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Costs of air pollutants from shipping: a meta-regression analysis

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ABSTRACT

This study estimated the external cost of air pollution from shipping by means of a meta-regression analysis, which has not been made before. Three pollutants, which were included in most of the primary studies, were considered: nitrogen oxides (NO_x), sulphur dioxides (SO_2) and particulate matters with a diameter of max 2.5 micrometres ($PM_{2.5}$). All primary studies included damages of health and a majority added impacts on agriculture and estimated the cost of air pollutants by transferring cost estimates from studies on costs of air emissions from transports in Europe. Different regression models and estimators were used and robust results were found of statistically significant emission elasticities of below one, i.e. total external costs increase by less than 1% when emissions increase by 1%. There was a small variation between the pollutants, with the highest elasticity for $PM_{2.5}$ and lowest for NO_x. Calculations of the marginal external cost of the pollutants showed the same pattern, with this cost being approximately six times higher for $PM_{2.5}$ than for the other pollutants. Common to all pollutants was that the marginal external cost decreases when emission increases. Another robust result was a significant increase in the cost of studies published in journals compared with other publication outlets. These findings point out some caution when transferring constant external unit cost of air pollutant from shipping, which is much applied in the literature, and the cost functions estimated in this study could thus provide a complementary transfer mechanism.

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Shipping; air pollution; costs; meta-regression analysis

1. Introduction

Shipping is an important mode of transport in international trade and accounts for 70% of the total value of global trade (United Nations Conference on Trade and Development [UNCTAD], 2018). Shipping causes environmental and ecological impacts through discharges to water, physical impacts and air emissions (Jägerbrand, Brutemark, Barthel

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Svedén, & Gren, 2019). The emissions of sulphur oxides (SO_x) and nitrogen oxides (NO_x) cause acidification and contribute to nutrient enrichment of seas and waters, and particulates are a source of health issues for humans (Barregard, Molnad, Jonson, & Stockfelt, 2019; Endresen, Lee Behrens, Brynestad, Bjørn Andersen, & Skjong, 2004; Eyring et al., 2010; Salo et al., 2016; Sofiev et al., 2018). Total shipping contributes to about 3% of annual global CO_2 emissions and to about 13% and 15% of anthropogenic emissions of SO_x and NO_x, respectively (Smith et al., 2015). Trade in goods with maritime transport is expected to grow by 3.8% per annum between 2018 and 2023 (UNCTAD, 2018). Similarly, economic growth is regarded as a source of growing demand for cruise and passenger transport (e.g. Wang, Wang, Zhen, & Qu, 2016).

In order to evaluate the costs and benefits of shipping to society, it is necessary to consider all costs, including the external costs of air pollutants. Similar to air pollutants from other emission sources such as transport on land and energy production, challenges arise in how to quantify this damage and assess it in monetary terms. The damage depends on the spatial and dynamic dispersal of pollutants from the emission source. The estimation in monetary terms requires information on people's perceived losses caused by these impacts. Calculations of external costs, therefore, require an integrated assessment in which different models of pollutant pathways, impacts and valuation in monetary terms are combined. Such assessments have been made for emissions of NO_x, SO_2 and particle matter (PM) from transports since the early 1980s, particularly for Europe (see Quinet (2004) for a review).

The calculation of external costs of air pollution from shipping is more recent, with Miola et al. (2009) providing an early contribution. Today there is a considerable number of studies involving different types of pollutants, damages, vessel activities and regions (see e.g. Tichavska and Tovar (2017) for a review). Because of the extensive modelling and data requirements for estimation of the external costs, several studies have transferred unit costs of different pollutants from primary studies. Such a cost transfer approach requires that the characteristics of the site to which costs estimates are transferred are very similar to the site of the original study. This is often not the case, and most of the studies on external costs of shipping apply population density as a means of transfer of results to the sites under study. However, the unit cost of air pollutant may also depend on other factors such as the quantity of emission and prosperity.

The question raised in the present study was whether the results from existing studies on external costs of air emissions from shipping could be used to quantify a function, which explains the relationship between external costs and quantities of air pollution that accounts for the different study characteristics. If so, the estimated cost function could be a useful transfer mechanism for assessing external costs of shipping. To this end, a meta-regression analysis (MRA) was performed. This has been used in a number of different studies as a means of acquiring information and data from previous research and projects (e.g. Stanley & Jarrell, 1989). MRA was first suggested by Glass (1976) and has the appeal of combining empirical evidence from a number of independent studies with different methods and datasets. It can provide insight into the determinants of effects when controlling for study characteristics and can thus provide more information than simply listing the results from a literature review.

To the best of the authors' knowledge, there has been no study involving an MRA of external costs of air pollutants originating from shipping. There has been a review of studies about the external costs of shipping (Tichavska & Tovar, 2017). With respect to external costs of transport, one study used MRA to examine the external costs of all transport modes except shipping (Quinet, 2004). It was applied to different modes of transport in western Europe, but did not assess the costs of different air pollutants. In the authors' view, the novel contribution of the present study is twofold: it is the first MRA of external costs of air pollutants from shipping and estimates the costs of different air pollutants. There is no limitation with respect to regional application.

The study is organised as follows. The basic approach for estimating external costs of air pollution from shipping is presented in Section 2. The choice of studies and a description of data are described in Section 3, and the results from the MRA are presented in Section 4. Based on the results in Section 4, marginal costs of different pollutants are calculated in Section 5. The study ends with a brief summary, discussion and the main conclusions.

2. Approaches to cost calculations

In principle, external costs of air pollution from shipping (and from other emission sources) are calculated as the associated impacts on human welfare, which occur from damages on health, agriculture, material and buildings and environment. For a given exposure, the damage of a unit emission is the same irrespective of emissions source, such as energy production, transports and shipping. The damage of a certain pollutant from a specific source, such as shipping, is then calculated by following a so-called impact pathway assessment (IPA), which comprises four steps: (*i*) the calculation of emissions of different pollutants, (ii) an assessment of the spatial and dynamic dispersal of each emission, (iii) the quantification of the impact of each emission on, for example, human health and ecosystems and (iv) an assessment of their impact in monetary terms (e.g. Bickel & Friedrich, 2005).

With respect to the first step, the calculation of emissions from a vessel involves information on emission factors related to, for example, engine type and transport distances. Several studies calculate emissions from ports, which require further data on the number of vessels in a port (e.g. Tichavska & Tovar, 2015; Wang et al., 2016). With information on vessel type and shipping route, it is possible to calculate emissions from seagoing vessels and create modelling systems for specific geographical areas (e.g. Jalkanen et al., 2009). An alternative method is to calculate emissions from data on the sales of different types of fuels (e.g. Wang & Corbett, 2007). Most studies on emissions from shipping include several pollutants, particularly NO_x, SO_2 and PM, but also volatile organic compound (VOC) and green-house gases (GHG).

In the second step, the dispersal of pollutants and final deposition where damage is caused are usually assessed by meteorological models such as the European Monitoring and Evaluation Programme (EMEP) model, which estimate dispersal and deposition in Europe from sources in Europe (e.g. Maffii, Molocchi, & Chiffi, 2007; Miola et al., 2009). The damage caused by the pollutants is quantified in the third step. Emissions of SO_x and NO_x formed during fuel combustion cause acidification of soil and water, a problem often referred to as "acid rain" (Salo et al., 2016). NO_x also contributes to eutrophication by increasing bioavailable nitrogen, and plays a role as a precursor for ground-level ozone and particles. In reaction with NO_x, VOC are precursors of ground-level ozone which damages health, vegetation and material (Endresen et al., 2004). Furthermore, through chemical transformations SO_x and NO_x may be converted into particles. Particles may lead to severe health issues in humans and create damages on the environment and materials (Eyring et al., 2010). For example, sulphur produced during combustion processes, such as burning of fossil fuels, can be oxidised to particulate sulphate (SO_4^{2-}) that can constitute a major proportion of airborne particles (Lack et al., 2009). These fine particles can be inhaled, get deep into the lungs and may also get into the bloodstream and are linked to human health problems such as asthma, heart disease and premature mortality (Corbett et al., 2007). Corbett et al. (2007) estimated that global emissions of PM caused the death of 60,000 people in 2002.

Emissions of GHG contribute to global climate change, whose damage has been assessed in a large number of studies (e.g. International Panel of Climate Change [IPCC], 2018). In general, the damages are non-linear in exposition to all air pollutants with increasing damages for a given increase in exposure. This would imply high damage cost per unit emission at high emission levels, which is represented by a cost function convex in emissions.

With regard to the final step, there are two main methods for translating quantified damage, e.g. health and the environment into monetary terms: revealed or stated preference methods. Preferences in markets are revealed directly or indirectly through market prices. For example, impacts on agriculture are calculated as the loss in yield multiplied by the market price (e.g. Miola et al., 2009). The monetary assessment of health impacts is not directly related to market prices in the same way as crop yield is. There is an old and large body of literature on the estimation of health impacts in monetary terms that applies a variety of methods (see e.g. Green, Brazier, and Deverill (2000) for a review). A common method is to assess the damage through indirectly revealed preferences. One example of this is the value of lost working days due to health degradation as associated decreases in salary, where the wage rate is found on the markets. Another example is expenses for healthcare, where the prices of goods and salaries, for example of nurses, are found in the markets. This is a commonly used and debated method for assessing the value of statistical life or the value of the life year, which is widely applied in road and infrastructure planning. However, this method does not capture the suffering caused by the illness, which is not reflected in the ability to work or expenses for healthcare. Therefore, methods based on stated preferences have been developed in which respondents are asked to state their willingness to pay for health improvements in hypothetical situations (e.g. Bahamonde-Birke, Kunert, & Link, 2015).

Irrespective of valuation method and scope, prosperity and wealth are likely to affect the external cost where, e.g. the cost of lost working days is large in high-income countries and the willingness to pay for accepting or avoiding damage is high for wealthy respondents (e.g. Carlsson & Johansson-Stenman, 2000). Further, the characteristics of the exposed areas, urban or rural, population density, etc. determine the scale of the effect such as a number of people exposed to health damage.

A full-fledged calculation of external costs of air emissions from shipping following the IPA requires extensive numerical modelling and data collection, which in many cases is not possible. Several studies, which are described in more detail in Section 3, therefore transferred external cost estimates from large-scale research projects involving researchers from different disciplines. Since the 1980s several projects have been undertaken, particularly in Europe, to assess the costs of air emissions from energy combustion following the IPA method: BeTa (Benefits Table database. Estimates of the marginal external costs of air pollution in Europe), CAFE (Clean Air for Europe), HEATCO (Developing Harmonised

European Approaches for Transport Costing) and NEEDS (New Energy Externalities Developments for Environmental Stressors in Europe). They include several air pollutants and combine dispersal and impact models to quantify and measure damage in monetary terms.

The projects consider several types of damage including health effects, impacts on crop yields, acidification and building materials, biodiversity and climate change. All projects provide estimates of costs of damages on health, which includes acute and chronic health impacts and are particularly detailed in HEATCO (Bickel et al., 2006). The projects differ with respect to the inclusion of other effects. CAFE considers damages on agriculture and provides damage estimates for different European countries (Holland & Watkiss, 2002). BeTa adds impacts on agriculture and buildings and provides estimates for emissions from shipping and for deposition on rural and urban areas in Europe (Holland & Watkiss, 2002). NEEDS provides the largest scope of damages by adding impacts on the environment (Korzhenevych et al., 2014; Preiss & Klotz, 2007). Similar to BeTa, estimates are provided for different European countries and for emissions from shipping. The projects use meteorological dispersal model, EMEP and response models as developed in the Extern-E (External costs of energy) EU projects. They estimate similar health impacts which include mortality, chronic and acute effects. With respect to damage valuation, such as the cost of mortality, which is the highest cost component, the research projects transfer results from valuation studies and adjust to GDP capita (see Korzhenevych et al., 2014 for a further description).

The projects have resulted in a recommended cost per unit emission, which are adjusted for the location of the emission source in different European countries and the characteristics of the affected area, such as urban and rural regions. For example, the marginal external cost of NO_x ranges between 1 thousand Euro (HEATCO) and 12 (CAFE) thousand Euro. The pattern of relative costs is similar for all projects where the cost of a unit emission of $PM_{2.5}$ is several times higher than that for SO_2, which, in turn, is slightly larger than that for NO_x. The magnitude of the costs is determined mainly by their impact on health, in particular, on mortality, which explains the high cost for $PM_{2.5.}$

Unlike emissions of NO_x, SO_2, PM and VOC, the location of the emission source has no impact on the damage caused by GHG since it is transformed into the atmosphere with global effects (e.g. IPCC, 2018). Therefore, if there were a consensus on the marginal damage cost of GHG, this should be used for all vessels. There are a large number of studies on the calculation of the social marginal cost of GHG that differ with respect to their method, time perspective, inclusion of different GHG, etc. (see e.g. Tol (2013) for a review). Since marginal damage is the same irrespective of location of emission source, this type of pollutant was excluded from the present study. An MRA of studies on the social cost of carbon would be more relevant (e.g. Van den Bijgaart, Gerlagh, & Liski, 2016).

3. Description of data

A systematic review was conducted during the collection of data on external costs and quantities of air emissions from shipping. The main challenges in the MRA were the selection of studies and choice of study-specific and contextual variables to code. These challenges were mitigated by a clear definition of the ultimate aim of the present study, which was to regress the costs of air emissions from shipping as a function of pollutant emissions, study-specific factors such as type of damage and vessel activity, and contextual factors including prosperity in the region.

3.1. Collection of studies

The first step was to search for studies using the keywords “external costs”, “air emissions” and “shipping” in the Google, Web of Science, Scopus and Google Scholar search engines. Google has the advantage of including reports from non-academic institutions and organisations, so-called grey literature, while Google Scholar identifies non-published academic works such as working papers. The number of studies was increased in the next step by applying the snowball method to selected studies published in international journals. In particular, references in and citations to review studies were useful (Tichavska & Tovar, 2017). The minimum requirements for the inclusion of studies were the availability of data on at least one type of emission, external costs of emissions, spatial application and ship activity. Results from studies relating emissions and costs per km or per ship, for example, without any possibility of calculating total emissions and costs, were not included. This excluded studies reporting, e.g. health damage in mortality terms without any monetary assessment, such as Corbett et al. (2007), Sofiev et al. (2018) and Barregard et al. (2019). The search was performed from September 2018 to July 2019. In total, 28 relevant studies with 106 observations were identified (Table A1 in Appendix). The mean value of 3.8 observations per study was in line with the average number of observations per study in meta-regression studies in environmental economics (Nelson & Kennedy, 2009).

The studies differed with respect to inclusion of pollutants and type of damage. Most studies included emissions of NO_x, SO_2 and PM, and all studies considered damage on health (see Table A1 in Appendix for a listing of primary studies and their inclusion of pollutants and type of damage). The studies used recommended cost estimates per unit emission by the research projects described in Section 2 but in different ways; a transfer of results from projects such as BeTa and CAFE (e.g. Castells Sanabra, Usabiaga Santamaría, & Martínez De Osés, 2014, Tzannatos, 2010a, 2010b), from a combination of projects and location-specific studies (e.g. McArthur & Osland, 2013; Miola et al., 2009) or from location-specific studies (e.g. Nerhagen, 2016; Song, 2014). The choice of transfer was adjusted to the specific conditions of the vessels under study, mainly population density in surrounding regions, to assess the costs of harm to health. However, in general, the studies did not account for the difference in other characteristics such as income level which would have an impact on the health cost (e.g. Berechman & Tseng, 2012; Tovar & Tichavska, 2019).

3.2 Effect size and moderator variables

Effect size refers to the dependent variable, which is the calculated external cost of emissions in this study, and the moderators refer to the explanatory variables. Despite the focus on studies with the estimation of external costs of emission from shipping in monetary terms, there is heterogeneity in the dataset because of e.g. inclusion of different pollutants and type of damage. The associated risk of comparing “apples” with oranges’ is a well-known problem in MRA, and different methods have been proposed to meet this challenge (see Tipton, Pusejovksy, & Ahmadi, 2019 for a historical development of MRA).

This includes division of the effect-size variable in subgroups and inclusion of covariates of study-specific characteristics and contextual factors.

The studies calculated the costs of emissions of NO_x, SO_2, VOC, $PM_{2.5}$ and/or PM_{10}. Almost all of the studies presented calculations of costs for each pollutant. Subgroups were therefore created where regressions were made with total emissions and separate pollutant emissions as effect-size variables and corresponding emissions as moderator variables.

However, since all pollutants generate several types of damages, problems with confounding may appear because of differences in damages. There is no data on the separation of damage types for different pollutants. Indicator variables were therefore introduced for the four types of damages included in the studies: health, agriculture, building material and environment in general. All studies considered damage on health and a majority added damages on agriculture (Table A1 in Appendix).

Another problem could be associated with the calculation of a specific damage type, such as health impacts, which could be made in different ways in the studies. This is usually accounted for in MRA of costs or values of environmental changes by introducing indicator variables for calculation methods such as stated or revealed preference method (e.g. Nelson & Kennedy, 2009). The studies included in the present study differed with respect to choice of transfer of damage cost, where a majority of the studies transferred external cost per unit of emissions from the different research projects on costs of transports of emissions discussed in Section 2. A few studies made own impact assessments (e.g. Miola et al., 2009; Nerhagen, 2016), and/or used country-specific damage costs (e.g. Gallagher, 2005; Nerhagen, Bergström, Forsberg, Johansson, & Eneroth, 2009). To account for these differences an indicator variable was introduced, "Method", which is unity when studies use own damage cost estimates and zero otherwise.

Study characteristics in addition to pollutants, type of damages and method were included as explanatory variables where the studies differed with respect to geographical application, and type of vessel activity. Indicator variables were constructed for regional application, distinguishing between Europe, the USA and other countries. Vessel activities were classified as being at port, at sea and all activities.

Contextual factors affecting the external costs included prosperity, population density in the exposed regions, and timing of the study. Gross domestic product (GDP) per capita was included as a measurement of prosperity in the countries of exposure and people per km^2 as a construct of population density. Another contextual factor was the timing of the study, where late studies may use updated or changed cost estimates, and a variable for the study year was included as an explanatory variable.

In the meta-analysis, the existence of publication bias is usually examined, i.e. whether results are influenced by the possibility of publication in scientific journals (e.g. Nelson & Kennedy, 2009). For example, insignificant results from the statistical analysis may be less likely to be submitted for publication. The existence of publication bias can be tested if there is information on variances and means of the included observations. This was not the case in the present study, and therefore an indicator variable was used where "published" = 1 when published in an international journal and zero otherwise.

All cost estimates were made at 2016 euro prices. When converting studies into this value, the real value of the cost in the currency used in the study was calculated, and then the 2016 euro exchange rate was applied. In total, 14 effect size variables were

Table 1. Descriptive statistics.

Variable	*N*	Mean	Standard deviation	Min.	Max.
Costs (million euros):					
Total	106	1000.84	3343.82	0.021	23,209.76
NO_x	92	557.45	1884.81	0.005	13,800.59
SO_2	92	499.77	1610.00	0.130	10,400.95
PM2.5	72	113.00	378.80	0.051	2131.89
PM10	8	10.74	22.32	0.436	65.61
VOC	44	13.46	33.47	0.001	148.26
Emissions ktonnes:					
Total	106	172.59	545.5	0.028	3616.2
NO_x	98	107.62	318.92	0.028	2008.6
SO_2	101	68.46	218.42	0.006	1450.7
PM2.5	78	4.36	20.56	0.001	154.00
PM10	12	0.74	1.19	0.001	3.67
VOC	48	9.40	27.33	0.001	156.90
Damage cost:					
Health	106	1	0	1	1
Agriculture	106	0.91	0.27	0	1
Buildings	106	0.72	0.45	0	1
Environment	106	0.29	0.46	0	1
Region:					
Europe	106	0.75	0.44	0	1
USA	106	0.10	0.31	0	1
Other	106	0.15	0.36	0	1
Activity:					
Port	106	0.65	0.49	0	1
At sea	106	0.23	0.42	0	1
All activities	106	0.12	0.33	0	1
Others:					
Method	106	0.22	0.41	0	1
GDP/capita, Euro	106	28,368	11,345	3736	79,075
People/km^2	106	375	1340	15	7096
Published in journals	106	0.79	0.41	0	1
Year	106	2008	4.91	1993	2016

included which is close to the average in MRA (Nelson & Kennedy, 2009). Descriptive statistics are presented in Table 1.

The range in cost estimates was large for all pollutants, which was also the case for emissions, GDP/capita and population density. Most studies calculated the costs of NO_x and SO_2 emissions. All studies included damages on health, and a majority also considered impacts on agriculture and buildings. Almost 80% of the studies were published in international journals with a referee system and 75% were applied to vessel activity in Europe. A majority, 60%, of the studies calculated the costs of emissions at ports.

4. Results from meta-regression analysis

To synthesise the effect on external pollutant costs of quantity of emissions and other explanatory variables, recent advances in MRA methods were used. The hierarchical nature of the dataset with study on the top level and observations from each study at the lower level, suggested a multi-level modelling which considers the variation and correlation within and between studies (e.g. Gelman & Hill, 2007). Within-study correlation may occur from the use of a specific calculation method, and between-study correlation from the transfer of unit costs of air pollutants from the same databases. A mixed-effect model was used which allows for differences among studies in the intercept and in the slopes of moderator variables. Regression was also made with varying intercept only, which corresponds to a random-effect model, but the statistical performance as measured by log-likelihood, AIC and BIC were improved when also allowing for the slope of emissions to vary among studies.

In order to mitigate the issue of heteroscedasticity, all effect size variables of cost and associated emission variables were transformed into natural logarithms. Since some studies reported costs for all emissions rather than separately for each emission, regressions were made with total cost as a dependent variable and separately for each pollutant. Because of the few observations for PM_{10} and VOC, which showed serious problems with outliers and multicollinearity, regressions for these pollutants were excluded. The following model was then estimated for total emissions and for each pollutant separately:

$$\ln Cost^{e,i,j} = \alpha_0^e + \alpha_1^e \ln Emission^{e,i,j} + \sum\nolimits_h \alpha_h^e X^{h,i,j} + \mu_0^{e,i} + \mu_1^{e,i} \ln Emission^{e,i,j} + \varepsilon^{e,i,j} \quad (1)$$

where the dependent variable $Cost^{e,i,j}$ is observation j of the cost estimate of emission e = *total emissions*, NO_x,SO_2, $PM_{2.5}$, in study i. The vector of explanatory variables, $X^{h,i,j}$, includes year of study, GDP per capita, population density, type of damage, method, publication outlet, regional application, and vessel activity, α_0^e,α_1^e, α_h^e are the meta-regression coefficients and $\varepsilon^{e,i,j}$ is the error term. The random effect in the intercept is presented by the term $\mu_0^{e,j}$ and the random effect in the slope of emission by the $\mu_1^{e,j} \ln Emission^{e,i,j}$.

With respect to expected signs of the meta-regression coefficients, they are expected to be positive for α_1^e since costs increase when emission is raised because of higher exposure to the pollutants. There are no priors on the relative magnitude of α_1^e between the pollutants. These coefficients show the increase in percent from 1% increase in emissions. Although external unit cost is highest for $PM_{2.5}$ as discussed in Section 2, this does not imply that the percentage increase is high. The regression coefficients for GDP/capita and population density are also expected to be positive since higher incomes and exposure to pollutants raise the costs.

When inserting the indicator variables, “Health”, “Port” and “Europe” were specified as the reference cases for type of damage, transport activity and region, respectively. Regressions were tested with the author as the panel group variable instead of study since the same authors could apply similar methods. Regressions were also made where the indicator variable “Europe” was divided into south and north Europe which could be justified because of different climate and other conditions. However, these alternative specifications resulted in lower values of statistical performance as measured by AIC and BIC.

Breusch–Pagan test showed concern with heteroscedasticity, and robust standard errors where therefore estimated. The Ramsey specification tests could not reveal the existence of missing moderator variables at the 5% confidence level for any model (Stewart, 2005). Regression results with the loglinear specification from the models with total emissions, NO_x, SO_2 and $PM_{2.5}$ are presented in Table 2. Regressions with $PM_{2.5}$ where made without indicator variables for damage type, since cost estimates of this pollutant included only damage to health.

The statistical performance of all models was satisfactory with significant explanatory power. Test results based on Cooks distance did not reveal that any of the observations were outliers or had high leverage in any of the models. Tests did not show any

Table 2. Regression results of mixed effect models for different emission categories.

	Dependent variable:			
Variable	lnCost$_{Tot}$	lnCost$_{NOx}$	lnCost$_{SO2}$	lnCost$_{PM2.5}$
Constant	−45.19	−42.29**	−73.65**	−45.137
	(0.227)	(0.012)	(0.025)	(0.675)
Ln Total emission	0.715***			
	(0.000)			
Ln NO_x emissions		0.762***		
		(0.000)		
Ln SO_2 emission			0.811***	
			(0.000)	
Ln PM2.5 emissions				0.865***
				(0.000)
Agriculture	−0.834	−2.575**	−0.829	
	(0.370)	(0.025)	(0.241)	
Material	1.001**	0.657	0.907	
	(0.033)	(0.289)	(0.145)	
Environment	−0.556	0.367	0.072	
	(0.384)	(0.532)	(0.895)	
Method	2.147**	1.110	1.075	2.625**
	(0.037)	(0.179)	(0.141)	(0.047)
USA	−0.256	1.093	−0.679	
	(0.760)	(0.207)	(0.377)	
Other countries	0.802***	0.751***	1.412***	1.282***
	(0.000)	(0.000)	(0.000)	(0.000)
Seagoing	−0.103	−0.468***	−0.178	−0.727*
	(0.769)	(0.000)	(0.462)	(0.088)
All vessel activity	1.188*	2.444**	2.339**	1.352
	(0.088)	(0.024)	(0.010)	(0.233)
Ln GDP/capita	0.581	−0.050	0.345	0.477*
	(0.227)	(0.844)	(0.120)	(0.050)
Ln Population density	0.071***	−0.195***	−0.054	−0.035
	(0.000)	(0.000)	(0.204)	(0.172)
Journal publication	1.431**	1.965**	2.871***	2.615**
	(0.013)	(0.039)	(0.000)	(0.015)
Year	0.021	0.023***	0.035**	0.021
	(0.291)	(0.006)	(0.035)	(0.694)
$\mu_1^{e,j}$	0.048	0.082	0.046	0.086
$\mu_0^{e,j}$	0.007–14	0.006–12	0.080	0.003–18
$\varepsilon^{e,i,j}$	0.196	0.111	0.149	0.417
N	106	92	92	69
Studies	28	22	22	17
Log likelihood	−114.96	−81.77	−83.18	−88.62
AIC	261.92	195.56	200.36	201.24
BIC	304.53	235.91	243.23	228.05
$p >$ chi2	0.000	0.000	0.000	0.000

Significance levels: ***$p < 0.01$.
**$p > 0.05$.
*$p < 0.10$.

concern for multicollinearity where none of the variance of inflation factors (VIFs) was above 7 (e.g. O'Brien, 2007).

Results common to all model specifications were the significant and positive estimates of all emission variables. The positive coefficients were expected since damage costs increase when emissions increase. These coefficients were interpreted as elasticities, i.e. the change in total cost in percent from a change by one percent in emissions. For example, the coefficient 0.715 for total emission showed that the total cost increased by 0.715% when total emissions increased by 1%. The results indicated differences in elasticities for the different pollutants, with $PM_{2.5}$ having the highest elasticity.

With respect to the impact of the indicator variables for damage type, the estimated coefficients show how the costs change when a study includes any other damage type in addition to health. When included, a robust result is the negative and significant impact of "Agriculture" and the positive of "Material". This indicates that the external costs decreases when "Agriculture" is included and increases when "Material" is added. The results also showed that studies with own estimates of damage give higher external costs since the coefficient of "Method" is positive for all models, and significant for total emissions and $PM_{2.5}$.

The indicator variables for regional application and shipping activity were interpreted in the same way, which implies that the studies applied on "Other countries" raise the cost and applied on seagoing vessels to reduce the cost. The positive sign of "Other countries" was explained by the inclusion of damages from shipping emission in the dense population Hong Kong region (Tovar & Tichavska, 2019).

Regarding the contextual explanatory variables, positive coefficient estimates of "ln GDP/capita" and "ln Population density" were expected from the discussion in Section 2, which were obtained only for the model with all emissions. These coefficients were also interpreted as elasticities and indicate that total external cost is elastic for the significant result and increased by 0.78% when GDP per capita increased by 1%. On the other hand, the elasticity with respect to population density is lower where, for the significant result, total cost increases by 0.071% when population density was raised by 1%. None of the pollutant specific regression models showed the expected results, but was significant only for the "ln Population Density" in the regression for NO_x. One reason could be that studies with NO_x consider rural areas with damage on agriculture and the environment.

Publication outlet showed significant and positive coefficient for all models, which indicates that external costs are higher for studies published in journals with referee system. This is also true for the timing of the studies, where reported costs are higher for later studies. This result is significant for the regression models with NO_x and SO_2.

5. Calculation of marginal external cost

All included studies applied constant marginal costs, i.e. the increase in total cost from an increase by one unit in emissions of each pollutant, to calculate external costs at different pollutant emission levels. However, if the marginal cost varies with the emission level, this approach may not give correct results. It is therefore of interest to calculate the marginal external cost based on the regression estimates of the cost functions presented in Table 2. Another aspect is the use of these cost functions as a mean for transfer of cost estimates as a complement to the unit cost transfer used in most primary studies. The marginal external cost was found by differentiating the loglinear cost functions presented in Table 2 with respect to quantity of emission e, which gives:

$$\frac{\partial Cost^e}{\partial Emission^e} = \beta^e \frac{Cost^e}{Emission^e} \tag{2}$$

where β^e is the estimated coefficient for pollutant e presented in Table 2. As shown in Equation (2), the marginal cost depends on the level of total cost and quantity of emission.

A high cost implies a high marginal external cost and *vice versa*. By calculating costs at different emission levels it would in principle be possible to use the estimated coefficient β^e to calculate costs at emission levels deviating from the point estimates made in most studies.

It can also be noted from Equation (2) that, in addition to emission levels, all explanatory variables affect $Cost^e$ and thereby the marginal cost. This means that the marginal cost is decreased when studies include “Agriculture”, increased when “Material” is added and so forth. The negative coefficient sign of “Agriculture” may reflect the lower cost in rural areas from crop damage, and the positive sign of “Material” by the closeness of ports to cities where damage on building material can be high.

In the reference case, the value of $Cost^e$ in the numerator on the right-hand side of Equation (2) was evaluated at the mean values of all explanatory variables. In order to examine the impact of emission levels on the marginal external cost, they were calculated for deviations by 50% from the average level of emissions. Marginal costs were also calculated for 10% increase in the number of studies including the different damage types. Since the coefficient estimates of “Material” and “Agriculture” were significant, calculations were made for these pollutants. Results are presented in Table 3.

The results indicated that the marginal external cost of $PM_{2.5}$ was approximately six times greater than for total emission, which reflected the relatively high costs of health impacts. The estimates of costs of single pollutants could be compared with the results from some other studies. The constant marginal cost of $PM_{2.5}$ ranged between 18 and 51 euro/kg $PM_{2.5}$ in Nerhagen (2016). The estimated average marginal cost of NO_x and SO_2 is close to estimates by Korzhenevych et al. (2014) who reported costs ranging between 1.8 and 5.9 euro/kg for NO_x and between 2.9 and 7.9 euro/kg for SO_2 in different European countries and for seagoing shipping. The relative difference in marginal costs for NO_x and $PM_{2.5}$ was in the same order of magnitude as reported for NO_x and $PM_{2.5}$ by Tichavska and Tovar (2015), but was lower than for similar estimates by Kalli and Tapaninen (2008) who found that the constant marginal cost of $PM_{2.5}$ can be ten times higher than for NO_x.

With respect to the impact of emission levels, all marginal costs decreased at higher emission levels, but to different degrees. The calculated external marginal cost of total emission deviates at the most by 23% from the mean level, and the external cost of $PM_{2.5}$ by 7%. The impacts of the indicator variables for health types showed that the external marginal cost of total emissions and NO_x would be reduced by 16% and 23%, respectively if the number of

Table 3. Marginal external cost evaluated at the mean values of explanatory variables, at 50% deviation from the average emission level, and at an increase by 10% in studies including damage on agriculture and material, thousand euros/tonne emission.

	Emission levels:			10% increase in studies with damage type[a]:	
	Average	50% decrease	50% increase	Agriculture	Material
Total emission	4.16	5.07	3.71		4.51
NO_x	3.94	4.80	3.57	3.03	
SO_2	5.92	6.73	5.58		
PM2.5	22.41	23.98	21.48		

[a]Calculations only for significant estimates of coefficients displayed in Table 2.

studies increased by 10%, which implies that all studies added agriculture since 91% of all studies included this damage type in the reference case (Table 2).

6. Discussion and conclusions

The purpose of this study was to estimate cost functions for air pollutants from shipping based on an MRA with observations obtained from studies that calculated costs. In total, 28 relevant studies were found, which differed with respect to inclusion of pollutants, damage types, regional focus, vessel activity and contextual factors. Most studies calculated costs of NO_x and SO_2, and were applied to ports and shipping in Europe, and made calculations for different damages and vehicle operative activity. In total, 106 observations could be used for the MRA.

Similar to air pollutants from other emission sources, the measurement of damage in monetary terms presents a challenge because of the need to estimate the spatial and dynamic dispersal of the pollutants, quantify the damage to health, yield on arable land, biodiversity, materials and buildings, and assess the damage in monetary terms. Most of the included studies transferred estimates of such damage from other studies and large-scale projects, and adjusted the values as measured in constant damage cost per tonne of emission to study specific conditions. The emissions and estimated costs showed a wide variation between the studies.

A drawback with the MRA in the present study was the lack of information about uncertainty in the measurement of costs in all the studies. Only five studies reported different value estimates for given emissions, which differed depending on the transfer of damage costs. This meant that all the observations were given equal weights. Studies with a small spatial scale would be expected to have relatively high precision with respect to the transfer of external cost of pollutants because of the availability of relevant information, for example, on population density. Studies with large spatial scales, such as seas in Europe, are likely to apply more assumptions regarding the characteristics of the affected regions. If so, the observations should be weighted according to their precision (e.g. Nelson & Kennedy, 2009; Stanley & Jarrell, 1989).

Nevertheless, the regression results of the cost functions showed robust results with respect to impacts of total emissions and specific pollutants (NO_x, SO_2, $PM_{2.5}$). A common result was the relatively inelastic response of costs to changes in emissions, i.e. that the per cent increase in total costs was below unity for a one percent increase in emissions of all pollutants. Calculations of marginal external cost at different emission levels showed that they can deviate by 25% from the average level which points out a need for careful transfer of constant unit costs. The estimated cost functions in this study could then provide a complementary transfer mechanism where external marginal costs can be calculated at different pollutant levels.

Another common result was that the cost decreased for inclusion of damage on agriculture and increased when material damage is considered. One reason for this result could be that studies including agriculture have a relatively large share of emission in the rural areas with crop damage the cost of which is lower than that of health damage. Health and material damage are likely to be complementary since they occur in densely populated urban areas.

It was also noticed that when transferring external cost to countries outside Europe, population density was applied as a transfer mechanism in several studies. Despite this, population density did not show any robust results in the regression analysis. This was also the case for the impact of GDP per capita. One reason can be that prosperity was not applied when transferring external costs. For example, Tovar and Tichavska (2019) used the average value of external cost for Europe when assigning external costs per unit emissions in Hong Kong and St Petersburg. This could be a reason for the non-expected, but the insignificant impact of GDP per capita on external cost in the present study.

The positive coefficient estimate of publication outlet was a robust result, which may indicate existence of publication bias where papers with relatively high estimates of external costs are submitted and accepted for publication in refereed journals. Similar results are common in the MRA literature, where in general significant results are more likely to be published (e.g. Stanley, 2005). Considering that 79% of the primary studies in present MRA were published in international journals, the real external cost can be lower than the published estimates. Another finding in the present study was that the external costs in the primary studies did not include damage to the sea from air pollution, which would increase the external cost of seagoing traffic. The evaluation of the net impact of eventual publication bias and expansion of the current cost calculation to include impacts on seas remain relevant areas of future research on MRA of external costs of air emissions from shipping.

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Appendix.

Table A1. Primary studies in the MRA.

	Authors and year of publication	Observations	Included pollutants	Damage types[a]
1	Holland and Watkiss (2002)	4 for different seas in Europe	NO_x, SO_2, VOC, $PM_{2.5}$	H, A, M
2	Gallagher (2005)	9 for different years	NO_x, SO_2	Not defined
3	Maffii et al. (2007)	5 for different seas in Europe	NO_x, SO_2, VOC	H, E
4	Wang and Corbett (2007)	2 for different ports	SO_2	H, A, M
5	Kalli and Tapaninen (2008)	3 for different vessel types	NO_x, SO_2, $PM_{2.5}$	H, A, M
6	Miola et al. (2009)	2 for different vessels	SO_2, $PM_{2.5}$	H, A, E
7	Nerhagen et al. (2009)	1	PM_{10}	H
8	Lee, Hu, & Chen, 2010	3 for different ports	NO_x, SO_2, VOC, PM_{10}	H, A, M
9	Tzannatos (2010a)	1	NO_x, SO_2, $PM_{2.5}$	H, A, M
10	Tzannatos (2010b)	1	NO_x, SO_2, $PM_{2.5}$	H, A, M
11	Berechman and Tseng (2012)	1	NO_x, SO_2, VOC, $PM_{2.5}$, PM_{10}	H, A, M
12	Jiang, Kronback, and Christensen (2013)		NO_x, SO_2, $PM_{2.5}$	H, A, M
13	McArthur and Osland (2013)	1	NO_x, SO_2, VOC, $PM_{2.5}$, PM_{10}	H, E
14	Castells Sanabra et al. (2014)	1	NO_x, SO_2, VOC, PM	H, E
15	Kilic and Tzannatos (2014)	1	NO_x, SO_2, VOC, $PM_{2.5}$, PM_{10}	H, A, M
16	Castells Sanabra et al. (2014)	13 for different ports	NO_x, SO_2, $PM_{2.5}$, VOC	H, A
17	Song (2014)	1	NO_x, SO_2, $PM_{2.5}$, PM_{10}	H, E

(*Continued*)

Table A1. Continued.

	Authors and year of publication	Observations	Included pollutants	Damage types[a]
18	Maragkogianni and Papaefthimiou (2015)	5 for different ports	NO_x, SO_2, $PM_{2.5}$	H, A, M
19	Tichavska & Tovar, 2015	4 for different vessel types	NO_x, SO_2, VOC, $PM_{2.5}$	H, A, M
20	Vierth and Sowa (2015)	2 for different transport routes	NO_x, SO_2, $PM_{2.5}$	H, E
21	Nerhagen (2016)	3 for different seas	NO_x	H, A
22	Sliskovic, Hadzic, and Vukic (2017)	1	NO_x, SO_2, PM	H
23	Yaramenka, Winnes, Åström, and Fridell (2017)	1	NO_x	H
24	Dragović, Tzannatos, Tselentis, Meštrović, and Škurić (2018)	6 for different ports and years	NO_x, SO_2, $PM_{2.5}$	H, A, M
25	Liu et al. (2018)	3 for different ports	SO_2, PM_{10}	H, E
26	Zhu, Fu, Ng, Luo, and Ge (2018)	2 for different transport routes	NO_x, SO_2, $PM_{2.5}$	H, A, M, E
27	Nunes, Alin-Ferraz, Martins, and Sousa (2019)	4 for different ports	NO_x, SO_2, $PM_{2.5}$, VOC	H, A
28	Tovar and Tichavska (2019)	12 for different ports	NO_x, SO_2, $PM_{2.5}$	H, A, M

[a]H = health, A = agriculture, M = material, E = environment.