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Research article

# Virtual flows of aquatic heavy metal emissions and associated risk in China

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ABSTRACT

Heavy metal pollution is posing a serious threat to ecosystem and human health in China. In addition to being emitted into the atmosphere, heavy metals generated by industrial processes are also emitted into water bodies. However, there is a lack of research exploring trade-induced aquatic heavy metals (AHM) emissions hidden in cross-regional supply chain networks. Such information can provide both consumer and producer perspectives on stakeholders' responsibility and involve them in pollution control along the entire supply chain including influencing consumption choices. Using a bottom-up AHM emission inventory (including mercury (Hg), cadmium (Cd), chromium (Cr), arsenic (As), and lead (Pb)) in 2010, we firstly accounted for production- and consumption-based AHM emissions and their virtual flows between China's 30 provinces. Additionally, we developed an integrated index, i.e. Equal Risk Pollution Load, to measure the risk associated with five AHM based on the corresponding reference dose. We found that richer provinces Guangdong, Jiangsu and Zhejiang through their consumption of metal products caused aquatic Hg, Cd, As and Pb pollution in provinces with nonferrous-metallic mineral resources such as Hunan, Yunnan, and Inner Mongolia. However, virtual aquatic Cr emissions were incurred in richer coastal regions (e.g. Guangdong, Zhejiang) for producing and exporting high value added products (electroplated products, printed circuit board and leather products) to less developed inland provinces. Finally, we propose measures from a supply chain perspective to mitigate aquatic pollution.

## 1. Introduction

The issue of heavy metal pollution has long been a global environmental concern (Lin et al., 2012; Sekhar et al., 2004). After entering into the environment, heavy metals are reported to be associated with a variety of toxicity, including developmental, immunological disorders, mutagenesis and carcinogenesis (Nriagu, 1979; Nriagu and Pacyna, 1988; Williams et al., 2009; Yamaguchi et al., 2014). United States Environmental Protection Agency (USEPA) recommended the toxicity values for health effects resulting from chronic exposure to heavy metals in the Integrated Risk Information System (IRIS) and called for the attentions from both academics and policy makers on this issue (USEPA, 2017). As an emission hot spot of the world (Lin et al., 2012; Pacyna et al., 2010; Shetty et al., 2008), China experienced a peak period of pollution events related to anthropogenic heavy metal emissions after 2009, such as Hunan Liuyang Cd incident in 2009, Guangxi Cd spill in 2012; limit-violating Pb levels in the blood of children in Fengxiang, As pollution in Linyi, Shandong (China Youth Daily, 2012). Most incidents are related to the heavy metal emissions from industrial wastewater. According to an official report in 2014, about 20% of arable land had been polluted by heavy metals, which caused a huge farming crisis (MEP, 2014; Wang et al., 2016). For comprehensively controlling heavy metal pollution, the State Council of China released the National Action Plan for Soil Pollution Control (State Council of China, 2016) in 2016. One of the most important measures is to control heavy metal emissions generated by industrial processes (Fan

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and Hong, 2013).

Most heavy metal emissions are from industrial sectors and processes including the chemical industry, printing and dyeing, mining, fossil fuels burning, metal smelting, and sewage irrigation (Sen and Peuckerehrenbrink, 2012; Vitousek et al., 1997). Hunan and Yunnan are the provinces with the highest amount of heavy metal emissions (Wu et al., 2015). A straightforward way to control heavy metal emissions is to upgrade production technology and improve end-of-pipe controls in these emission-dominated industries and provinces (Chen et al., 2018), which is reflected in China's strategy and current policies (The State Council of China, 2016). However, these industrial production activities are ultimately driven by final demand of consumers. Accounting for life-cycle emissions generated throughout entire supply chains allows to identify downstream consumers in the supply chain, to provide them with information about their impact to heavy metal pollution elsewhere, and consequently to involve them in pollution control through influencing consumption choices (Liang et al., 2014).

Consumption-based emission and resource accounting has grown significantly after a discussion about the principles of producer and consumer responsibility since the end of the 20th century (Eder and Narodoslawsky, 1999; Lenzen et al., 2007; Munksgaard and Pedersen, 2001), which is widely used in accounting for global and regional carbon emissions (Davis and Caldeira, 2010; Feng et al., 2013; Peters, 2008) or water footprint (Feng et al., 2014; Zhang and Anadon, 2013, 2014) as well as natural resources (Verones et al., 2017; Wiedmann et al., 2015a, 2015b; Yu et al., 2013). In recent years, consumption-based accounting of air pollution (Liu and Wang, 2017; Moran and Kanemoto, 2016; Zhao et al., 2015), related air quality (Lin et al., 2014, 2016) and health impacts (Jiang et al., 2015; Zhang et al., 2017a) embodied in global or interregional trade has achieved wide-spread recognition. Regarding heavy metals in China, interregional virtual atmospheric heavy metal flows were found to be generally transferred from inland regions to the east coast (Liang et al., 2014). Unlike atmospheric emissions that have been discussed by existing studies, aquatic heavy metals (AHM) emissions have rarely been quantified so far (Zhang et al., 2017b).

To fill the gap, this study compiled a bottom-up AHM emission inventory, including mercury (Hg), cadmium (Cd), chromium (Cr), arsenic (As), and lead (Pb), for 30 industrial sectors in 30 provinces of China, which were based on *First China Pollution Source (FCPSC)* database (Editorial committee on First China Pollution Source Census, 2011). We then used an environmentally extended multiregional input-output (EE-MRIO) model to calculate the AHM emissions from a consumer perspective and analyzed interprovincial virtual transfers triggered by cross-provincial trade. Furthermore, we developed an integrated index, Equal Risk Pollution Load (ERPL), to measure the risk transferred along with five AHM based on toxicity weights.

## 2. Methodology and data

### 2.1. Environmentally extended multiregional Input−Output analysis

Environmental extended input-output analysis is a widely used approach to analyze the technological and environmental ties between economic sectors based on input-output tables (Miller and Blair, 2009; Wiedmann, 2009). The advantage of the input-output approach lies in the ability to reflect the direct and indirect environmental impacts between final production of one sector and other sectors through Leontief inverse matrix (Wiedmann, 2009). Multi-regional input-output (MRIO) analysis, which can reflect the inter-regional trade of commodities and services, is an extension of single region input-output models. In combination of regional and sectoral resource and emission inventories, the MRIO model is widely used for calculation of "consumption-based" or "trade-embodied" accounting of global or regional environmental impacts (Hubacek et al., 2017).

Here, we apply China's MRIO of 2010 rather than the latest year of 2012 after considering the availability of matched bottom-up emission inventory data (see section 2.2 for more details). The MRIO table was compiled by Liu et al. (2014), based on China's original provincial monetary input-output tables for 2010, which has been widely used for similar studies (Mi et al., 2017; Wang et al., 2018; Wu and Wang, 2017; Zhao et al., 2016, 2017). The MRIO table includes 30 provinces (excluding Tibet, Hong Kong, Macau and Taiwan due to the lack of data) and 30 economic sectors for each province (Table S2 in Supplementary Information (SI)).

There are $m$ regions, each with $n$ sectors, each of which is assumed to produce a homogenous product (Zhang and Anadon, 2014). Following our previous MRIO framework (Feng et al., 2014), the technical coefficient submatrix $A^{rs} = (a_{ij}^{rs})$ is given by $a_{ij}^{rs} = z_{ij}^{rs}/x_j^s$, in which $z_{ij}^{rs}$ is the inter-sector monetary flow from sector $i$ in region $r$ to sector $j$ in region $s$; $x_j^s$ is the total output of sector $j$ in region $s$. $y_i^{rs}$ is a final demand vector revealing the final domestic demand (contains household and government consumption, and capital formation) of region $s$ for goods produced by region $r$'s sector $i$. Note that the international export was not considered in this study because this paper mainly focus on domestic inter-provincial trade-related emissions. In this paper, matrices are indicated by italicized capital letters; vectors are denoted by italicized lower case letters; the notation ˆ indicates the diagonalization of corresponding column vectors. Then, we have,

$$\boldsymbol{x} = (I - A)^{-1} \times \boldsymbol{y}^d \tag{1}$$

Here, $\boldsymbol{x}$ represents the total output caused by domestic final demand; $I$ is identity matrix and $(I - A)^{-1}$ is the Leontief inverse matrix, which captures both direct and indirect inputs to satisfy one unit of domestic final demand in monetary value (Miller and Blair, 2009). $\boldsymbol{y^d}$ denotes total domestic final demand supplied by region $r$ with elements $y_i^r = \sum_s y_i^{rs}$.

$\boldsymbol{f}$ refers to sectoral AHM emissions intensity (i.e. Hg, Cd, Cr, As and Pb) for each region, respectively. $\boldsymbol{f}$ is a column vector of AHM emission per unit of economic output for all economic sectors in all regions. $\hat{\boldsymbol{y}}^s$ and $\hat{\boldsymbol{y}}^r$ are diagonal matrixes with the corresponding final domestic demands supplied by all regions and consumed by region $s$ and $r$, respectively. Then we have following equations,

$$E^{rs} = \hat{f}^r (I - A)^{-1} \hat{\boldsymbol{y}}^s \tag{2}$$

$$E^{rs} = \hat{f}^s (I - A)^{-1} \hat{\boldsymbol{y}}^r \tag{3}$$

$$E_{net}^{rs} = E^{rs} - E^{sr} \tag{4}$$

Here, $\hat{f}^r$ and $\hat{f}^s$ are diagonal matrixes with the corresponding values of sectoral emissions intensity for region $s$ and $r$, respectively, but zeroes for all other regions. $E^{rs}$ denotes virtual AHM emissions flows from region $r$ to region $s$ induced by their goods exchange; and $E^{sr}$ denotes virtual AHM emissions flows from region $s$ to region $r$. $E_{net}^{rs}$ denotes net virtual AHM emissions flows between region $s$ and region $r$. If $E_{net}^{rs} > 0$, it means net emission flows from region $r$ to region $s$; if $E_{net}^{rs} < 0$, it means net emission flows from region $s$ to region $r$.

$$E_p^r = \sum_{s=1}^{m} \hat{f}^r (I - A)^{-1} \hat{\boldsymbol{y}}^s \tag{5}$$

$$E_c^r = \sum_{s=1}^{m} \hat{f}^s (I - A)^{-1} \hat{\boldsymbol{y}}^r \tag{6}$$

Here, $E_p^r$ represents total emissions occurred in region r induced by all regions' final domestic demands, which is called production-based accounting of AHM emissions for region $r$; and $E_c^r$ represents total emissions occurred in all regions and induced by region $r$'s final domestic demands, which is called consumption-based accounting of AHM emissions for region $r$.

### 2.2. Bottom-up AHM emission inventory compilation

Most existing bottom-up heavy metal accounting focuses on atmospheric emissions rather than its release into the water body (Tian et al., 2015). Few studies focus on AHM in China, which are coarsely downscaled from global emission inventories like Nriagu and Pacyna (1988), Liu et al. (2016). These downscaled inventories are limited by the lack of localized data, particularly refined emission factors from various industrial sectors in different provinces (Tian et al., 2015). In this context, it's necessary to compile a relatively reliable bottom-up AHM emission inventory for China for it is the basis for the total analysis of virtual flows and associated risk.

A bottom-up approach was adopted to evaluate AHM emissions from China's industrial sources in 2010. The detailed process of emission inventory compilation has been elaborated and published by our previous research (Wu et al., 2018). In short, the AHM emissions in 2010 were calculated based on the sectoral activity level in 2010 and the integrate emission factors from the First China Pollution Source (FCPSC) database as shown in the following equation.

$$E_T = \sum_i \sum_j (A_{i,j}(t) \cdot EF_{i,j,p}(t)) \tag{7}$$

where $E_T$ is the total emissions of AHM, $A$ is the annual activity level (the gross outputs) that were obtained from 2010 prefectural statistical documents, $EF$ is the AHM-specific EFs for the industrial sectors, $t$ is the calendar year, $i$ is the prefectural city, $j$ is the industrial sector, and $p$ is the type of AHM.

To calculate the AHM-specific EFs for the industrial sector, we developed an integrated EF model based on FCPSC database as shown in the following equation.

$$EF_{i,j,p}(t) = COI_{\ i,j,p}(t) \cdot (1 - PF_{i,j,p}(t)) \tag{8}$$

where $EF$ is the AHM-specific EFs for the industrial sectors; $COI$ is the released AHM amounts per gross output value; $PF$ represents the removal rate of end-of-pipe control measures for AHM emissions; $t$ is the calendar year, $i$ is the prefectural city, $j$ is the industrial sector, and $p$ is the type of AHM. The released AHM amounts per gross output value and the removal rate of end-of-pipe control measures for AHM emissions for different sectors were directly obtained from the FCPSC Database. It should be noted that the FCPSC database is regarded as one of the most authoritative survey data sources for pollution in China. This bottom-up emission inventory contains five types of industrial AHM, including Hg, Cd, Cr, As and Pb. It almost covers all officially registered enterprises in China including 1.58 million industrial sources (Editorial committee on First China Pollution Source Census, 2011; Wang et al., 2014). Moreover, we merged our city-level inventory to provincial-level one in order to correspond to the MRIO table of 2010.

### 2.3. Equal Risk Pollution Load

Generally, industrial AHM emissions would increase the heavy metal level in soils and potentially contaminate the food system (Chang et al., 2014). Oral intake seems to be most important way through which AHM in industrial wastewater enters the human body and causes health risks (Zhang et al., 2017b). To measure the risks of different types of heavy metals, the index of ERPL was developed by weighting the emissions of five types of heavy metals by the corresponding oral reference dose for risk potential. The oral reference dose indicates noncarcinogenic risks of a certain type of heavy metal that are directly demonstrated by toxicity weights from USEPA (USEPA, 2017). The measurement unit of oral reference dose is mg/kg/day. In the calculation of ERPL, however, we only take the values of oral reference dose without the measurement unit as the weights. Therefore, the measurement unit of ERPL is the same with heavy metal emissions. ERPL provides a coarse but integrated measure on the joint hidden risk potentials. The index of ERPL is calculated as follows:

$$ERPL_i = \sum_p (E_{ip}/RfD_i) \tag{9}$$

where $i$ and $p$ are the prefectural city and the type of AHM, respectively; $ERPL$ refers to the Equal Risk Pollution Load; $E$ refers to AHM emissions; $RfD$ refers to the risk potential weight, the value of which equals to the oral reference dose. The values of oral reference dose for Hg, Cd, Cr, As and Pb are 0.0001, 0.0005 0.003, 0.0003 and 0.00006, respectively. They are used by EPA's Risk-Screening Environmental Indicators (RSEI) model to get RSEI results, including RSEI Hazard and RSEI Scores (USEPA, 2017). It has also been widely used in international research (Cordioli et al., 2013; Palmiotto et al., 2014; Zhang et al., 2017b).

Theoretically, actual AHM intakes rather than AHM emissions should be combined with reference dose to determine the risk. In this case, however, estimating AHM intakes based on AHM emissions are greatly challenged by the complicated pollutant transfer and transformation processes from upstream of the rivers to downstream as well as the unclear exposure behavior patterns. Thus, the definition of ERPL in this study only indicated the potential for causing the associated risk rather than the actual risk level.

## 3. Results

In 2010, national total emissions of aquatic Hg, Cd, Cr, As, and Pb amounted to 1.8 tons (t), 52.1t, 1702.9t, 259.8t, and 245.4t. After weighting the AHM emissions with the corresponding reference dose, the value of ERPL was calculated to be 5850.3 t in 2010, indicating the AHM associated health risk potential.

Fig. 1a showed the AHM emission characteristics by sector. The most significantly negative differences of the producer and consumer perspectives were observed in sectors of metal smelting, metal products, chemical, metal mining and clothing-leather. Among them, metal smelting sector was the main sectoral emitter of Hg, Cd, As, Pb emissions and ERPL from the producer perspective, accounting for 28.2%, 78.8%, 29.3%, 51.9% and 44.0% of the national total, respectively. However, the emissions of metal smelting were ignorable from the consumer perspective. Regarding the chemical sector, it contributed 38.6% and 43.3% of national total Hg and As emissions from the producer perspective while it accounted for only about 8% from the consumer perspective. Similar patterns were observed in sectors of metal products and Clothing-Leather for Cr, and metal mining for Pb, Hg and As.

Almost all sectors with significantly negative differences of the producer and consumer perspective are located upstream in the supply chains. In other words, these sectors produced primary products (e.g., mineral ores) and semi-manufactured products (e.g., metals products) as inputs for further processing in downstream production for final use products e.g. electrical appliances, electronic devices. Correspondingly, the downstream sectors, such as construction activities, equipment manufacturing and services, have the most significantly positive differences of the producer and consumer perspectives. Taking construction sector as an example, it has no direct AHM emissions but with large indirect emissions of Hg, Cd, Cr, As, Pb and ERPL (36.1%, 44.9%, 28.3%, 34.4%, 43.8% and 40.9% of the national total, respectively) associated with their intermediate material inputs such as metal products and fuels.

The dependence of regions on the specific industries determined the AHM emission and ERPL characteristics by regions. We observed significant differences between production-based and consumption-based AHM emissions and ERPL among 30 provinces (Fig. 1b), indicating large transfers of AHM emission and the risk embodied in interprovincial trades. Specifically, the economies of Hunan, Inner Mongolia, Yunnan, Guangxi, Jiangxi, and Gansu dominated by upstream industries caused more AHM emissions from the producer perspective (Fig. 1b). As the relatively developed regions located downstream of China's supply chains, the economy of Zhejiang, Jiangsu, Shandong,

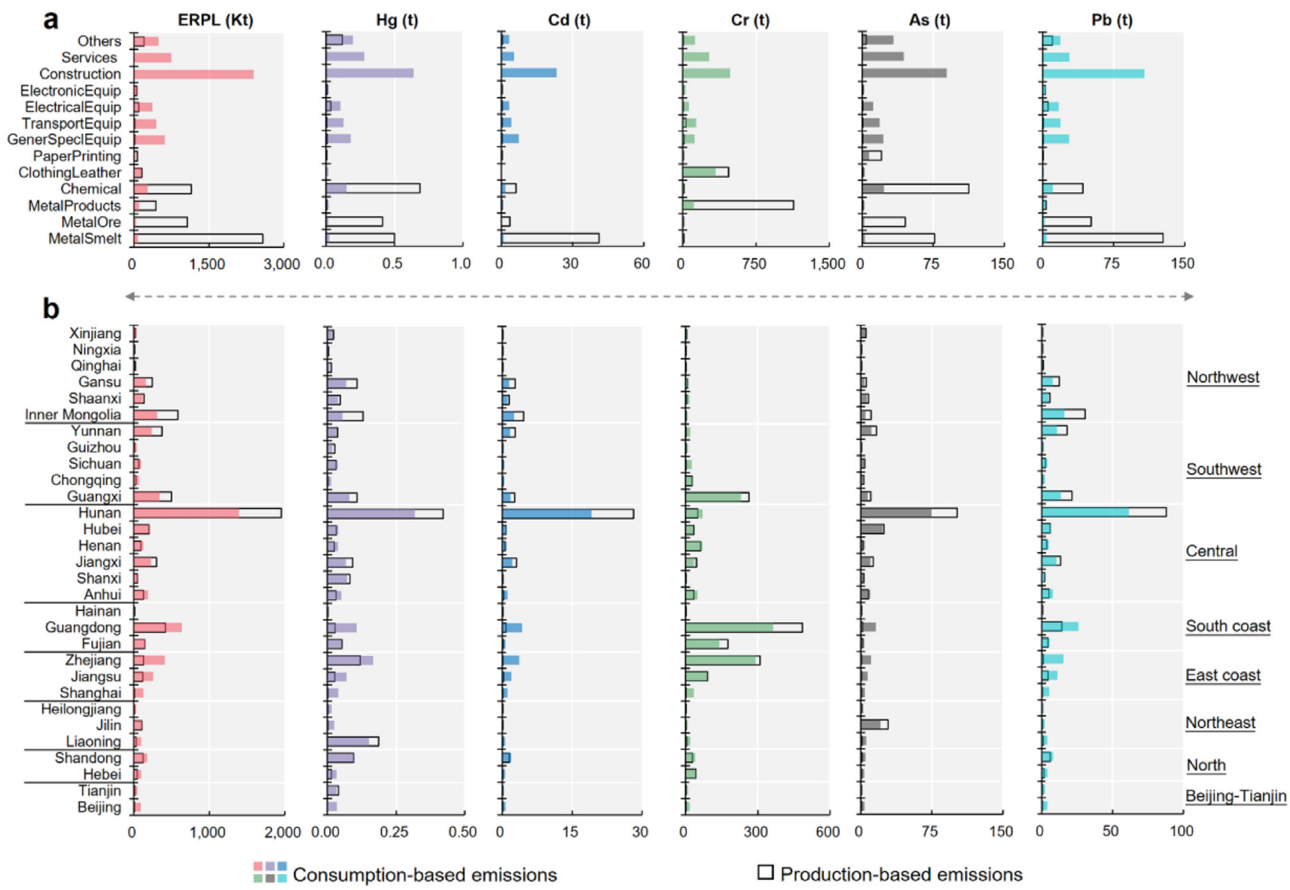


**Fig. 1.** The emissions of aquatic heavy metals (AHM) and ERPL in China from both the producer and consumer perspectives: (a) emissions by sector; (b) emissions by province.

Beijing, Tianjin, Shanghai, Sichuan, Chongqing caused more emissions from the consumer perspective (Fig. 1b). The case of Guangdong is tricky. For Hg, Cd, As, Pb and ERPL, Guangdong shared the similar patterns to other developed provinces and emitted less from the producer perspective. However, it caused more Cr emissions from the producer perspective, which is a differential finding from the common census of prior studies.

After aggregating the results of interprovincial emission transfer matrix (30 × 30), we identified the main net exporters and importers of ERPL and five AHM emissions (Fig. 2). The top net exporters and importers of Hg, Cd, As, Pb and ERPL are quite similar. Less-developed provinces located downstream in the supply chain such as Hunan, Inner Mongolia, Guangxi, Yunnan, Gansu, Jiangxi, Jilin and Liaoning are the top net exporters. Developed provinces such as Guangdong, Zhejiang, Jiangsu, Shanghai, Beijing were the top net importers. Among them, an important case in term of AHM among them is Hunan province which contribute the most to virtual net exports of aquatic Hg, Cd, As, Pb and ERPL as well as the emissions from both the producer and the consumer perspective. Specifically, Hunan discharged 23.8%, 54.0%, 39.2%, 35.7% and 33.3% of national total aquatic Hg, Cd, As, Pb emissions and ERPL, respectively (Fig. 1b). The net emissions of Hg, Cd, As, Pb and ERPL incurred by Hunan amounted to 0.1t, 9.0t, 26.9t, 26.1t and 557.8t (Fig. 2), of which 28.2%, 24.4%, 26.9%, 24.2% and 24.1% were incurred for the production of exports to Guangdong, 15.3%, 22.2%, 15.6%, 22.2% and 21.2% to Zhejiang (Fig. 3).

The patterns above were not applicable to the case of Cr. We observed aquatic Cr emissions were incurred in some richer coastal regions such as Guangdong, Fujian, Zhejiang that extensively export electroplating products to less developed inland provinces. This pattern is opposite to the case of Hg, Cd, As, Pb emissions and ERPL as well as the common scientific census. Among them, Guangdong is the largest net exporter with total exports of 120.7t, of which 12.1% were exported to Sichuan, 11.5% to Hunan, 10.6% to Yunnan, 8.9% to Shanghai, 8.7% to Jiangsu, and 7.3% to Zhejiang (Fig. 3).

## 4. Discussion and policy implications

Interprovincial trade-related aquatic heavy metal emissions show different patterns from other embodied pollution (i.e. carbon dioxide (Feng et al., 2013; Su and Ang, 2014; Zhang et al., 2014), sulfur dioxide (Liu and Wang, 2017), nitrogen oxides and fine particular matters (Zhao et al., 2015), etc.) as well as atmospheric heavy metal emissions (Liang et al., 2014). In terms of atmospheric pollutants, China's interregional trade-related emissions transfers show a close relationship with fuel mix and energy use, industrial structure as well as regional development level. Virtual atmospheric emissions were mostly transferred from less developed central and western China (Shanxi, Hebei, Guizhou, etc.) with primary energy or pollution-intensity products to richer coastal regions (Jiangsu, Shanghai, Guangdong, etc.) (Hubacek et al., 2017), whereas AHM transfers are quite different. The provinces (such as Hunan, Yunnan, Inner Mongolia, Guangxi) with nonferrous-metallic mineral resources were the origins of virtual transfers of aquatic Hg, Cd, As and Pb to either developed or less developed regions. For the first time, we found that the virtual flows of aquatic Cr emissions were incurred in the richer coastal regions, especially Guangdong province, that extensively export electroplating products and printed circuit board to less developed inland provinces. Such a transfer pattern has rarely been performed before, which call for innovative solutions to address it. To further understand the underlying drivers, we give a detailed supply chain analysis for Guangdong provinces (Fig. 4). More

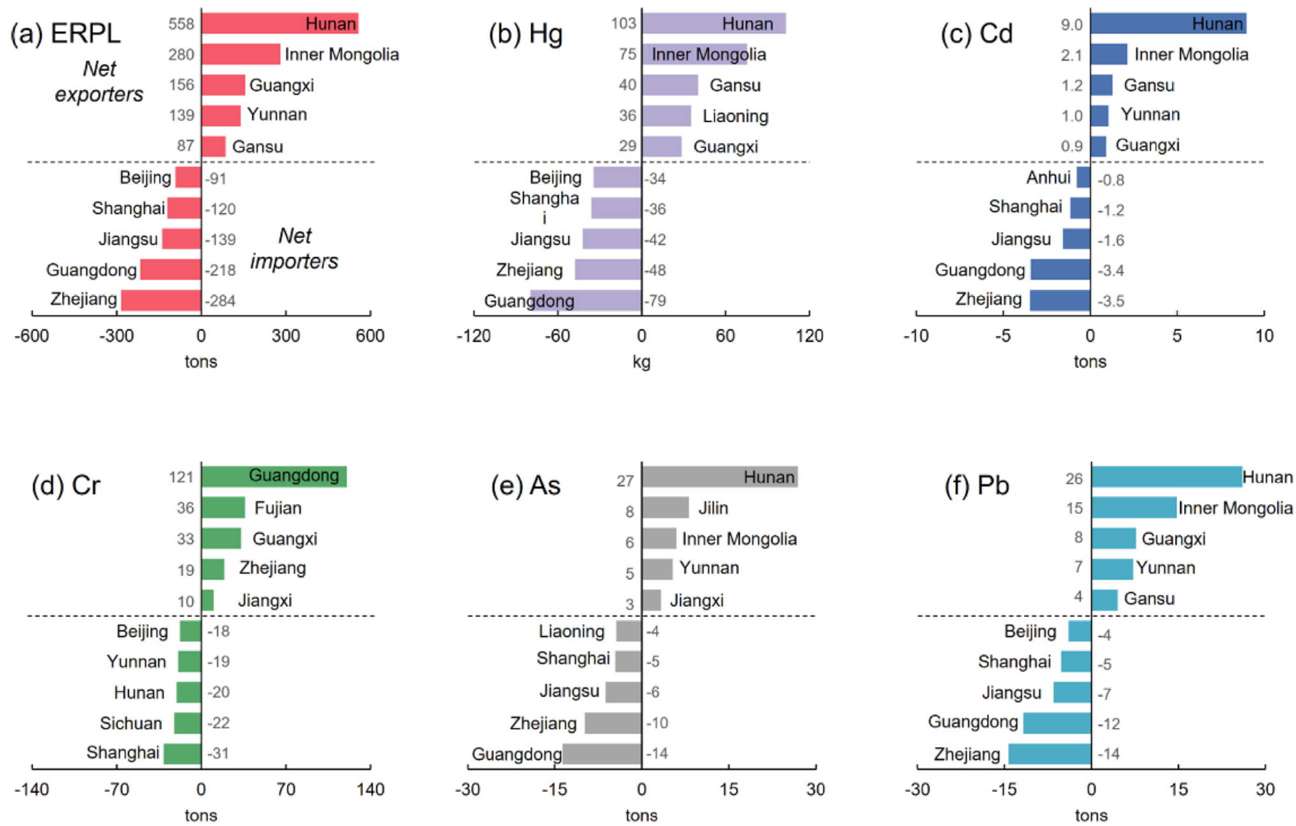


**Fig. 2.** Top five net exporters and importers of ERPL and five AHM emissions in 2010.

than 40% of Cr emissions in Guangdong are related to products and services that are locally produced but consumed in other regions. The supply chains associated with electroplating products in Guangdong are mostly equipment, construction activities and services. Through these primary chains, Zhejiang, Hunan and Guangxi are the three provinces outsourcing the largest amount of Cr emissions to Guangdong. Specifically, Zhejiang outsourced Cr emissions mainly by importing electroplating products from Guangdong to its construction (7.7t, 25.3%) and general and special equipment industry (7.7t, 25.1%). Hunan outsourced Cr emissions mainly from "Guangdong-Metal products" to "Hunan-Construction" (4.9t, 31.6%) and "Hunan- General and special equipment" (4.0t, 26.0%). The supply chains between Guangxi and Guangdong are the third largest, mainly from "Guangdong-Metal Products" to "Guangxi - Construction" (8.6t, 60.0%).

Given the great toxicity of heavy metals (Zhang et al., 2017b), huge environmental risk potentials would be hidden behind large flows of AHM emissions. The index of ERPL we developed in this study provides a rough but integrate measure on the joint hidden risk. The results of ERPL revealed that Yunnan, Hunan, Guangxi and Inner Mongolia provinces with non-ferrous metal mineral resources obtained economic benefits by exporting primary and semi-manufactured metal products but burdened with huge joint hidden risk. Among them, the most important province in terms of AHM related risk is Hunan province that deserved a detailed supply chain analysis. The supply chains associated with metal smelting, the backbone industry in Hunan are mostly for the production of general and special equipment, transport equipment and construction activities. Through them, Guangdong and Zhejiang are the two dominant provinces outsourcing Cd emissions to Hunan. Specifically, Guangdong outsourcing Cd emissions mainly by importing semi-manufactured metal products from Hunan to its construction industry (0.8t, 37.8%), electrical equipment industry (0.3t, 13.4%), and transport equipment industry (0.2t, 10.0%). The supply chains between Zhejiang and Hunan are mainly from "Hunan-Metal smelting" to "Zhejiang-Construction" (0.8t, 39.7%), "Zhejiang- General and special equipment" (0.4t, 23.4%), "Zhejiang-Transport equipment" (0.3t, 15.2%). In general, almost one third of Cd emissions in Hunan are related to products and service consumed in other regions (Fig. 4). The transfer of Cd emissions to Hunan no wonder intensified the situations of great public health concerns attributable to Hunan cadmium contaminated rice events (Williams et al., 2009). What make things worse is that as the leading rice producer in China, the risk of Hunan cadmium rice are further amplified by the rice supply chain and transferred to many provinces like Guangdong (Nanfang Daily, 2013). This makes the transfer of hidden risk more complicated and greatly change the original risk patterns within the country. At this stage, the hidden risk in trade are still poorly understood by both governments and the lay public. Local governments have so far ignored supply chain thinking in their frameworks for national risk management and risk communication. The lay public does not know of potential AHM-related risks and potential income effects for mitigating those risks.

The findings of this study also have important policy implications. Provinces that have frequent trade of high AHM-intensive products should share both production-based and consumption-based AHM emission reduction responsibilities in the future according to studies suggesting demand-side policies (Chen et al., 2018; Peters, 2008; Rodrigues and Domingos, 2008). However, trade patterns and virtual flows of Hg, Cd, As, and Pb were quite different from those of Cr, which suggests different shares of responsibility along the supply chains influencing effectiveness of policies and pollution control.

The results show that the main emitters of Hg, Cd, As and Pb were low value-added industries such as metal smelting. As suggested by previous research, the lower the profits and tax revenues the industry generates, the lower the local protection the sector receives from stakeholders (Bai et al., 2004). This would imply that stricter environmental regulations and enforcement of existing policies such as enhanced end-of-pipe emission controls imposed upon these industries

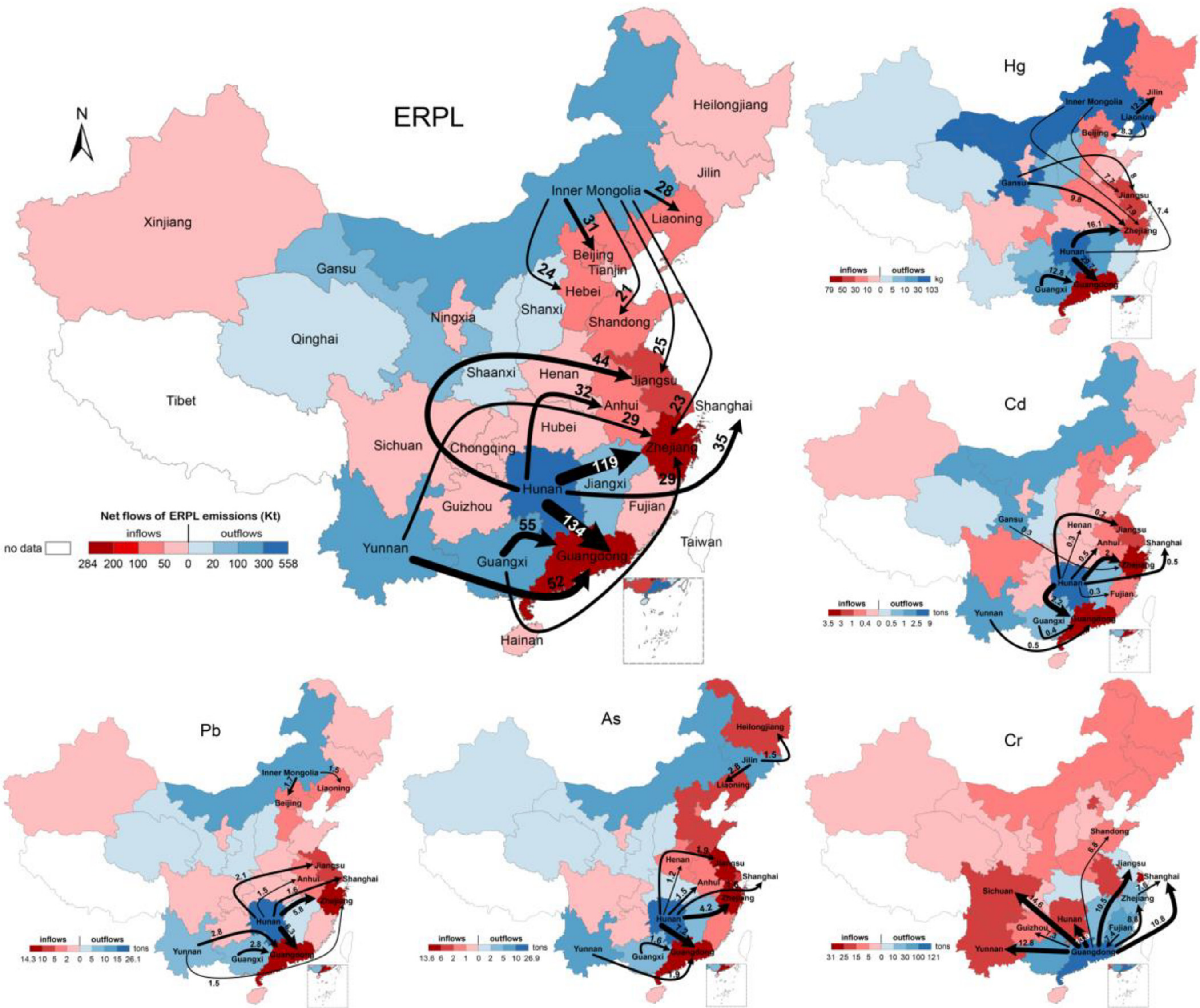


**Fig. 3.** Interprovincial AHM emission and ERPL spillovers in China. The colors of the map indicate the amount of total net inflows or outflows in a certain province. The numbers expressed by arrows indicate the amounts of net flows of embodied emissions from one province to another. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

would receive relatively weaker resistance from interest groups. Meanwhile, the 'accessibility' to nonferrous mineral ores in China is controlled by a few big state-owned metal smelting enterprises (Environmental Protection Department of Hunan, 2016). In such a supplier market, downstream producers have little influence on upstream producers' production behavior, let alone their environmental performance in controlling aquatic Hg, Cd, As, and Pb. In this context, we suggest that upstream producers take a higher share of the responsibility for mitigation than downstream producers in controlling aquatic Hg, Cd, As, and Pb pollution from industrial wastewater. The government should provide legal and/or monetary incentives for upstream producers by setting strict emission standards and internalizing environmental costs to prices of products through imposing taxes on AHM-related products. The right mix of legal and market instruments should depend on the toxicity of the heavy metals and prices might not be sufficient to ensure public health outcomes. Depending on the market structure and consumers' price elasticity a share of the mitigation costs will be transferred to final consumer. For example, imposing environmental taxation on heavy metal emissions based on its ecosystem damage may not only improve the production efficiency of upstream producers using heavy metal as production inputs, but also induces final consumers to pay for their impacts which may reshape their consumption choices and reduce their consumption-based heavy metal footprint (Liang et al., 2014).

However, for controlling aquatic Cr pollution, the story seems to be different. On the one hand, upstream producers are relatively higher value-added industries producing products, which are more likely to receive local protection (Bai et al., 2004). In this context, stricter environmental regulations on these upstream producers would suffer greater resistance from interest groups. In addition, downstream producers are mostly well-known or transnational corporations, e.g. Apple, Samsung, Haier, Huawei, Nike, Toyota, etc., with relatively strict technical and environmental standards also for upstream inputs such as printed circuit boards, electroplated accessories, and leather from Guangdong and Zhejiang. They usually use contractual mechanisms to regulate upstream producers' production behavior. More importantly, good life-cycle environmental performance and corporate social responsibility is one of the most important ways for these well-known transnational corporations to shape their social reputation (Ma et al., 2017), which provides the government with an opportunity to involve

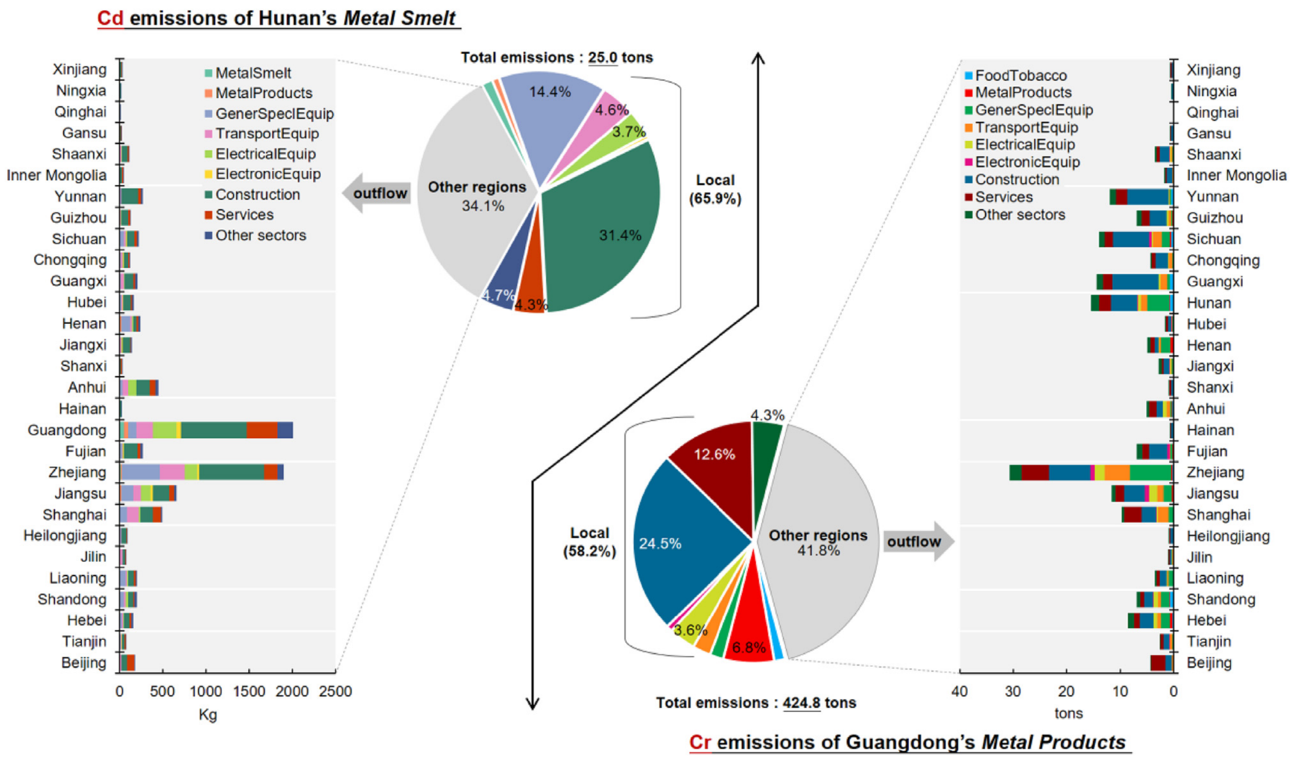


**Fig. 4.** Consumer-based Cd emissions from Metal Smelting in Hunan and Cr emissions from Metal Products in Guangdong. The grey in pie charts indicates the proportion of emissions related to products and service that are locally produced but consumed by other regions. Other colors in pie charts indicate the proportions of emissions related to products and services that are locally produced and also consumed by local industries. The bar charts indicate the emissions related to products and services that are locally produced but consumed by different industries in other regions. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

these corporations in controlling aquatic Cr emissions through wielding their market power, logistical prowess, and vertical integration or contractual mechanism. To further incentivize such upstream supply chain management, the government could introduce an environmental tax or product standards that take consumption-based emission and ERLP index into account.

## 5. Uncertainty analysis

### 5.1. Uncertainties from emission inventory

Our AHM emission inventory are calculated based on the sectoral activity level in 2010 and the emission factors from the First China Pollution Source (FCPSC) database for 2007. There is no wonder that the activity levels in 2010 can partially reflect the impacts of global financial crisis of 2008 on economy growth and industrial structures that are critical for AHM emissions. However, it should be noted that assuming emission factors of 2010 the same as those of 2007 will introduce some bias. There absolutely are gaps in technical levels between 2007 and 2010. Moreover, the global financial crises between 2007 and 2010 will also change the technical structure and technical levels. For example, many small-scale enterprises usually with larger emission intensities were closed down during the global financial crisis. Thus, the ignorance of technical improvement will result in the overestimates of AHM emissions. However, there are no available emission factor database better than FCPSC so far. The FCPSC are state actions with strong law enforcement, which definitely ensured that the individual enterprises would submit their real information as the decree of the State Council of the People's Republic of China (No. 508) required. The FCPSC has established strict and standard procedures in data collection, field investigation, data analysis, and data quality assurance/quality control etc. in First China Pollution Source Census Implementing Plan. According to Decree No. 508, all regulations, criteria, and plans are expected to keep the surveyed data in relatively uniform quality in terms of errors and uncertainty. Additionally, errors and uncertainty are expected to be kept as low as possible.

There are 9 technical regulations and 5 procedure regulations regarding each step of data collection, including field survey, field monitoring, data checking, data entry to the system, data aggregation, etc. Field survey means data collection and reporting in site (industrial factories etc.) by staff of local environmental protection agencies; Field monitoring is deployed with online monitoring system, such as CEMS (Continuous Emissions Monitoring System) on stack, and site environmental monitoring; Data checking mainly refers to checking the original data by local environmental protection agencies, such including magnitude and units checking; Data entry refers to the entry of the original data (paper based tables) into the system; Data aggregation refers to aggregating the dataset bottom up to national level, namely from dataset in local environmental protection bureau to city environmental protection agencies, then to provincial environmental protection agencies and ultimately to MEP.

The leading group of the FCPSC implemented the first data quality check immediately following the completion of the data collection. This is accomplished by random sampling of 1274 industrial sources, 2946 household sources, 166 pollution treatment facilities, and 3871 agriculture sources. The qualified data (data errors less than 2%) accounted for 87.47% of the total number of the sample. The FCPSC units at every level (province, city, and county) were required by the leading group to carefully verify the data according to the problems identified in the first data quality check. The second data quality check included 300,800 samples (including industrial sources, household sources, pollution treatment facilities, and agriculture sources). The data quality has been improved significantly after this national wide data validation and verification. The qualified data accounted for 94.48% of the total in the second data quality check. Therefore, we can say that the uncertainty of the FCPSC data is no more that 6%. The FCPSC are widely used in existing studies (Cai et al., 2016; Liang et al., 2017; Wang et al., 2014; Xue et al., 2013; Zhang et al., 2015), which can indicate that our AHM emission inventory based on FCPSC are reliable. Certainly, future updates on AHM emission inventory are encouraged once the latest emission factor database are available.

### 5.2. Uncertainties from MRIOT

There are additional uncertainties from MRIO analysis which include but is not limited to linking trade through supply chain among different regions. Uncertainties mostly come from the source (survey) data and data manipulation of MRIO model (Lenzen et al., 2010; Wiedmann et al., 2008; Wilting, 2012). Moreover, the uncertainties may also come from MRIO's sector detail, region coverage, and the number of environmental extensions (Moran and Wood, 2014; Tukker and Dietzenbacher, 2013; Wiedmann et al., 2015b; Rodrigues et al., 2018). A study from Lin et al. (2014) found that the uncertainties in Chinese input-output model are relative small, which contribute about 10% of total errors in export-related pollutant emissions (Lin et al., 2014). The MRIO table we used is based on each province's single IOT in 2010 released by National Statistics Bureau (NSB). The method to compile the MRIO table we used is also scientific and reasonable, which is widely used to in published paper or book (Chen et al., 2015; Zhang et al., 2012; Zhang and Qi, 2012). Regarding the sector resolution of Chinese MRIO, some sectors can be directly matched to the important sectors in term of AHM emissions; some are even divided at more refined sector resolution. See Table S4 for the mapping relations from the important AHM sectors to the sectors in Chinese MRIO table. It shows the validity to use Chinese MRIO because the important sectors in term of AHM are different from IO classification.

## 6. Conclusions

In this study, we estimated both production-based and consumption-based AHM emissions, the associated risk potential and their virtual flows among 30 provinces of China. Significant differences between production-based and consumption-based AHM emissions and ERPL were observed, indicating large transfers of AHM emission embodied in interprovincial trades. The virtual transfer patterns in terms of AHM are quite different from the common scientific census on other pollutants, which calls for additional attentions for differentiated strategies. In addition, the risk transferred together with AHM in the trade complicated the original risk patterns in China, which need to be further explored in the future study.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2019.109400.

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