Zhou Shan, Shanghai, Hong Kong and Dalian dominate the port-level inventories, with individual emissions accounting for 28%–31%, 10%–14%, 10%–12% and 8%–14% of total emissions, respectively.

1. Introduction

With the rapid increase in the global economy, seaborne transport has become widely acknowledged as the most important mode of international trade. Meanwhile, emissions from seaborne transport contribute significantly to global air pollution (Paxian et al 2010, Viana et al 2014, Eyring et al 2010). Some studies indicate that shipping emissions contribute to substantial increases in ambient concentrations of air pollutants over vast land and sea areas, and are responsible for increases in premature deaths related to cardiopulmonary diseases and lung cancer in these areas (Corbett et al 2007, Matthias et al 2010). This is because ships operate much closer to where people live and work than previously recognized (Fan et al 2016). Typically, about 70% of emissions from international shipping occur within 400 km of the coastline (Endresen et al 2003).

To quantify the maritime contribution to air pollution, three scales of emission inventories, on global-, regional- and port-scales, have been compiled for various locations. For the global inventories of shipping emissions, the International Maritime Organization (IMO) compiled the global shipping
inventory (IMO 2015), estimating that total shipping emissions for 2012 were approximately 938 million tons of CO₂ or 961 million tons of CO₂e (including greenhouse gases: CO₂, CH₄ and N₂O). Corbett et al (2007) generated a global shipping emission inventory for 2002. They found that most shipping-related mortality effects occurred within Asia and Europe, where dense populations and high shipping-related particulate matter (PM) concentrations coincide. Such studies can be used to assess the impact of oceangoing ships on the atmosphere and climate. On a regional scale, Van Aardenne et al (2013) compared present-day emissions from various high intensity locations of the shipping emission inventory using different methodologies and data. This study suggests that emissions of NOₓ from international maritime transport within European waters are likely to increase and could equal land-based sources by 2020. The United States Environmental Protection Agency (USEPA 2009) calculated a 2002 base year inventory using a ‘bottom-up’ approach, reflecting data for vessel calls, emission factors and port activity. Based on growth rates, the USEPA also created 2020 and 2030 baselines, as well as control inventories. These regional emission inventories provide the most comprehensive information on activity and are often used as basic data for emission control area (ECA) proposals. Our recent study (Liu et al 2016) uses combined terrestrial and satellite AIS data to analyze marine engine performance and apply an improved activity-based shipping emission method with two nested domains to calculate shipping emissions. At port level, many European and North American ports, including Los Angeles (Starcrest Consulting Group 2014), Vancouver (Merk 2014) and Portland (Browning and Bailey 2006), have their own emission inventories, which have been compiled over recent years. Some of these ports have adopted shipping-specific emission-reduction technology based on their shipping emission inventories, such as shore power and vessel speed reduction (Feng et al 2014).

Shipping in China plays a global role, and has led worldwide maritime transportation for the last decade. World cargo information for loaded and unloaded volumes (China Port Press 2014) shows that six of the top ten largest ports are located within China. This expansion in shipping transport has contributed significantly to its economic growth, but has also led to increased air pollution from oceangoing ships, which are still predominantly powered by diesel or high-sulfur residual (or bunker) fuel (Fan et al 2014, Song 2014, Yang et al 2007). As we previously indicated, such shipping emissions can have a large impact on human health, causing more than 24,000 premature deaths annually in East Asia (Liu et al 2016). With the rapid growth of ship activity and seaborne trade, more advanced technologies, such as selective catalytic reduction, shore power and scrubbers, are clearly needed to reduce shipping emissions in coastal ports, and along the major waterways of China (Feng et al 2014). To control shipping emissions, China has implemented, ‘Limits and measurement methods for exhaust pollutants from marine compression ignition engines (China I, II)’ (MEP 2015), which requires all new ship engines sold within China to meet the China I standard as of 1 January 2018, and the China II standard as of 1 January 2021. During the transition from China I to China II, the reduction in PM emissions is expected to be around 40% to 60%, revealing that China has made a huge effort to control vessel emissions within a short time frame.

However, the studies on shipping emissions in Asia are only limited to a few individual ports or regional studies. Zhang et al (2017). National-level or port cluster shipping emissions have not been reported until now. Our previous study of shipping emissions in East Asia (Liu et al 2016) mainly focuses on densely populated areas, without taking national or local port boundaries into account. Such limitations make it more difficult for policy makers to evaluate and control shipping emissions in China.

In this study, we build upon our previous work, providing national- to port-level inventories for China. Here, we estimate the emissions of all kinds of oceangoing vessels and coastal vessels within Chinese coastal waters, excluding inland vessels that only sail on rivers, lakes and channels. We compiled a national shipping emission inventory for China as well as high-resolution spatial distribution maps of these emissions. Based on these emission maps, we identified three hotspots with high emission intensities. The emission intensities were calculated for these three hotspot regions, as well as for individual ports within these regions to compare them with previous studies. Our intention was to better understand shipping emissions in China, and to identify hotspot regions with high-intensity emissions that should be the focus of emission-reduction plans.

2. Material and methods

2.1. Study area
China has more than 18,000 km of coastline, along which are located more than 50 ports (Chinese government website 2017). Previous shipping emission inventories in other countries (Entec UK Limited 2010, USEPA 2015) have generally included emissions ranging up to 200 nautical miles from the coastline, encompassing the so-called exclusive economic zone (EEZ). Because of Asia’s complex geographical setting, the boundary line for emission calculation in this study is a nine-dash line in the South China Sea, but an EEZ line in the East China Sea. In the Yellow Sea, the boundary is the median line between the Chinese and Korean coastlines (see figure 1).

2.2. Emission calculation
In this study, we used vessels’ automatic identification system (AIS) data to determine their activity
profiles, which included detailed modeling of port emissions. Based on previous port emission inventories for the US (Starcrest Consulting Group 2012, Starcrest Consulting Group 2013, Starcrest Consulting Group 2014), four operating modes (at sea, maneuvering, at anchor and at berth; supplementary table 1 available at stacks.iop.org/ERL/12/114024/mmedia) as well as ten different ship types (supplementary table 2) and three different engine types (main engine, auxiliary engine and auxiliary boiler) were evaluated in this study.

The activity-based methodology and AIS data were applied to all marine vessel emissions in this study. Equation (1) gives the total emissions for shipping-related activities:

\[ E_{\text{total}} = E_{\text{ME}} + E_{\text{AE}} + E_{\text{boiler}} \]  

where \( E_{\text{total}} \), \( E_{\text{ME}} \), \( E_{\text{AE}} \) and \( E_{\text{boiler}} \) are the emissions in g for the total engines, main engine (ME), auxiliary engine (AE) and auxiliary boiler (AB), respectively.

Equation (2) was used to calculate the main engine emission:

\[ E = P_{\text{MCR}} \times EF \times LLA \times T \]  

where \( E \) (g) is the engine emission, \( P_{\text{MCR}} \) (kW) is the maximum continuous rated power of the main engine; \( LF \) is the engine load and \( EF \) (g kWh\(^{-1}\)) represents the base emission factor of the propulsion engine given a fuel with a specific sulfur content, as described in supplementary table 3; \( LLA \) is the low load adjustment for the emission factor, when \( LF \) is lower than 20% (supplementary table 4); and \( T \) (h) is the duration of operation on the vessel trip. \( LF \) is calculated using equation (3), as follows:

\[ LF = \left( \frac{V_{\text{actual}}}{V_{\text{max}}} \right)^3 \]  

where \( V_{\text{actual}} \) (knot) is the actual speed and \( V_{\text{max}} \) (knots) is the maximum speed.

Figure 1. Spatial distribution of particulate matter (PM) emissions from maritime transportation in China. Map (a) shows sea routes within Chinese waters, while maps (b), (c) and (d) are detailed maps of hotspot regions centered on the Bohai Rim Area (BRA), Yangtze River Delta (YRD) and Pearl River Delta (PRD) respectively.
Equation (4) is used to calculate both auxiliary engine and boiler emissions:

\[ E = EF \times \sum_i \sum_j P_{i,j} \times T_i \]  

(4)

where \( E \) (g) is the engine emission, \( i \) is the vessel operating mode (at sea, maneuvering, at anchor and at berth), \( j \) is the vessel type (supplementary table 2), \( P \) (kW) is the actual engine power for each ship type and operation mode (supplementary tables 5 and 6), \( EF \) (g kWh\(^{-1}\)) is the emission factor (supplementary tables 7 and 8), and \( T \) (h) is the operation time for each mode.

2.3. Data sources

The AIS data is mostly used to exchange information between the vessels and AIS base stations (Last et al. 2015). The IMO requires an AIS device to be installed aboard international voyaging ships greater than 300 gross tons (GT) and passenger ships of all sizes (Fan et al. 2016). The AIS system provides detailed vessel activity information, including its maritime mobile service identity (MMSI), longitude, latitude, actual speed and the time the message was received by the receiver station (Jalkanen et al. 2016). In this study, we used AIS data from January to December 2013, comprising a combination of terrestrial and satellite AIS data available from the Marine Safety Administration of China (MSA).

The static information was obtained from Lloyd’s Register and the China Classification Society (CCS). Ships heavier than 500 GT are usually listed by their IMO numbers in Lloyd’s Register. The CCS provides supplementary information for ships smaller than 500 GT, travelling along China’s coastline. The ship properties used for our static information database include their MMSI code, vessel type, rated engine speed, rated engine power, length, width, height, designed maximum speed, dead weight tonnage, maximum draught and build year. Actual power was calculated quite simply, using the Propeller Law, while data for the auxiliary engine and boiler power was generally not available. Based on previous studies, the auxiliary engine and boiler power for various ship classes and modes are given in supplementary tables 5 and 6.

The fuel type and specifications are critical in determining marine vessel emission factors. Marine vessels often have four types of fuel, including heavy fuel oil (HFO), marine diesel oil (MDO), marine gas oil (MGO) and liquefied natural gas (LNG). In this study, the fuel type was determined from the engine type, e.g. LNG Otto-cycle engines only use LNG. We assumed that harbor service vessels, such as work vessels, tugs, crew boats, pilot vessels, government vessels and fireboats used MDO/MGO as their fuel, given their special work requirements and the limited space they have onboard to place a boiler, while other vessels generally use HFO. For HFO and MDO/MGO fuels, the average sulfur content was 2.43% and 0.13% in 2013, as reported by the IMO Maritime Environment Protection Committee (IMO 2015).

Because of the lack of local emission factors for China, the baseline emission factors used in this study were adopted from previous ones (IMO 2015, Starcrest Consulting Group 2012). Additional fuel correction factors for NO\(_x\), SO\(_2\) and PM were applied for various sulfur content. When the main engines operate below 20% power, emission factors tend to increase as the load decreases. All emission factor adjustments used in this study are listed in supplementary table 4.

Data processing is an important procedure for the correct use of AIS data (Traut et al. 2013). Firstly, all of the data, which has the same maritime mobile service identity (MMSI), is put together based on its time sequence. Secondly, we check the validation of the vessel speed and geo-location data. Data which is identified as faulty will be deleted. For instance, if the vessel speed is above 100 knots, the speed data will be removed. Then we will check for duplicated messages of the AIS data. Because AIS data is obtained from combined terrestrial and satellite receivers, some duplication may appear. We designed the software so that if the terrestrial or satellite AIS receivers obtained the same AIS message, one of them would be deleted automatically. However, bad weather conditions, complicated geographical locations and many other unpredictable reasons may cause a time delay and lead to duplicate messages that are not identical. When each ship’s AIS data is ranked according to the time sequence, the time interval of the duplicate message is 0 in most cases. Therefore, the duplicate messages would have no effect on our results. Even if some of the duplicate messages have a different time, then the total time between distinct messages is divided into two time intervals, and the results are the same.

Supplementary figure 1 provides both the percentages of the number and the total duration time by the time intervals of our AIS data. It is clear that 80% of the AIS data has a time interval of less than 60 seconds. Only 0.44% of our AIS data has a time interval of more than 40 min. However, the time fraction of large periods (>130 h) without any messages contributes 38.21% of the total time. These large periods do not necessarily mean the data quality is poor. They may have been caused by bad weather distraction, the distance between the vessel and the land-based station being too far, complicated geographical conditions interfering with some of the messages, or other uncertain reasons. Our research targets a region instead of global cruising. Thus, if the ship cruises out of the region and then comes back into it, a long time interval will result because the AIS location was excluded from this study. For these long interval messages (two points that have an interval longer than 10 min), the speeds between each of the two AIS data points were generated by interpolation for every 10 min along the voyage trajectory. Emissions between each pair of adjacent speed points were calculated for each single vessel. We have
developed a two nested domain method to reduce the impact of these long time interval messages. If the ship goes out of the first domain and comes back into it, all emissions during that time will be assigned to the boundary of the first domain. Then, all the emissions along the boundary, including those generated by the long-duration messages are cut to get the emissions in the second domain.

The coverage of the ship fleet by AIS data is good in China. We assume that the ships registered in this region multiplied by the percentage in service are considered to be the ships in service inside our research domain. The coverage of the ship fleet by AIS data is the number of ships observed on the AIS divided by the number of ships in service (IMO 2015). The ship number registered in this region comes from the Lloyd’s Register database and the China Classification Society (CCS), the ratio of the in-service percentage is referenced from the third IMO GHG study (IMO 2015), the number of ships observed on the AIS is from our AIS database based on the MMSI codes of each ship. supplementary figure 2 shows the coverage ratios of the ship fleet by the AIS data from our study and the IMO study. It is clear that the coverage ratios of transport ships and non-transport ships from our study are 81% and 61%, which are higher than those from the IMO study. There are several reasons for the apparently incomplete coverage. For instance, the amount of ships classified as in service was not actually so. In addition, because the data is limited to the Chinese ocean region, the in-service ships may actually voyage in other regions. According to the above factors, the AIS data provides good coverage ratios in China.

3. Results and discussion

3.1. National shipping emission inventory

The shipping emissions for CO, NMVOC, NOx, PM, SO2 and CO2 in China in 2013 were 0.0741 ± 0.0004 Tg yr⁻¹, 0.0691 ± 0.0004 Tg yr⁻¹, 1.91 ± 0.01 Tg yr⁻¹, 0.164 ± 0.001 Tg yr⁻¹, 1.30 ± 0.01 Tg yr⁻¹ and 86.3 ± 0.3 Tg yr⁻¹, respectively. Compared with global international shipping emissions for 2012 from the IMO report (2015), Chinese shipping emissions account for approximately 9%, 11%, 11%, 12%, 13% and 11% of the global emissions of CO, NMVOC, NOx, PM, SO2 and CO2, respectively.

Using the AIS-based method, the spatial distribution of shipping emissions was generated by adding the emissions in each grid box. Because of their similar spatial distribution, we use the PM as an example to illustrate their general characteristics (spatial distribution for CO, NMVOC, NOx and CO2 is shown in supplementary figure 3. Figure 1(a) shows the spatial distribution of shipping-related PM emissions at a resolution of 1 km × 1 km. It is clearly apparent that high-density traffic occurs on major waterways and within major ports along the coastline. Figures 1(i)–(d) further provide a detailed emission distribution in three high-density regions.

Figure 2(a) shows the percentage of emissions for each pollutant for various vessel types. Clearly, the bulk and container vessels had the highest percentage of overall emissions, contributing 29%–31% and 25%–28% to the total, respectively. Contribution rates from tankers, general cargo vessels and roll-on roll-off cargo vessels were 19%–21%, 7%–8% and 6%–8%, respectively. These results are consistent with our previous
Figure 3. Emission intensities for various pollutants with the three hotspots: Bohai Rim Area (BRA), Yangtze River Delta (YRD) and Pearl River Delta (PRD). Sea areas of China, BRA, YRD and PRD are $2997 \times 10^4$, $104 \times 10^4$, $97 \times 10^4$ and $41 \times 10^4$ km$^2$ respectively. PM = particulate matter; NMVOC = non-methane volatile organic compounds.

3.2. Regional shipping emission inventory

Figures 1(b)–(d) show detailed maps of three hotspots with high emission intensities, defined by the Bohai Rim Area (BRA), Yangtze River Delta (YRD), and Pearl River Delta (PRD). The intensities of PM emissions in these three regions are distinct, because of their particular geographical setting. Specifically, there are many large ports within the BRA (figure 1(b)), belonging to the Tianjin, Hebei, Liaoning and Shandong provinces, and spanning the Bohai coastline. Thus, this hotspot is related to high intensity emissions along major shipping routes to large ports of this region, such as those of Tianjin and Dalian. The YRD forms at the junction of the Yangtze River and the East China Sea. High shipping emissions in this region are centered on the ports of Zhou Shan, Ningbo and Shanghai. These three ports are large transfer stations for goods being transported to north and south China, or along the Yangtze River (figure 1(c)). As an important junction between maritime and land transportation, as well as south to north maritime routes in China, the PRD has the most intensive cluster of ports (figure 1(d)).

The total emissions and intensities for the three hotspots were calculated and compared (figure 3). Overall, ships contributed about 37% of the total emissions in these three regions, and these regions only account for 8% of the total area of Chinese ocean. Of the total shipping emissions in China, about 20%, 9%, and 8% were from the YRD, PRD and BRA, reflecting the high traffic to and from ports in these three regions. Although the ocean area of the PRD hotspot is the smallest, its emission intensity is one of the highest. Taking NO$_x$ as an example, its emission intensity in the PRD is 1.05, 2.79 and 6.18 times higher than the YRD, BRA and Chinese average values, respectively. There are many busy ports in the PRD, including ports of Hong Kong, Shenzhen and Guangzhou. These high-density ports with their heavy maritime activity result in high emission intensity in this area.

China has established three domestic emission control areas (DECAs) on the PRD, YRD and BRA, which have a similar area to the three hotspots identified in our study. Thus, the strategy to start from these three regions is reasonable, and will target 37% of total shipping emission in the Chinese ocean. However, 37% is still far from enough. With more understanding on shipping emissions and their impact, effort should be made for all coastal regions to achieve the best control effects.

3.3. Shipping emission inventory for individual ports

To identify the main contributors within the three hotspots, the individual port emission inventories
Figure 4. Individual port shipping emissions within the three hotspot regions: Bohai Rim Area (BRA), Yangtze River Delta (YRD) and Pearl River Delta (PRD).

1, 2, 3, 4, 5, 6 on X-axis represent CO, NMVOC, NOx×10^3, PM, SO2×10^3, CO2×10^6 respectively; Y-axis is the shipping emission in terms of tonne.

This study  Fu et al 2012  Ng et al 2013  Yang et al 2015  Tan et al 2014

were calculated. Given the lack of official local port information within China, we defined the boundary for individual ports based on information from the local maritime safety administration (MSA) and the catalogue of port and fairway charts of China (supplementary figure 4). Figure 4 shows the shipping emissions for individual ports within the three hotspots. The ports of Ningbo-Zhou Shan, Shanghai, Hong Kong and Dalian dominated the port emission charts, accounting for 28%–31%, 10%–14%, 10%–12% and 8%–14% of total shipping emissions within the hotspot regions. According to the port yearbook (China Port Press 2014), the port cargo throughput of Ningbo-Zhou Shan, Shanghai, Dalian and Hong Kong was 8.1×10^8 tons, 7.7×10^8 tons, 4.1×10^8 tons and 2.8×10^8 tons in 2013, ranking them as no. 1, no. 2, no. 7 and no. 10 of Chinese coastal port cargo throughput, respectively. The ranks for port emissions and port goods throughput do not exactly match, except for nos. 1 and 2 ports. Their cargo throughput is significantly larger than the others and so are the emissions. The reasons for the difference in the rank of emissions versus cargo throughput are many. Firstly, the domain for the emission inventory calculation is based on the administration zone of each local MSA. The area of these zones is not defined by cargo throughput, but mainly based on the geometric and political boundaries. Secondly, there are many ships which voyage across the port without berthing, contributing pollutants to that port. For instance, ships which voyage to Tianjin port go through the Dalian port area. Thirdly, the cargo handling capacity of the port is also an important factor, and will affect ship berthing time and therefore shipping emissions. In addition, marine engine technologies (divided into tier 0, tier 1 and tier 2 based on the year built) also have an impact on NOx emission. Supplementary figure 5 presents the marine engine technology distribution in Chinese ports. Tier 0 engines dominate the ship fleets, accounting for 60%–76% of all engines. In the meantime, fewer than 3% ships have tier 2 engines installed. Between all the ports, the difference in vessel technology is very small, which means that engine technology is not the main reason for emission variation. In our study, fuel
quality remains the same for different ports as no special requirements were implemented in 2013.

We also presented a comparison between the ports in China with that of Los Angeles. Los Angeles port is the largest container port in the United States, and is ranked 19th in the top twenty of the port league table (China Port Press 2014). Comparing the shipping emissions of Los Angeles port (Starcest Consulting Group 2014) and the Chinese ports in this study, we found that NO\(_x\) and PM emissions from the Ningbo-Zhou Shan port, Shanghai port, Hong Kong port and Dalian port individually are 12–39 and 42–147 times higher than those from Los Angeles. There are many reasons for the higher PM emission in Chinese ports. Firstly, the cargo throughput of Ningbo-Zhou Shan port, Shanghai port, Hong Kong port and Dalian port is 2.5–8.0 times that of Los Angeles (US Maritime Administration website 2017). More vessel activity causes the greater pollution in Chinese ports. Secondly, Los Angeles port is located in the North America emission control area (ECA), requiring vessels to use marine diesel oil (MDO) and after-treatment to control SO\(_2\) and PM emissions. At the same time, their vessels generally use heavy fuel oil (HFO) when they travel through Chinese ocean and port areas. The PM emission factor of HFO is about 6.7 times higher than that of MDO (see supplementary tables 3 and 7). In addition, the port areas of Ningbo-Zhou Shan, Shanghai, Hong Kong and Dalian in our study are approximately 2.5, 1.1, 0.12 and 2.6 times larger than the area of Los Angeles, which indicates that the size of the research domain has a great effect on PM emission. It is noteworthy that there are many ships which pass through the port water without berthing, contributing lots of PM emissions to it. That is why even the Hong Kong port, which has a much smaller area than Los Angeles, still has higher PM emissions. We also compared the PM emission intensity between the Shanghai port and Los Angeles, and found that the PM emission intensity of Shanghai is more than 20 times higher than that of Los Angeles. Zhao et al (2013) showed the PM\(_{2.5}\) ambient concentration at Shanghai port is 62.6 \(\mu g\cdot m^{-3}\), while Minguillón et al (2008) showed the PM\(_{2.5}\) ambient concentration is about 14 \(\mu g\cdot m^{-3}\) at Los Angeles. Source apportionment results from the two studies (Zhao et al 2013, Minguillón et al 2008) also indicate that the primary PM\(_{2.5}\) contribution from ship traffic at Shanghai port is 2–20 times higher than the results from the study of Los Angeles port.

Figure 4 also compares our collated data with all available shipping emission studies, which includes emissions for the ports of Shanghai (Fu et al 2012), Hong Kong (Ng et al 2013), Shenzhen (Yang et al 2015) and Dalian (Tan et al 2014). These studies also adopted a ’bottom-up’ approach to calculate shipping emissions. Our results are very close to previous port emission estimates, with only slight differences caused by variable shipping activity in different years as well as differences in the boundary lines. In this study, 24 port-level emission inventories were compiled for the first time using unified methods and data, providing important data to estimate global port emissions, especially as eight of these ports are ranked within the top twenty of the port league table for 2013 based on throughput.

In China DECAs, the sulfur content of any fuel oil used on board vessels berthing at major ports, including the ports of Guangzhou, Shenzhen, Zhuhai, Nantong, Ningbo-Zhou Shan, Shanghai, Suzhou, Tianjin, Qinhuangdao, Tangshan and Gangzhou, should not exceed 0.5% m\(^{-1}\) from 1 January 2017. This restriction will be extended to berthing at all ports within DECAs from 1 January 2018. After 1 January 2019, the sulfur content shall not exceed 0.5% m\(^{-1}\) when vessels enter DECAs. From this study, the emissions from these target ports located on our hotspot areas (which are included in 2017 measures) are about 19% and 17% of total shipping SO\(_2\) and PM emissions in China, respectively. Moreover, the port-level emission inventory should be used to evaluate shipping emission reduction within DECAs based on the port calculation one-by-one.

3.4. Comparison with national on-road mobile emissions

Figure 5 is the comparison between shipping emissions in our study and national vehicle emissions reported by the Ministry of Environmental Protection (MEP 2014). Ships generally use diesel engines, which produce large amounts of NO\(_x\) and PM. In this study, NO\(_x\) and PM from shipping emissions were found to be equivalent to about 34% and 29% of all vehicle (gasoline and diesel) emissions in China, respectively. The ratio of NO\(_x\) from shipping emissions, diesel vehicles (on land) and gasoline vehicles is about 1: 2.2: 0.85.

To control shipping emissions, China implemented ’Limits and measurement methods for exhaust pollutants from marine compression ignition engines (China I, II)’ (MEP 2015), which requires all new ship engines sold within China to meet the China I standard as of 1 January 2018, and the China II standard as of 1 January 2021. During the transition from China I to China II, the reduction in PM emissions is expected to be around 40% to 60%, revealing that China has made a huge effort to control vessel emissions within a short time frame. Reviewing the process on diesel vehicle emission control, the PM emission factor from the engine was reduced by about 92% from China I to China V (MEP 2001, MEP 2005). There is still potential to further reduce emissions from ships, by adding after-treatment devices and improving engine technologies.

Currently, the fuel sulfur limits for key ports in DECAs were set to 0.5% in 2017–2019. The requirements are only for berth in 2017–2018, which means only for auxiliary engines. Except for long-distance oceanoing vessels, the auxiliary engines usually use MDO/MGO instead of HFO. Compared with on-road diesel limits (10 ppm on sulfur) and off-road
diesel limits (350 ppm on sulfur), the marine fuel sulfur content is 14 times higher than off-road diesel, and 500 times higher than on-road diesel. On January 1, 2015, the European Union (EU) stipulated that the sulfur content in marine fuel must be lower than 0.1% for ships entering the Baltic and North Sea ECA (Sys et al. 2016). To further control SO\(_2\) emissions from shipping, China should follow the EU’s step and continuously reduce the sulfur content limits of marine diesel on DECA.

3.5. Uncertainly analysis
In this study, uncertainty in national shipping emission estimates is mainly derived from uncertainty in the input parameters for individual ships, including operational parameters, ship specifications and emission factors. Among all the operational parameters, instantaneous speed is the most important contributor to uncertainty. This is because the main engine power is approximated as the cube of its speed. Thus, small deviations in AIS speed are magnified into larger variations in power. From the IMO report (2015), the standard deviation in uncertainty of AIS speeds is 11%, with respect to the mean speed. Ship specifications were obtained from a combined Ship Specification Database, including information from the Lloyd’s Register and the China Classification Society. Together, these sources provide relatively complete ship specification information. The uncertainty in emission factors also needs to be taken into consideration. Based on the IMO report (2015), the uncertainty in the average CO\(_2\) emission factor is small, and set as ± 1%. The uncertainty in emission factors for the other pollutants is less well defined. A recent study by Beecken et al. (2015) reports a measurement uncertainty of 21% for SO\(_2\) and around 25% for NO\(_x\). Considering the shipping fleet average, the uncertainty for the other compounds is probably higher than CO\(_2\); it was set to ± 10% in this study. We used a Monte Carlo method with 100,000 simulations to evaluate the uncertainty in the calculation of our bottom-up shipping emission inventories.

4. Conclusions
In this study, we used AIS shipping data integrated into an activity-based shipping emission model, to calculate shipping emissions from a national to regional scale, as well as for individual ports in China. The results show that national shipping emissions of CO, NMVOC, NO\(_x\), PM, SO\(_2\) and CO\(_2\) in China in 2013 were 0.0741 ± 0.0004 Tg yr\(^{-1}\), 0.0691 ± 0.0004 Tg yr\(^{-1}\), 1.91 ± 0.01 Tg yr\(^{-1}\), 0.164 ± 0.001 Tg yr\(^{-1}\), 1.30 ± 0.01 Tg yr\(^{-1}\) and 86.3 ± 0.3 Tg yr\(^{-1}\), respectively. Bulk carrier and container ships were the largest contributors to shipping emissions, accounting for 29%–31% and 25%–28% of total shipping emissions, respectively. From a spatial analysis of shipping emissions, we identified three hotspots with high emission intensities. These include the regions of the Bohai Rim Area (BRA), Yangtze River Delta (YRD), and Pearl River Delta (PRD). Although these three hotspots accounted for only 8% of the ocean area, they contributed around 37% of total shipping emissions. Of the 24 ports for which emissions were calculated, the Ports of Ningbo-Zhou Shan, Shanghai, Hong Kong and Dalian dominated the emission charts, accounting for 28%–31%, 10%–14%, 10%–12% and 8%–14% of the shipping emissions from the three hotspot regions, respectively. NO\(_x\) and PM emissions from

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**Figure 5.** Comparison of shipping emissions and vehicle emissions for China (MEP 2014). PM = particulate matter; NMVOC = non-methane volatile organic compounds. The units of emissions for ship and vehicle are one million tons.
Ningbo-Zhao Shan, Shanghai, Hong Kong and Dalian individually are 12–39 and 42–147 times higher than those from Los Angeles port. Previous studies also indicate that the primary PM\(_{2.5}\) contribution from ship traffic at the Shanghai port is 2–20 times higher than the results from a study of Los Angeles, which can support our results to some extent. NO\(_x\) and PM from shipping emissions were found to be equivalent to about 34% and 29% of all vehicle (gasoline and diesel) emissions in China, respectively. Ongoing regulation of shipping emissions within these regions over the next few years will likely have immediate and marked effects on shipping-related pollution.

This study provides a point of reference to evaluate future efforts for shipping emission controls. The same method could be used to reflect the emission variation for each port in the future. Using the same research domain and method, the real change from emission controls can be evaluated. In addition, the emission inventory also serves as an input for atmospheric transport models to judge and predict the air quality impact from maritime emissions. Currently, concentrated evaluation is already being attempted on a regional level (Liu et al. 2016), and the detailed port-level emission inventories will benefit urban level source apportionment or forecasting. Nowadays, the Chinese government is pushing studies and adopting various measures to prevent air pollution (Liu and He 2016). With the increase of seaborne trade, more attention should be paid to shipping emissions for air quality protection in the port regions.

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