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Transboundary air pollution and cross-border cooperation: Insights from marine vessel emissions regulations in Hong Kong and Shenzhen

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ABSTRACT

Many coastal cities regulate shipping emissions within their jurisdictions. However, the transboundary nature of air pollution makes such efforts largely ineffective unless they are accompanied by reciprocal, legally-binding regulatory agreements with neighbouring cities. Due to various technical, economic, and institutional barriers, it has thus far been difficult to isolate the effects of legally-binding cross-border cooperation on vessel emissions at the city-level. We exploit the unique administrative characteristics of Hong Kong and its relationship with neighbouring cities in China's Pearl River Delta to isolate the effect of legally-binding cross-border cooperation. Using a regression discontinuity design, we find that Hong Kong's unilateral implementation of marine vessel fuel control policy left the city exposed to SO_2 from marine vessel emissions originating in Shenzhen. Only when Shenzhen implemented its own legally binding policy did such pollution in Hong Kong reduce significantly across all seasons. While international agreements on air pollution are important, they face well-known difficulties related to scale and multilateral complexity. Our findings therefore suggest that contiguous cities—-whether or not they straddle an international border—can play an important role in the timely development of effective emissions standards.

1. Introduction

Major port cities facilitate global trade and act as drivers of economic growth. At the same time, trade via marine vessels causes significant air pollution that negatively impacts residents in these cities (Kalayci, 2019; Mao et al., 2017). Direct negative effects of air pollution include increases in respiratory disease and premature deaths, along with social costs such as elevated anxiety, psychiatric disorders, and an avoidance of outdoor activities (An, Zhang, Ji, & Guan, 2018; Butt et al., 2017; Delfino, Sioutas, & Malik, 2005; Dockery et al., 2013; Friedrich, Heinen, Kamakaté, & Kodjak, 2007; Gu & Yim, 2016; Neidell, 2009; Stallings-Smith, Zeka, Goodman, Kabir, & Clancy, 2013). Many coastal cities are committed to improving air quality to reduce these hazards by regulating domestic emissions and transitioning to renewable energy sources (Alvarez-Herranz, Balsalobre-Lorente, Shahbaz, & Cantos, 2017; Tsai & Chou, 2005).

Unlike most environmental pollutants, air pollutants are transboundary (Yamineva & Romppanen, 2017). Smith et al. (2015) estimate that marine vessels are responsible for roughly 13% of global anthropogenic sulphur dioxide (SO_2) emissions, while Zhang et al. (2017) estimate that around 12% of total premature deaths caused by fine suspended particulates ($PM_{2.5}$) are induced by transboundary air pollutants. Cheung, He, & Pan, 2020 find that particulate matter from the Pearl River Delta (PRD) accounts for 50%–60% of Hong Kong's average air pollution. Similarly, Chuang et al. (2020) determine that average air pollution levels can be affected by $PM_{2.5}$ transported from adjacent regions, due to seasonal winds. The transboundary nature of air pollution means that many coastal cities are unable to reduce air pollutants through domestic policy and regulations alone (Lu et al., 2020). To address transboundary air pollution, cross-border cooperation has been pursued, primarily in North America and Europe (Eades, 2018; Koolen & Rothenberg, 2019; Moses et al., 2020). The International Maritime

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Organization (IMO) has also addressed marine vessel emissions[1] (Topali & Psaraftis, 2019). Despite these efforts across a range of scales, regulations implemented in port cities largely remain non-binding and rudimentary as a result of asymmetries in capacity, economic interests, and political pressure, amongst other reasons (Lee & Paik, 2020; Yamineva & Romppanen, 2017).

One region particularly vulnerable to transboundary air pollution is the PRD, which consists of nine cities in mainland China and the special administrative regions (SAR) of Hong Kong and Macau. One of the world's most densely populated urban areas, the PRD is home to two major port cities: Hong Kong and Shenzhen (WSC, 2021). Estimates suggest that marine vessel emissions in the region contribute around 36% and 54% of ambient SO_2 concentrations in Hong Kong and Shenzhen, respectively (Fung, Zhu, Becque, & Finamore, 2014; Mao et al., 2017; Mason et al., 2019). Emissions from marine vessels represent a major public health concern for the two cities' dense populations (Lai et al., 2013). To address this issue, in December 2008, China's National Reform and Development Commission suggested measures to improve air quality with respect to marine vessel emissions in the Plan for the Reform and Development of the Pearl River Delta (2008–2020). This led authorities in both Hong Kong and Shenzhen to implement new marine vessel emission standards requiring low-sulphur fuel, on which we elaborate in Section 2. It is worth noting that although the IMO introduced a global sulphur cap of 3.5% m/m on vessel emissions three years prior to Hong Kong's Fuel for Vessels policy, the global guidelines were inadequate for addressing locally-specific conditions. When Hong Kong implemented the Fuel for Vessel policy with 0.5% sulphur cap in 2015, the MARPOL guideline (3.5% cap) was still much looser. If Hong Kong and Shenzhen had followed this international guideline, it would have had little effect on immediately improving local air pollution resulting from vessel emissions.

We exploit the staggered implementation of marine vessel emission standards in Hong Kong and Shenzhen to isolate the effect of transboundary cooperation between two cities on marine vessel emissions and air pollution. Due to differing economic interests and state capacities, and as a result of differing domestic pressures, attempts to reach state-to-state agreements on air pollution often falter (Lee & Paik, 2020). By contrast, cities within the same country are subject to the same national policy, allowing them to implement their goals at the same time or in phases by region (NEPIA, 2020). Although Hong Kong and Shenzhen are part of one country, Hong Kong remains autonomous with respect to its socio-economic and environmental policies and functions as a separate customs territory. The city's SAR status enables the Hong Kong government to implement policies according to separate timelines. Hong Kong's status therefore provides an opportunity for a natural experiment that tests the potential efficacy of city-to-city bilateralism as it can exist apart from the alternately permissive or restrictive features of intra- or inter-state relations, respectively. A proof-of-concept for such bilateralism has the potential to be broadly relevant, providing insights that will be valuable to neighbouring cities, especially cross-border ones, seeking to forge cooperative agreements that need not be rooted in a statist paradigm.

2. Transboundary air pollution in Hong Kong and Shenzhen

Hong Kong and Shenzhen are adjacent cities in the PRD located on the coast of the South China Sea (see Fig. 1). Both cities share the same climate, characterised as a tropical transitional marine climate zone (Bai et al., 2020). Due to the influence of the southwest monsoon in summer, hot and humid air masses mainly originating from the South China Sea and Indochina affect the conurbation (HKO, 2021). Abundant precipitation in both Hong Kong and Shenzhen falls during summer, which is generally conducive to reducing atmospheric pollutants (Bai et al., 2020). Conversely, the East Asian monsoon generates north-easterly prevailing synoptic winds in winter, accelerating the transport of air pollutants from Shenzhen to Hong Kong (Lee & Savtchenko, 2006; Luo, Hou, Gu, Lau, & Yim, 2018; Wong, 2018). These seasonal transboundary air pollution issues have been a major issue in Hong Kong because of unbalanced emissions between the two cities. Over the past several decades, Shenzhen has focused on developing emission-intensive high-tech and manufacturing industries, while Hong Kong's economy remains based on the service industry, which produces relatively lower emissions. All major air pollutants in Shenzhen are higher than Hong Kong (Hopkinson & Stern, 2003), a finding that our data confirms.

Discussions concerning transboundary air pollution in the PRD date back to 1990, when Hong Kong was still a British colony. These early discussions led to the formation of the Hong Kong-Guangdong Environmental Protection Liaison Group, which sought to enhance cooperation on reducing air pollution between the Chinese province and what would become the Hong Kong SAR in 1997 (HKPD, 2002). The efforts of the Liaison Group, however, were hampered by a lack of transparency and low-quality data. In the late 1990s, however, a marked deterioration in visibility in Hong Kong raised public concern surrounding transboundary air pollution (Hopkinson & Stern, 2003). This catalysed discussions between the Hong Kong and Guangdong governments, which led to a joint study on regional air quality in 2002. A joint air pollution monitoring programme, the PRD Regional Air Monitoring Network, was established in 2005 to monitor SO_2, nitrogen dioxide (NO_2), ozone (O_3), and particulate matter (PM_{10}). In 2014, the programme added carbon monoxide (CO) and fine suspended particulates ($PM_{2.5}$) to its monitoring mandate (RAQMN, 2018). Beyond regional monitoring, however, the Hong Kong government was unwilling to address the more complicated issue of transboundary air pollution, preferring to focus on tackling local sources of air pollution (Hopkinson & Stern, 2003).

The continued growth of maritime trade has pushed discussions about transboundary air pollution to include maritime activities alongside inland ones (Wong, 2018). Aware of the deleterious effects of maritime emissions on Hong Kong and Shenzhen, China's National Reform and Development Commission set out the aforementioned Plan for the Reform and Development of the Pearl River Delta (2008 – 2020), which included a mandate for the Hong Kong SAR and Guangdong provincial governments to tighten marine vessel fuel and emission standards. This was formalised in the Framework Agreement on Hong Kong-Guangdong Cooperation in 2010 (Loh & Booth, 2012). Hong Kong implemented its Fuel for Vessels policy regulating SO_2 emissions from maritime vessels on 1 July 2015, requiring the use of marine fuel with sulphur content not exceeding 0.5 percent. Shenzhen implemented its own low-sulphur fuel policy on 1 October 2016 (EPD, 2019; NEPIA, 2020; Ng, 2016).

3. Methods

3.1. Methods and data

To estimate the effect of local or transboundary air pollution, many studies use panel data with fixed-effects or time series methods with atmospheric models such as Weather Research and Forecast, Community Multiscale Air Quality Modelling System, or back trajectory analysis using the Hybrid Single Particle Lagrangian Integrated Trajectory (HYSPLIT) model (Jung, Choi, & Yoon, 2021; Kwok, Fung, Lau, & Fu, 2010; Reis et al., 2018). Several studies also use a regression discontinuity design (RDD) to find the causal effects of policy interventions (Cai, Nan, Zhao, & Xiao, 2021; Davis, 2008; Fu & Gu, 2017; Viard & Fu, 2015).

Considering these methods, we collected high-frequency daily SO_2

[1] The IMO established the International Convention for the Prevention of Pollution from Ships or Marine Pollution (MARPOL) Annex VI in 1997, forcing a global sulphur cap of 4.5% in bunker fuel in 2005. The IMO further reduced the global sulphur cap to 3.5% in 2012 and 0.5% in 2020 (IMO, 2021).

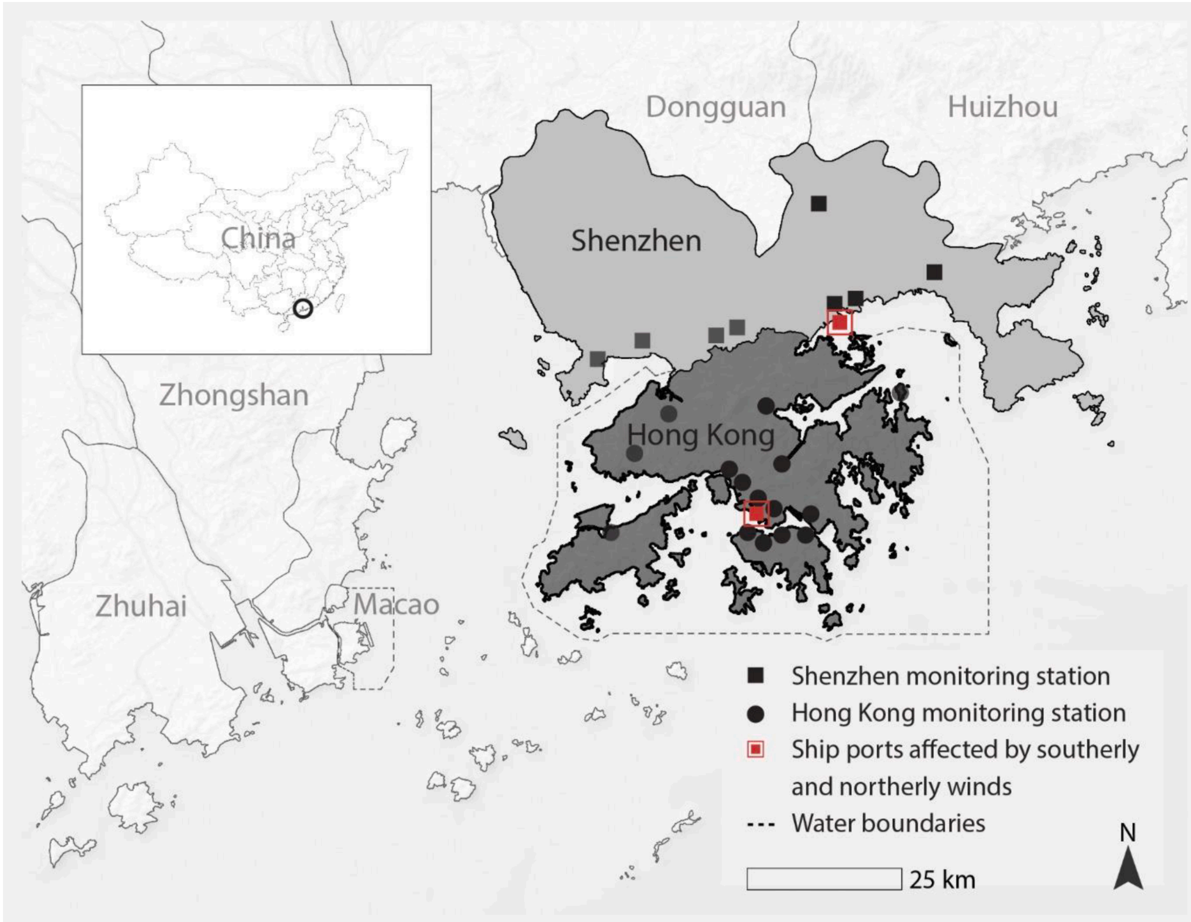

Fig. 1. Hong Kong and Shenzhen.

concentration data from 15 monitoring stations in Hong Kong and nine monitoring stations in Shenzhen between 2013 and 2019 (see Supplementary Figure 1 and Supplementary Table 2). To account for wind direction, we performed trajectory analysis. Specifically, to calculate diffusion directions and distances, we adopted a back trajectory analysis using the U.S. National Oceanic and Atmospheric Administration's (NOAA) HYSPLIT model (see Fig. 2). We used a backward 24-hour trajectory frequency plot from a starting location of 22.334°N, 114.097°E at 10 m above ground level using archived Global Data Assimilation System (GDAS) meteorological data. This allowed us to identify air parcel trajectories and travel distances (Dianat, Radmanesh, Badavi, Mard, & Goudarzi, 2016; Soleimani et al., 2015). In addition to the HYSPLIT simulations, we generated wind rose plots (see Supplementary Figure 2) to compare and cross-validate our daily wind directions and speeds, identifying the optimal wind directions for our transboundary air pollution analyses (Farsani et al., 2018; Naimabadi et al., 2018). Our trajectory analysis led us to exclude nine monitoring stations (four stations in Hong Kong and five stations in Shenzhen), which are located outside the range of 130° and 240° (southerly) from the Kwai Chung-Tsing Yi Container Terminals in Hong Kong and between the range of 0° and 65° (north-easterly) from Yantian port in Shenzhen.

We include a comprehensive set of covariates in our analysis that are theoretically and/or empirically relevant to our analysis (see Table 1). Specifically, we use five weather variables including daily mean temperature, daily relative humidity, daily total rainfall, daily prevailing wind direction, and daily mean wind speed to account for the effect of meteorological factors[2] on the transportation of ambient air pollutants (Bai et al., 2020; Fung & Wu, 2014; Luo et al., 2018). We collected this data from the Hong Kong Observatory and the Meteorological Bureau of the Shenzhen Municipal Government (HKSAR, 2021; MBSM, 2021). As marine vessel-induced SO_2 emissions are associated with maritime traffic and distances between the port and monitoring stations, we collected data from the Marine Department of the Hong Kong Government on the monthly number of ocean vessels arriving at reach port and distance variables (Deniz, Kilic, & Cıvkaroglu, 2010; Merico et al., 2021; Zabrocki, Leroutier, & Bind, 2021) (Supplementary Table 1). While the prevailing wind directions are southerly in summer and north-easterly in winter, wind direction and speed constantly change and exhibit irregular daily variations. We therefore use daily mean data for both SO_2 and our meteorological variables rather than seasonal mean data.

To test the robustness our findings, we also collected data on NO_2, O_3, CO, PM_{10}, and $PM_{2.5}$. We used the World Air Quality Project (AQICN, 2019) inventory and cross-validated the data with historical PRD Regional air quality monitoring reports from the Hong Kong Environmental Protection Department (EPD, 2021).

[2] We elected to exclude tropical cyclones, winter monsoon characteristics, and surface pressure as they are represented by the daily mean wind speed, prevailing wind direction, and rainfall variables.

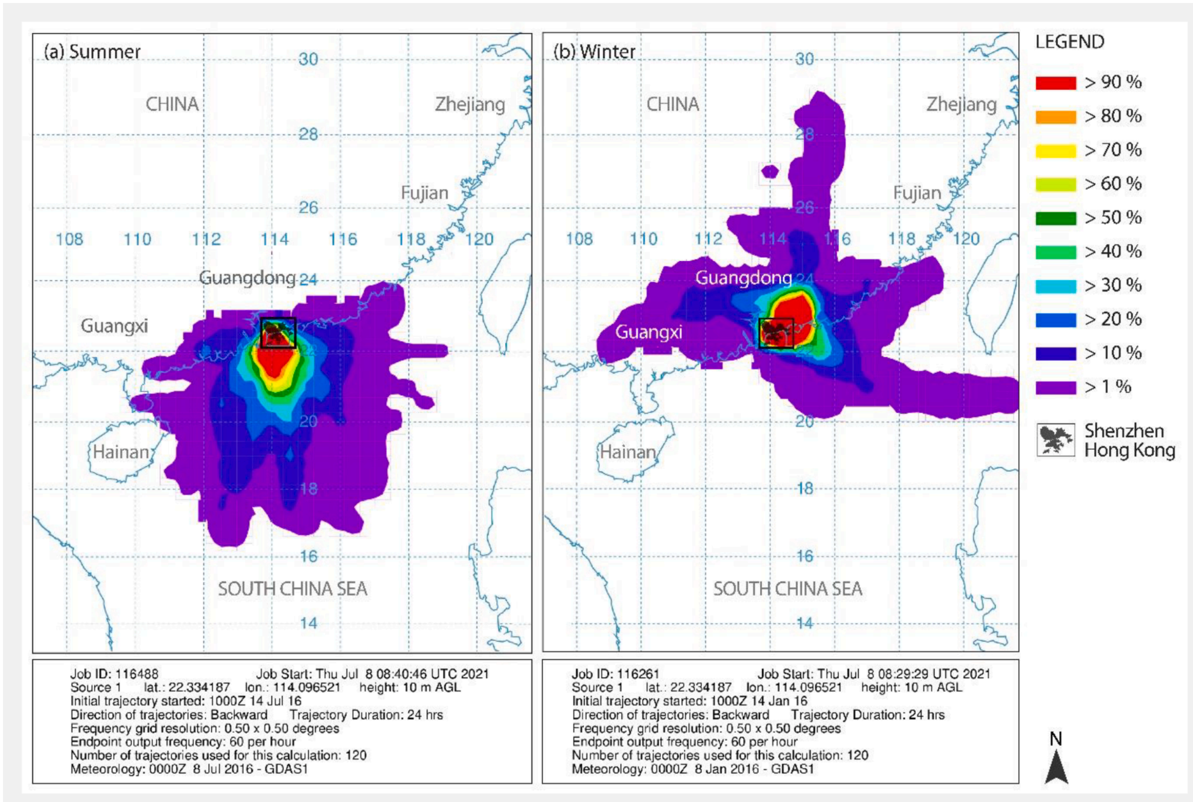

Fig. 2. Backward trajectory frequency map.
*Note: Model: NOAA HYSPLIT model (Stein et al., 2015). Meteorology data: Archived Global Data Assimilation System (GDAS - 1°, global, 2006-present). Number of trajectory starting location: 3 (residence time option 3). Vertical Motion: Model vertical velocity. Modelling periods: (a) June 13 - July 14, 2016; (b) December 13, 2015 – January 14, 2016. Level 1 height: 10 m atmospheric heights above ground level (AGL). Software: https://www.ready.noaa.gov/HYSPLIT_traj.php.

Table 1
Descriptive statistics.

Variable	Description	Mean	Median	Std. dev.	Min	Max	98 Percentile
SO_2	Sulphur dioxide (μg/m^3)	3.96	3.00	2.81	0.88	47.00	12.00
NO_2	Nitrogen dioxide (μg/m^3)	20.97	18.62	12.95	1.00	97.00	58.00
CO	Carbon monoxide (10 μg/m^3)	6.02	5.86	2.62	1.00	46.00	13.00
O_3	Ozone (μg/m^3)	29.67	26.00	18.90	1.00	167.00	82.00
PM_{10}	Respirable suspended particulates (μg/m^3)	36.49	34.00	17.70	1.00	150.00	80.02
$PM_{2.5}$	Fine suspended particulates (μg/m^3)	79.72	74.00	35.51	4.00	257.09	165.00
Temperature	Mean temperature (°C)	23.86	24.90	5.12	4.90	32.40	30.60
Humidity	Mean relative humidity (%)	78.48	79.00	10.29	29.00	99.00	95.00
Rainfall	Total rainfall (mm)	6.86	0.05	20.82	0.00	273.60	66.70
Wind Direction	Prevailing wind direction (degrees)	117.31	80.00	94.26	10.00	360.00	360.00
Wind Speed	Mean wind speed (km/h)	22.93	22.40	10.09	4.00	102.10	45.10
Distance HK	Distance (Hong Kong port – stations) (km)	21.36	19.74	19.32	1.05	88.94	88.94
Distance SZ	Distance (Shenzhen port – stations) (km)	28.72	28.97	19.22	1.89	106.79	106.79
Vessel	Number of ocean vessels (monthly)	2253.56	2239.93	243.71	1241.49	2760.00	2702.00

*Note: Pollutant levels are daily average values. Weather variables are daily averages. HK represents Hong Kong and SZ represents Shenzhen. Number of observations: 61,344.

3.2. *Estimation strategy*

We exploit the different timing of the implementation of domestic policies for SO_2 emissions from maritime vessels in Hong Kong and Shenzhen and use RDD[3] to estimate the effect of transboundary cooperation on air pollution. Specifically, we examine whether there are discontinuous changes in SO_2 concentrations at the point of implementation in each city, while accounting for wind direction. We estimate the following general specification:

$$\log(SO2_{it}) = \gamma_0 + \gamma_1 FV_{it} + \gamma_2 f(x_i) + \gamma_3 FV_{it} \times f(x_i) + \lambda X_{it} + \rho_i + \omega_t + \varepsilon_{it} \quad (1)$$

where $SO2_{it}$ represents the SO_2 concentration in location i at time t. FV_{it} is a dummy variable of the Fuel for Vessels (FV) policy. x_i is the forcing variable, $x_i = [t - FV_{it}]$ denotes the number of days since the FV policy was implemented (and thus x_i is 0 on FV implementation day), x_i is a negative value before implementation, and x_i is a positive value after implementation. f is a polynomial function with x_i as the independent

[3] RDD is a quasi-experimental statistical method that aims to measure causal effects of treatment by assigning thresholds before and after the relevant interventions.

variable. We use second- and fourth-order polynomials in this study. X_{it} is a set of control variables, including temperature, humidity, rain, wind speed, maritime traffic, and distances between ports and monitoring stations, which are used to control for the influence of a range of meteorological and emission factors on SO_2 concentration. ρ_i is the city fixed effect of location i, ω_t is time-fixed effects, and ε_{it} represents the error term. In Eq. (1), the main coefficient of concern is γ_1, which captures the difference in SO_2 concentrations before and after the FV policy implementation.

4. Results

4.1. Regression discontinuity design estimations

We report the main estimates of our RDD analysis in Table 2. In Column 1, we estimate the effects of the FV policy in Hong Kong on SO_2 when the wind is blowing from south to north. Column 2 contains estimates for the same effects when the wind is blowing from north to south. Columns 3 (wind blowing from south to north) and 4 (wind blowing from north to south) contain estimates for Shenzhen's FV policy, which was implemented 15 months after Hong Kong's policy.

Our findings in Column 1 suggest that Hong Kong's FV policy is systematically associated with a 12.9%[4] reduction in SO_2 concentration levels in Hong Kong and Shenzhen on southerly wind days at the 5% significance level. As seen in Column 2, we find that SO_2 concentrations in Hong Kong increased during northerly wind days after the implementation of Hong Kong's FV policy. Our finding is statistically significant at the 5% level. Unsurprisingly, in Column 3 we find that the implementation of Shenzhen's fuel control policy had no statistically significant effect on SO_2 concentrations in Hong Kong on southerly wind days. However, as seen in Column 4, we find that the implementation of Shenzhen's FV policy is systematically associated with a 31.2% decrease in SO_2 concentrations in Hong Kong on northerly wind days at the 1% significance level. Thus, the results from all models in Table 2 provide evidence in support of the ability for FV regulations to lower transboundary air pollution.

Fig. 3 illustrates our findings graphically using a fourth-order polynomial plot to account for the 30-month bandwidths used in our analysis. Our graphical representation reiterates our two main findings. First, starting with panels (1) and (4), we see an immediate reduction in SO_2 concentrations in the study area. This reduction occurred immediately after the implementation of Hong Kong and Shenzhen's local FV policies, suggesting a causal association. Second, our graphical representation suggests that SO_2 concentrations in Hong Kong were affected by SO_2 emissions from Shenzhen. This is represented in panel 2 (Fig. 3), which depicts how SO_2 concentrations increased in Hong Kong even after it implemented its FV policy due to northerly winds, which continued to transport unregulated SO_2 from Shenzhen. To test whether our model is overfitted, we generated a second-order polynomial plot (supplementary figure 3). The two polynomial plots reveal the same pattern and discontinuity at the break points, confirming that our fourth-polynomial plot is not an overfitting model.

Table 2
Regression discontinuity design estimations.

	(1) July 1, 2015 (Wind: S → N)	(2) July 1,2015 (Wind: N → S)	(3) October 1, 2016 (Wind: S → N)	(4) October 1, 2016 (Wind: N → S)
$FV_{it} = 1$	−0.122**	0.122**	0.127	−0.272***
	(0.050)	(0.062)	(0.077)	(0.065)
Temperature	−0.027***	−0.008***	−0.016**	−0.010***
	(0.008)	(0.002)	(0.008)	(0.003)
Humidity	−0.010***	−0.012***	−0.013***	−0.013***
	(0.002)	(0.001)	(0.002)	(0.001)
Rainfall	0.001	−0.002	0.006	0.009
	(0.003)	(0.007)	(0.004)	(0.010)
Wind Speed	−0.007***	−0.011***	−0.010***	−0.013***
	(0.001)	(0.001)	(0.001)	(0.001)
Distance HK	0.001	−0.071***	−0.001	−0.073***
	(0.001)	(0.009)	(0.006)	(0.008)
Vessel	0.004***	0.002***	0.001**	0.010**
	(0.001)	(0.001)	(0.001)	(0.004)
Fixed effects	Yes	Yes	Yes	Yes
Constant	2.991***	2.200***	2.607***	3.131***
	(0.535)	(0.147)	(0.489)	(0.187)
R-squared	0.288	0.355	0.399	0.416
Observations	904	1446	906	1306

*Note: The dependant variable is daily logged SO_2. Standard errors (robust to heteroskedasticity) in parentheses. All regressions include year and month dummies to control for time-specific fixed effects. Quadratic time trends are included in all model specifications. A bandwidth of 912 days is used to the left and right of the cut-offs. The prevailing wind directions in each model are based on wind rose plots in Supplementary Figure 2. Non-logged specification is in Supplementary Table 3. ***$p < 0.01$. **$p < 0.05$. *$p < 0.10$.

4.2. Robustness checks

To test the validity of our main findings, we conducted three separate robustness checks. First, we conducted a mixed effects ANOVA test to check for differences in covariates and their interaction effects between Hong Kong and Shenzhen. Second, we changed the bandwidth around the policy intervention to 450 days instead of the 912 days we used in our main analysis. We did this to test for systematic bias related to unobserved factors such as other local policies that may have influenced local pollutant concentration levels. Third, we conducted a placebo test with different wind patterns. We did this to test whether our results may be biased due to unobserved time-variant factors. Finally, we re-estimated the impact of the FV policies on other major pollutants, including NO_2, O_3, CO, PM_{10}, and $PM_{2.5}$.

4.2.1. Interaction effects

Based on our mixed effects ANOVA test, we estimate RDD with the covariates and interaction variables that are statistically significant at the 5% level. From our main estimations (Table 2), we exclude rainfall and added the interaction variables of temperature, rainfall, wind speed, and number of ocean vessels. The results are broadly the same, with slightly increased R-squared values (Table 3).

4.2.2. Variation in bandwidth

In our main analysis, we used a 912-day bandwidth around each policy intervention. To account for the possibility that unobserved factors such as other local policies directly or indirectly influenced the local SO_2 concentration levels, we test for any systematic biases by selecting a smaller bandwidth of 450 days. While the selection of a significantly smaller bandwidth is likely to reduce precision, it is also likely to identify any systematic biases arising from unobserved factors (Cattaneo, Titiunik, & Vazquez-Bare, 2020). Our findings are presented in Table 4. As can be seen, our results hold in terms of the direction of the sign and statistical significance.

4.2.3. Placebo test

Our main analytical model is premised on using two daily prevailing wind directions to identify the treatment effect of transboundary air pollutants. To account for the possibility that our results may be systematically biased by unobserved time-variant factors or omitted meteorological variables, we run a placebo test that uses easterly and westerly winds instead of northerly and southerly winds. As Table 5 shows, our findings for the placebo test are not statistically significant at

[4] As the dependent variable is log-transformed, the effect of one unit change in an independent variable is equal to $\exp(\beta)-1$. Hence, the effect of the Fuel for Vessels policy on SO_2 is $\exp(0.122)-1 \approx 12.9\%$.

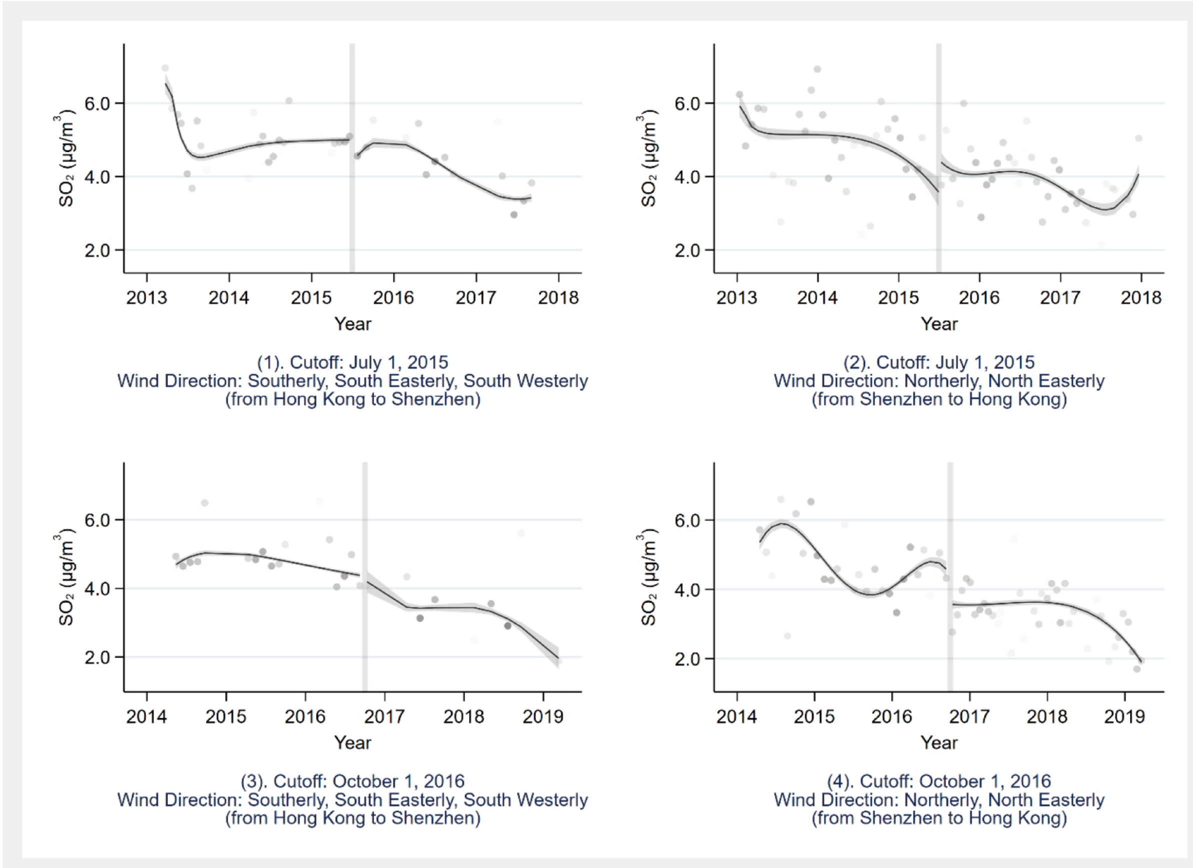

Fig. 3. Fourth-order polynomial plot of regression discontinuity design estimations.
*Note: Grey vertical lines indicate the effective dates of marine fuel vessel regulations in Hong Kong (July 1, 2015) and Shenzhen (October 1, 2016), respectively. Solid grey circles are the binned observations. Grey areas along with the polynomial lines are the 95% confidence bands.

Table 3
Regression discontinuity design estimations with interaction effects.

	(1) July 1, 2015 (Wind: S → N)	(2) July 1, 2015 (Wind: N → S)	(3) October 1, 2016 (Wind: S → N)	(4) October 1, 2016 (Wind: N → S)
$FV_{it} = 1$	−0.102**	0.123**	0.131	−0.284***
	(0.050)	(0.061)	(0.084)	(0.066)
Other variables	Yes	Yes	Yes	Yes
Fixed effects	Yes	Yes	Yes	Yes
Constant	2.300***	2.675***	4.221***	3.021
	(0.612)	(0.231)	(0.605)	(0.230)
R-squared	0.295	0.369	0.407	0.427
Observations	908	1442	904	1304

*Note: The dependent variable is daily logged SO_2. Standard errors (robust to heteroskedasticity) in parentheses. All regressions include year and month dummies to control for time-specific fixed effects. Other variables include temperature, humidity, wind speed, distance from major ports, number of vessels, and city interaction terms with temperature, rainfall, wind speed, and number of ocean vessels. Quadratic time trends are included in all model specifications. A bandwidth of 912 days is used to the left and right of the cut-offs. ***$p < 0.01$. **$p < 0.05$. *$p < 0.10$.

the 5% level, which provides further assurance that our main results presented in Table 2 are robust. A plausible reason that the coefficient in column (4) in Table 5 is statistically significant at the 10% level may be the prevalence of an unusual easterly wind carrying SO_2 from other regions (Supplementary Figure 5).

Table 4
Regression discontinuity design estimations with reduced bandwidth.

	(1) July 1, 2015 (Wind: S → N)	(2) July 1, 2015 (Wind: N → S)	(3) October 1, 2016 (Wind: S → N)	(4) October 1, 2016 (Wind: N → S)
$FV_{it} = 1$	−0.148**	0.103**	0.151	−0.309***
	(0.058)	(0.127)	(0.114)	(0.088)
Other variables	Yes	Yes	Yes	Yes
Fixed effects	Yes	Yes	Yes	Yes
Constant	2.300***	2.675***	4.221***	3.021
	(0.612)	(0.231)	(0.605)	(0.230)
R-squared	0.186	0.266	0.380	0.340
Observations	584	696	424	686

*Note: The dependent variable is daily logged SO_2. Standard errors (robust to heteroskedasticity) in parentheses. All regressions include year and month dummies to control for time-specific fixed effects. Quadratic time trends are included in all model specifications. A bandwidth of 450 days is used to the left and right of the cut-offs. ***$p < 0.01$. **$p < 0.05$. *$p < 0.10$.

4.2.4. Other major pollutants

As Hong Kong and Shenzhen's FV regulatory policies are limited to sulphur content, their effects should be restricted to changes to SO_2 concentrations (EPD, 2019). To verify whether changes in SO_2 concentrations are a direct result of the FV policy or instead the result of, for example, unobserved local environmental regulations or omitted variables, we re-run our estimations using five pollutants that are not directly related: NO_2, O_3, CO, PM_{10}, and $PM_{2.5.}$ We present our results in Table 6. As can be seen, CO and $PM_{2.5}$ concentrations recorded an

Table 5
Regression discontinuity design estimations with different wind directions.

	(1) July 1, 2015 (Wind: W → E)	(2) July 1, 2015 (Wind: E → W)	(3) October 1, 2016 (Wind: W → E)	(4) October 1, 2016 (Wind: E → W)
$FV_{it} = 1$	−0.036	0.017	0.136	−0.127*
	(0.108)	(0.073)	(0.697)	(0.071)
Other variables	Yes	Yes	Yes	Yes
Fixed effects	Yes	Yes	Yes	Yes
Constant	0.018	3.240***	0.826	4.076***
	(0.754)	(0.220)	(0.778)	(0.258)
R-squared	0.340	0.398	0.411	0.425
Observations	418	1020	408	1016

*Note: The dependant variable is daily logged SO_2. Standard errors (robust to heteroskedasticity) in parentheses. All regressions include year and month dummies to control for time-specific fixed effects. Quadratic time trends are included in all model specifications. A bandwidth of 450 days is used to the left and right of the cut-offs. ***$p < 0.01$. **$p < 0.05$. *$p < 0.10$.

Table 6
Fuel for vessels policy's transboundary effects on NO_2, O_3, CO, PM_{10}, and $PM_{2.5}$.

Dependant variable	(1) July 1, 2015 (Wind: S → N)	(2) July 1, 2015 (Wind: N → S)	(3) October 1, 2016 (Wind: S → N)	(4) October 1, 2016 (Wind: N → S)
NO_2	0.034	−0.020	−0.674**	−0.068
	(0.043)	(0.053)	(0.341)	(0.057)
O_3	−0.028	−0.052	−0.548*	0.001
	(0.080)	(0.069)	(0.328)	(0.084)
CO	0.076**	−0.014	0.014	0.115***
	(0.037)	(0.039)	(0.127)	(0.044)
PM_{10}	0.026	−0.170***	−0.692*	0.079
	(0.056)	(0.052)	(0.373)	(0.057)
$PM_{2.5}$	0.136***	−0.053	−0.993*	0.151***
	(0.053)	(0.049)	(0.543)	(0.053)

*Note: The table reports estimates from 20 separate regressions. The dependant variable is the logged pollution level. The reported coefficients correspond to 1 (Fuel for Vessels), an indicator variable equal to one after July 1, 2015 in columns (1) and (2) and October 1, 2016 in columns (3) and (4), respectively. Standard errors (robust to heteroskedasticity) are in parentheses. All regressions include daily temperature, humidity, rainfall, wind speed, distances between ports and monitoring stations, the number of ocean vessels (monthly), as well as year and month dummies to control for time-specific fixed effects. Quadratic time trends are included in all model specifications. A bandwidth of 912 days is used to the left and right of the cut-offs. ***$p < 0.01$. **$p < 0.05$. *$p < 0.10$.

increase after the policy interventions in Hong Kong in 2015 and in Shenzhen in 2016. We suggest that this may be due to the installation of scrubber systems on ships for sulphur abatement. While they help to lower SO_2 emissions, they use significant amounts of energy, generate carbon emissions, and have little effect on reducing particulate matter (PM) (Abadie, Goicoechea, & Galarraga, 2017; Comer, Georgeff, & Osipova, 2020).

5. Discussion

We exploit a natural experiment between two cities with different administrative systems and customs borders to isolate the effect of transboundary cooperation on marine vessel emissions. We find that when Hong Kong implemented a legally binding regulatory policy to limit SO_2 emissions from marine vessels, SO_2 concentrations decreased when the wind direction was southerly. When the wind direction was northerly, however, SO_2 concentrations increased. This is consistent with marine vessel emissions travelling south to Hong Kong from Shenzhen. When Shenzhen implemented a corresponding legally binding regulatory policy to limit SO_2 emissions from marine vessels in its jurisdiction, SO_2 concentrations for Hong Kong reduced by as much as 32.8% (1.42 $\mu g/m^3$) when the wind direction was northerly in our long-term window (912 days before and after the policy implementation). With 95% confidence intervals, the effects range between 16.7% (0.72 $\mu g/m^3$) and 51.1% (2.21 $\mu g/m^3$). We find that the reduction effect is 3.4 percentage points higher (36.2%) in the shorter-term window with the 450-day bandwidth.

Our findings are highly statistically significant and robust. Importantly, our use of the regression discontinuity design allows for causal inference. With respect to the spatial extent of the SO_2 reduction effect, the FV policy also influences other neighbouring cities in the PRD since 90% of wind frequency covers more than half of Dongguan and Huizhou (neighbouring cities to the north and northeast of Shenzhen, respectively) in the HYSPLIT forward trajectory frequency map (Supplementary Figure 4). However, wind speed can affect the spatial extent (Liu, Zhou, & Lu, 2020). Since our study area has a relatively high wind speed (mean = 22.9 km/h, min = 4 km/h), the respective FV policy influences SO_2 concentration in both cities in under 24 h. In contrast, local FV policies may be less effective in regions where average wind speeds are relatively low, such as Thailand (10 km/h) and Malaysia (6.4 km/h).

Our findings support existing studies concluding that pollution sources transported from adjacent regions can affect average air pollution levels (Cheung, He, & Pan, 2020; Chuang et al. (2020). Furthermore, our results confirm Lu et al. (2020)'s argument that many coastal cities are unable to reduce air pollutants through domestic policy and regulations alone due to the transboundary nature of air pollution. In our reading of the literature, however, we find a lack of empirical data quantifying the impact of transboundary cooperation on air pollution outcomes. We therefore make a notable empirical contribution by employing an identification strategy that allows us to isolate the treatment effect of transboundary cooperation on air quality outcomes. Our findings suggest that while local policies regulating transboundary pollution can be effective regardless of neighbouring regions' actions (or lack thereof), they are not sufficient. By themselves, domestic policies and regulations have limited capacity to improve local air quality due to the transboundary nature of air pollution (Lu et al., 2020; Yamineva & Romppanen, 2017). Many existing transboundary air pollution policies are also non-binding and rudimentary due to various technical, economic, and institutional asymmetries (Lee & Paik, 2020). This suggests that tackling transboundary environmental challenges in the long term may require a more fundamental reduction in cross-border disparities. In the meantime, however, our results demonstrate that cooperative policies can significantly improve air quality across wider spaces.

Our findings have two main implications. First, we present some of the first causal evidence for the effectiveness of cooperative, legally binding regulations on air pollution outcomes. Second, we suggest that in some cases, transboundary air pollution may be best regulated at the level of city dyads. It stands to reason that cross-border cities are likely to recognise their mutual vulnerability to transboundary air pollution and are thus likely to be similarly motivated to tackle transboundary air pollution by enacting comparable and legally binding domestic regulations. While this may well be a complex task, especially when cities differ socio-economically or belong to different national and political jurisdictions, we argue that cooperation at this scale is worth pursuing, especially because municipalities can generally act more quickly than international organizations.

6. Conclusion

Existing studies lack quantitative data regarding the impact of transboundary cooperation on air pollution outcomes. Our study fills this gap by exploiting the staggered implementation of marine vessel emission standards in the unique administrative settings in Hong Kong and Shenzhen. This research is novel in three respects. First, using the RDD framework, we estimate the causal effects of transboundary atmospheric cooperation by analysing legally binding regulations in neighbouring cities. Second, we provide additional evidence by carrying out robustness checks, which include interaction variables, different bandwidths, various wind directions, and other air pollutants to control for unobserved time-variant factors and other possible confounding factors. Third, we cross-check daily wind direction and speed with the HYSPLIT back trajectory model to ensure accurate modelling of the complex transportation and dispersion of ambient air pollutants. Ultimately, we find that the implementation of Shenzhen's FV policy is systematically associated with a decrease in SO_2 concentrations in Hong Kong of 32.8% (1.42 μg/m^3) in the long term and 36.2% (1.47 μg/m^3) in the short term on northerly wind days. Our findings suggest that transboundary air pollution may be best controlled at the level of city dyads. Neighbouring, cross-border cities can enhance local air pollution by enacting comparable legally binding regulations.

Data availability

The data and code required to replicate our study are available at https://figshare.com/s/c87b97e9f8bb4bd18a7e.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

Supplementary material associated with this article can be found in the online version at doi:10.1016/j.scs.2022.103774.

References

Abadie, L. M., Goicoechea, N., & Galarraga, I. (2017). Adapting the shipping sector to stricter emissions regulations: Fuel switching or installing a scrubber? *Transportation Research Part D: Transport and Environment, 57*, 237–250.

Alvarez-Herranz, A., Balsalobre-Lorente, D., Shahbaz, M., & Cantos, J. M. (2017). Energy innovation and renewable energy consumption in the correction of air pollution levels. *Energy Policy, 105*, 386–397.

An, R., Zhang, S., Ji, M., & Guan, C. (2018). Impact of ambient air pollution on physical activity among adults: A systematic review and meta-analysis. *Perspectives in Public Health, 138*(2), 111–121.

AQICN. (2019). *Air quality historical data platform*. Retrieved from https://aqicn.org/data-platform.

Bai, S., Wen, Y., He, L., Liu, Y., Zhang, Y., Yu, Q., et al. (2020). Single-vessel plume dispersion simulation: method and a case study using CALPUFF in the Yantian Port Area, Shenzhen (China). *International Journal of Environmental Research and Public Health, 17*(21), 1–29. https://doi.org/10.3390/ijerph17217831, 7831.

Butt, E., Turnock, S., Rigby, R., Reddington, C., Yoshioka, M., Johnson, J., , … Spracklen, D, et al. (2017). Global and regional trends in particulate air pollution and attributable health burden over the past 50 years. *Environmental Research Letters, 12*(10), 1–11. https://doi.org/10.1088/1748-9326/aa87be, 104017.

Cai, H., Nan, Y., Zhao, Y., & Xiao, H. (2021). Impacts of winter heating on the population mobility of northern China's prefectural cities: Evidence from a regression discontinuity design. *Kybernetes*, 1–25. https://doi.org/10.1108/K-07-2020-0480

Cattaneo, M. D., Titiunik, R., & Vazquez-Bare, G. (2020). The regression discontinuity design. In L. Curini (Ed.), *The Sage handbook of research methods in political science and international relations* (pp. 835–857). Thousand Oaks, California: Sage Publications.

Cheung, C. W., He, G., & Pan, Y. (2020). Mitigating the air pollution effect? The remarkable decline in the pollution-mortality relationship in Hong Kong. *Journal of Environmental Economics and Management, 101*, 1–19. https://doi.org/10.1016/j.jeem.2020.102316, 102316.

Chuang, M.-T., Ooi, M. C. G., Lin, N.-H., Fu, J. S., Lee, C.-T., Wang, S.-H., et al. (2020). Study on the impact of three Asian industrial regions on PM 2.5 in Taiwan and the process analysis during transport. *Atmospheric Chemistry and Physics, 20*(23), 14947–14967.

Comer, B., Georgeff, E., & Osipova, L. (2020). *Air emissions and water pollution discharges from ships with scrubbers*. International Council on Clean Transportation website. Retrieved from the https://theicct.org/publications/air-water-pollution-scrubbers-2020.

Davis, L. W. (2008). The effect of driving restrictions on air quality in Mexico City. *Journal of Political Economy, 116*(1), 38–81.

Delfino, R. J., Sioutas, C., & Malik, S. (2005). Potential role of ultrafine particles in associations between airborne particle mass and cardiovascular health. *Environmental Health Perspectives, 113*(8), 934–946.

Deniz, C., Kilic, A., & Cıvkaroglu, G. (2010). Estimation of shipping emissions in Candarli Gulf, Turkey. *Environmental Monitoring and Assessment, 171*(1), 219–228.

Dianat, M., Radmanesh, E., Badavi, M., Mard, S. A., & Goudarzi, G. (2016). Disturbance effects of PM10 on iNOS and eNOS mRNA expression levels and antioxidant activity induced by ischemia–reperfusion injury in isolated rat heart: Protective role of vanillic acid. *Environmental Science and Pollution Research, 23*(6), 5154–5165.

Dockery, D. W., Rich, D. Q., Goodman, P. G., Clancy, L., Ohman-Strickland, P., George, P., et al. (2013). Effect of air pollution control on mortality and hospital admissions in Ireland. *Health Effects Institute, 176*, 3–109.

Eades, L. (2018). Air pollution at the US-Mexico border: Strengthening the framework for bilateral cooperation. *Journal of Public & International Affairs, 64*, 64–79. Retrieved from https://jpia.princeton.edu/sites/jpia/files/media/jpia_2018.pdf#page=64.

EPD. (2019). A guide to the air pollution control (fuel for vessels) regulation. Retrieved from https://www.epd.gov.hk/epd/english/environmentinhk/air/prob_solutions/guide-air-pollution-control-fuel-for-vessels-regulation.html.

EPD. (2021). *Pearl River Delta regional air quality monitoring report*. Retrieved from Hong Kong: Https://www.epd.gov.hk/epd/english/resources_pub/publications/m_report_archives.html.

Farsani, M. H., Shirmardi, M., Alavi, N., Maleki, H., Sorooshian, A., Babaei, A., et al. (2018). Evaluation of the relationship between PM10 concentrations and heavy metals during normal and dusty days in Ahvaz, Iran. *Aeolian Research, 33*, 12–22.

Friedrich, A., Heinen, F., Kamakaté, F., & Kodjak, D. (2007). Air pollution and greenhouse gas emissions from ocean-going ships: Impacts, mitigation options and opportunities for managing growth. *Air pollution and greenhouse gas emissions from ocean-going ships: Impacts, mitigation options and opportunities for managing growth*. Washington D.C.: The International Council on Clean Transportation. Retrieved from https://theicct.org/sites/default/files/publications/oceangoing_ships_2007.pdf.

Fu, S., & Gu, Y. (2017). Highway toll and air pollution: Evidence from Chinese cities. *Journal of Environmental Economics and Management, 83*, 32–49.

Fung, F., Zhu, Z., Becque, R., & Finamore, B. (2014). *Prevention and control of shipping and port air emissions in China*. Natural Resources Defence Council (NRDC).

Fung, W. Y., & Wu, R. (2014). Relationship between intraseasonal variations of air pollution and meteorological variables in Hong Kong. *Annals of GIS, 20*(3), 217–226.

Gu, Y., & Yim, S. (2016). The air quality and health impacts of domestic trans-boundary pollution in various regions of China. *Environment International, 97*, 117–124.

HKO. (2021,. 24 January 2020). Backward Trajectory Map. Retrieved from https://www.hko.gov.hk/en/wxinfo/trajectory/traj_example.htm.

HKPD. (2002). *Guangdong and Hong Kong cross-boundary co-operation on environmental issues*. Hong Kong. Retrieved from https://www.pland.gov.hk/pland_en/p_study/comp_s/hk2030/eng/wpapers/pdf/InformationNote_05.pdf.

HKSAR. (2021). *Weather and radiation level report*. Retrieved from https://data.gov.hk/en-data/dataset/hk-hko-rss-weather-and-radiation-level-report.

Hopkinson, L., & Stern, R. (2003). One country, two systems, one smog cross-boundary air pollution policy challenges for Hong Kong and Guangdong. *China Environment Series, 6*(6), 19–36.

IMO. (2021). IMO 2020 – cutting sulphur oxide emissions. Retrieved from https://www.imo.org/en/MediaCentre/HotTopics/Pages/Sulphur-2020.aspx.

Jung, J., Choi, A., & Yoon, S. (2021). Transboundary air pollution and health: Evidence from East Asia. *Environment and Development Economics*, 1–25. https://doi.org/10.1017/S1355770X21000115

Kalayci, C. (2019). The impact of economic globalization on CO2 emissions: The case of NAFTA countries. *International Journal of Energy Economics and Policy, 9*(1), 356–360. https://doi.org/10.32479/ijeep.7233

Koolen, C. D., & Rothenberg, G. (2019). Air pollution in Europe. *ChemSusChem, 12*(1), 164–172. https://doi.org/10.1002/cssc.201802292

Kwok, R. H., Fung, J. C., Lau, A. K., & Fu, J. S. (2010). Numerical study on seasonal variations of gaseous pollutants and particulate matters in Hong Kong and Pearl River Delta Region. *Journal of Geophysical Research: Atmospheres, 115*(D16), 1–23. https://doi.org/10.1029/2009JD012809

Lai, H.-K., Tsang, H., Chau, J., Lee, C., McGhee, S., Hedley, A., et al. (2013). Health impact assessment of marine emissions in Pearl River Delta region. *Marine Pollution Bulletin, 66*(1–2), 158–163.

Lee, T., & Paik, W. (2020). Asymmetric barriers in atmospheric politics of transboundary air pollution: A case of particulate matter (PM) cooperation between China and South Korea. *International Environmental Agreements: Politics, Law and Economics, 20*(1), 123–140.

Lee, Y., & Savtchenko, A. (2006). Relationship between air pollution in Hong Kong and in the Pearl River Delta Region of South China in 2003 and 2004: An analysis. *Journal of Applied Meteorology and Climatology, 45*(2), 269–282.

Liu, Y., Zhou, Y., & Lu, J. (2020). Exploring the relationship between air pollution and meteorological conditions in China under environmental governance. *Scientific Reports, 10*(1), 1–11.

Loh, C., & Booth, V. (2012). The fair winds charter - A model PPP in Hong Kong. *European Procurement and Public Private Partnership Law Review, 7*, 129–137.

Lu, X., Zhang, S., Xing, J., Wang, Y., Chen, W., Ding, D., et al. (2020). Progress of air pollution control in China and its challenges and opportunities in the ecological civilization era. *Engineering, 6*(12), 1423–1431. https://doi.org/10.1016/j.eng.2020.03.014

Luo, M., Hou, X., Gu, Y., Lau, N.-C., & Yim, S. H.-L. (2018). Trans-boundary air pollution in a city under various atmospheric conditions. *Science of The Total Environment, 618*, 132–141. https://doi.org/10.1016/j.scitotenv.2017.11.001

Mao, X., Cui, H., Roy, B., Olmer, N., Rutherford, D., & Comer, B. (2017). Distribution of air pollution from oceangoing vessels in the Greater Pearl River Delta, 2015. *The International Council on Clean Transportation, 10*, 1–8.

Mason, T. G., Chan, K. P., Schooling, C. M., Sun, S., Yang, A., Yang, Y., et al. (2019). Air quality changes after Hong Kong shipping emission policy: An accountability study. *Chemosphere, 226*, 616–624.

MBSM. (2021). *Shenzhen weather*. Retrieved from http://weather.sz.gov.cn/en/index.html.

Merico, E., Cesari, D., Gregoris, E., Gambaro, A., Cordella, M., & Contini, D. (2021). Shipping and air quality in Italian port cities: State-of-the-art analysis of available results of estimated impacts. *Atmosphere, 12*(5), 1–22. https://doi.org/10.3390/atmos12050536, 536.

Moses, E.C., .Beatriz &; Seddon, J. (2020). The most successful air pollution treaty you've never heard of. Retrieved from https://www.wri.org/blog/2020/02/most-successful-air-pollution-treaty-youve-never-heard-of.

Naimabadi, A., Shirmardi, M., Maleki, H., Teymouri, P., Goudarzi, G., Shahsavani, A., et al. (2018). On the chemical nature of precipitation in a populated Middle Eastern Region (Ahvaz, Iran) with diverse sources. *Ecotoxicology and Environmental Safety, 163*, 558–566.

Neidell, M. (2009). Information, avoidance behavior, and health the effect of ozone on asthma hospitalizations. *Journal of Human Resources, 44*(2), 450–478.

NEPIA. (2020). China: Fuel sulphur cap and emission control areas. Retrieved from https://www.nepia.com/industry-news/china-fuel-sulphur-cap-and-emission-control-areas.

Ng, S. K. W. (2016). Controlling ship emissions in the pear river delta. *Controlling ship emissions in the pear river delta* (pp. 1–16). Hong Kong: Civic Exchange.

Raqmn. (2018). *Guangdong-Hong kong-Macao Pearl River Delta Regional Air Quality Monitoring Network: Statistical summary of the first quarter monitoring results (January to March 2018)* (pp. 1–22). Hong Kong: Guangdong-Hong Kong-Macao Pearl River Delta Regional Air Quality Monitoring Network. PRDAIR-2018-1. Retrieved from https://www.epd.gov.hk/epd/sites/default/files/epd/english/resources_pub/publications/files/PRD2018_1-en-1.pdf.

Reis, S., Liška, T., Vieno, M., Carnell, E. J., Beck, R., Clemens, T., et al. (2018). The influence of residential and workday population mobility on exposure to air pollution in the UK. *Environment International, 121*, 803–813.

Smith, T., Jalkanen, J., Anderson, B., Corbett, J., Faber, J., Hanayama, S. et al. (2015). Third IMO greenhouse gas study 2014.

Soleimani, Z., Parhizgari, N., Dehdari Rad, H., Akhoond, M. R., Kermani, M., Marzouni, M. B., et al. (2015). Normal and dusty days comparison of culturable indoor airborne bacteria in Ahvaz, Iran. *Aerobiologia, 31*(2), 127–141.

Stallings-Smith, S., Zeka, A., Goodman, P., Kabir, Z., & Clancy, L. (2013). Reductions in cardiovascular, cerebrovascular, and respiratory mortality following the national Irish smoking ban: Interrupted time-series analysis. *PloS One, 8*(4), 1–7. https://doi.org/10.1371/journal.pone.0062063. e62063.

Stein, A., Draxler, R. R., Rolph, G. D., Stunder, B. J., Cohen, M., & Ngan, F. (2015). NOAA's HYSPLIT atmospheric transport and dispersion modeling system. *Bulletin of the American Meteorological Society, 96*(12), 2059–2077.

Topali, D., & Psaraftis, H. N. (2019). The enforcement of the global sulfur cap in maritime transport. *Maritime Business Review, 4*(2). https://doi.org/10.1108/MABR-12-2018-0050

Tsai, W.-t., & Chou, Y.-. H. (2005). Overview of environmental impacts, prospects and policies for renewable energy in Taiwan. *Renewable and Sustainable Energy Reviews, 9*(2), 119–147.

Viard, V. B., & Fu, S. (2015). The effect of Beijing's driving restrictions on pollution and economic activity. *Journal of Public Economics, 125*, 98–115.

Wong, M. (2018). Shenzhen and Hong Kong show how intercity alliances can tackle pollution. *South China Morning Post*. Retrieved from https://www.scmp.com/comment/insight-opinion/article/2132586/shenzhen-and-hong-kong-show-how-intercity-alliances-can.

WSC. (2021). Top 50 world container ports. Retrieved from https://www.worldshipping.org/about-the-industry/global-trade/top-50-world-container-ports.

Yamineva, Y., & Romppanen, S. (2017). Is law failing to address air pollution? Reflections on international and EU developments. *Review of European, Comparative & International Environmental Law, 26*(3), 189–200.

Zabrocki, L., Leroutier, M., & Bind, M.-A. (2021). Estimating the causal effects of cruise traffic on air pollution using randomization-based Inference. *arXiv preprint* arXiv: 2105.03996.

Zhang, Q., Jiang, X., Tong, D., Davis, S. J., Zhao, H., Geng, G., et al. (2017). Transboundary health impacts of transported global air pollution and international trade. *Nature, 543*(7647), 705–709. https://doi.org/10.1038/nature21712