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Estimated health impacts from maritime transport in the Mediterranean region and benefits from the use of cleaner fuels

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ABSTRACT

Ship traffic emissions degrade air quality in coastal areas and contribute to climate impacts globally. The estimated health burden of exposure to shipping emissions in coastal areas may inform policy makers as they seek to reduce exposure and associated potential health impacts. This work estimates the $PM_{2.5}$-attributable impacts in the form of premature mortality and cardiovascular and respiratory hospital admissions, from long-term exposure to shipping emissions. Health impact assessment (HIA) was performed in 8 Mediterranean coastal cities, using a baseline conditions from the literature and a policy case accounting for the MARPOL Annex VI rules requiring cleaner fuels in 2020. Input data were (a) shipping contributions to ambient $PM_{2.5}$ concentrations based on receptor modelling studies found in the literature, (b) population and health incidence data from national statistical registries, and (c) geographically-relevant concentration-response functions from the literature. Long-term exposure to ship-sourced $PM_{2.5}$ accounted for 430 (95% CI: 220–650) premature deaths per year, in the 8 cities, distributed between groups of cities: Barcelona and Athens, with > 100 premature deaths/year, and Nicosia, Brindisi, Genoa, Venice, Msida and Melilla, with tens of premature deaths/year. The more stringent standards in 2020 would reduce the number of $PM_{2.5}$-attributable premature deaths by 15% on average. HIA provided a comparative assessment of the health burden of shipping emissions across Mediterranean coastal cities, which may provide decision support for urban planning with a special focus on harbour areas, and in view of the reduction in sulphur content of marine fuels due to MARPOL Annex VI in 2020.

1. Introduction

Shipping is one of the most efficient modes of freight transportation per ton of cargo, especially when compared to aircraft or trucks (Grewal and Haugstetter, 2007; Karl et al., 2019; Micco and Pérez, 2001), and as a result has increasing relevance for economic exchange globally. This is also enhanced by the globalisation of manufacturing processes and the increase of global-scale trade (Corbett and Fischbeck, 2000; Marmer et al., 2009; US-EPA, 2009). In Europe, currently almost 90% of the European Union import and export freight trade is seaborne (Karl et al., 2019), with a total of 2.1 million vessels calling in main European ports in 2016 (EUROSTAT, 2019). In addition, tourist cruises and ships are also increasing across Europe, even in inland waterways (EEA, 2018), with a total of 397 million passengers in EU ports in 2016, a rise of 0.4% from the previous year (EUROSTAT, 2019). In this framework, the Mediterranean region stands out due to the significant growth in the number of vessels calling in ports between 2011 and 2016 (Spain, 9.7%; Croatia, 20.1%, Malta, 14.8%; Cyprus, 11.1% between 2015 and 2016) when compared to Northern ports (Belgium, −8.7%; Denmark, −23.2%; Germany, −0.5%), and also due to the relevant share (36%) of the total cargo handled in EU28 ports in 2016 (EUROSTAT, 2019). This intensive Mediterranean shipping traffic takes place in a region known to be densely populated (EUROSTAT, 2019) and environmentally sensitive (Dulac et al., 2016), with low atmospheric dispersion (Millán et al., 1991).

The fact that ship traffic contributes to air quality degradation in

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coastal areas and climate impacts globally is well known (Eyring et al., 2010; Sorte et al., 2019b). Shipping emissions are constituted by primary and secondary particulate matter, mainly in the fine size fraction ($PM_{2.5}$) and including black carbon (BC), and in addition by sulphur dioxide (SO_2), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOC) and carbon dioxide (CO_2) (Corbett et al., 2007; Eyring et al., 2007, 2005; Fridell, 2019; Fridell et al., 2008; Johansson et al., 2017; Johnson et al., 2018; Lack and Corbett, 2012; Landrigan et al., 2017; Moldanová et al., 2013). The air quality impacts of shipping emissions have been described for European coastal areas and port cities (Cesari et al., 2014; EEA, 2013; Karagulian et al., 2015; Ledoux et al., 2018; Merico et al., 2016, Merico et al., 2017; Nunes et al., 2019; Pérez et al., 2016; Sorte et al., 2019a,b; Viana et al., 2014, 2009; among others). Maritime transport emissions have also been the subject of modelling studies globally as well as in the Mediterranean region (among others, Eyring et al., 2010, 2007; Jalkanen et al., 2012, 2009; Monteiro et al., 2018; Russo et al., 2018; Sofiev et al., 2018).

Similarly, the health impacts of exposure to atmospheric particulate matter, including that emitted by ships, are described in the literature (Anderson et al., 2012; Lelieveld et al., 2015; Pope et al., 1995; Valavanidis et al., 2008). The WHO-HRAPIE project concluded that shipping was the third among the top six air pollution emission source categories (of a total of 16) posing an emerging health risk, after road transport and space heating and air conditioning (Héroux et al., 2015). The ability to assess the health effects and the chemical and physical characteristics of ship pollutant exposure were identified as key knowledge gaps and driving forces of this emerging risk (Henschel and Chan, 2013). According to the 2015 Global Burden of Disease Study (Cohen et al., 2017), exposure to outdoor $PM_{2.5}$ is the fifth leading risk factor for death worldwide, accounting for 4.2 million deaths and 103 million disability-adjusted life-years in 2015. However, this burden is mostly reported for bulk $PM_{2.5}$ concentrations and it is not usually disaggregated by emission source. Estimating the source-specific health burdens is key to achieving a deeper understanding of the role of different emission sources (such as shipping) as determinants of health impacts, as well as to support the design of effective and targeted mitigation strategies. Health impact assessment (HIA), which may have different purposes (Wernham, 2011), is used in this framework to quantify these health impacts on the basis of the combination of health impact functions and risk estimates from the epidemiology literature to relate hypothesized air quality changes to a population at risk (Fann et al., 2011). The body of literature is sustainably increasing on this topic (Andersson et al., 2009; Broome et al., 2015; Caiazzo et al., 2013; Castro et al., 2017; Corbett et al., 2007; Fantke et al., 2019; Holnicki et al., 2017; Malmqvist et al., 2018; Mueller et al., 2017; Oudin et al., 2016; Rouïl et al., 2019; Silva et al., 2016; Smith et al., 2009; Sofiev et al., 2018; Viana et al., 2015). Specifically in Barcelona (Spain), all-cause mortality excess risks (95% CIs) showed a statistically significant association with inter-quartile range increases in oil-combustion derived $PM_{2.5}$ (Ostro et al., 2011).

Because of the known adverse effects on human health and the environment of shipping emissions, the International Maritime Organization (IMO) regulates air pollution from maritime transport through its convention MARPOL (Marine Pollution Convention), Annex VI (Regulation for the Prevention of Air Pollution from Ships; IMO, 2008). Sulphur is the target of this reduction (in addition to NOx and PM) due to shipping's significant contribution to global sulphur inventories (13% of total SOx emissions annually; IMO, 2014). In the EU, Council Directive 1999/32/EC relating to a reduction in the sulphur content of certain liquid fuels limits the maximum sulphur content of marine fuel. It also contains additional fuel-specific requirements for ships calling at EU ports, specifically obligations related to the use of cleaner fuels when vessels approach the EU coastline. A detailed description of the legislative status of shipping emissions in the EU may be found in (EEA, 2013). Regarding new scenarios, MARPOL Annex VI establishes that, after the year 2020, the sulphur content of fuels used on-board ships must be 0.5% or lower (as opposed to 3.50% m/m (mass by mass) for ships operating outside Emission Control Areas since 2012; (IMO, 2019). This limit is expected to drive environmental benefits around the globe (Lack et al., 2011; Rouïl et al., 2019; Sofiev et al., 2018; Winebrake et al., 2009), especially in regions where stricter limits do not yet apply. Globally, the post-2020 cleaner marine fuels are expected to reduce ship-related premature mortality and morbidity by 34 and 54%, respectively, representing a > 2% global reduction in $PM_{2.5}$ cardiovascular and lung cancer deaths (Sofiev et al., 2018). In the Mediterranean region, the potential implementation of an emission control area (ECA) could avoid nearly 1730 premature deaths each year (Rouïl et al., 2019).

Using the HIA, the present work aims to contribute to the deeper understanding of the environmental and health impacts of shipping emissions, by addressing an especially sensitive region: the Mediterranean basin. Building on the literature available on health impacts of shipping at global scale (Sofiev et al., 2018), the focus on this region will allow for a deeper understanding of the differences in environmental and health impacts observed between specific cities. Three intermediary objectives proposed are: (1) to review the source apportionment literature to update the knowledge on shipping contributions to air quality degradation in Mediterranean coastal cities, (2) to estimate the health burden of shipping emissions in 8 Mediterranean cities, and (3) to quantify the health benefits expected after the implementation of the 2020 0.5% sulphur limit on marine fuels (in contrast to 3.50%).

2. Materials and methods

2.1. Literature review

Studies on source apportionment of shipping emissions and their contribution to coastal air quality degradation were reviewed, based on research databases Scopus and ScienceDirect. The search terms included "shipping", "air quality", "maritime", "oil", "fuel oil", "air pollution", "source apportionment" and "receptor model". The focus of this review was on receptor modelling studies (e.g., Karagulian et al., 2015; Viana et al., 2008), as opposed to dispersion modelling ones (e.g., Karl et al., 2019). The aim of this review was to focus on experimentally-determined PM mass concentrations and chemical composition, which had been sampled in the different cities using comparable sampling methods (most of the time, EU reference PM samplers according to the EU Air Quality Directive) and for which the data were processed using a limited number of receptor models (Principal Component Analysis, PCA; Positive Matrix Factorization, PMF; Multilinear Engine, ME; Chemical Mass Balance, CMB, and aerosol chemical composition as a function of wind direction analysis; Viana et al., 2009). The absence of uncertainty estimates (e.g., by bootstrapping), which is frequent in receptor modelling studies, should be considered a limitation of this work. Despite this, targeting receptor modelling studies was estimated to facilitate comparability between studies due to the comparable sampling methodologies and the limited number of receptor models used for data processing (see Table 2). The uncertainties of the alternative approach (i.e., dispersion models) are described in the literature (EEA, 2013; Gong et al., 2018; Johansson et al., 2017; Karl et al., 2019). The publication years ranged between 2013 and 2018, with the aim to update and expand the results published by Viana et al. (2014). To be included in this work literature studies were required to discriminate and quantify contributions from shipping emissions to $PM_{2.5}$ urban background concentrations in a Mediterranean coastal city, in terms of concentrations ($\mu g/m^3$) and % of $PM_{2.5}$ mass concentration, and as annual means. The shipping contributions reported at urban background locations were assumed to be representative for each entire city. Studies reporting source apportionment data along ship routes (5 studies) were excluded under the assumption of having low impacts on human exposure. Similarly, 3 studies were excluded due to the absence

Table 1
Overview of studies reporting endpoints and concentration-response functions (CRFs) used in this work, for long-term health impacts.

Endpoint	Pollutant	Reference	Beta	Distribution Beta	Uncertainty
Hospital Admissions, Respiratory	$PM_{2.5}$	Hoek et al. (2013)	0.002956	Normal	0.002740312
Hospital Admissions, Cardiovascular	$PM_{2.5}$	Hoek et al. (2013)	0.005827	Normal	0.000962763
Mortality	$PM_{2.5}$	Beelen et al. (2014)	0.0117	Normal	0.0058
Mortality	$PM_{2.5}$	Hoek et al. (2013)	0.003922	Normal	0.000491

of quantitative contributions. In total, 34 studies were initially identified and 26 were selected for subsequent analysis (Table S1).

2.2. Health impact assessment and scenario analysis

The literature review facilitated the identification of the cities for which source apportionment data for shipping were available. Subsequently, the health burden of shipping emissions was estimated using the US-EPA BenMap-CE model (Environmental Benefits Mapping and Analysis Program - Community Edition; https://www.epa.gov/benmap). BenMAP-CE is a publicly available software which estimates the air pollution-related mortality and morbidity. While similar tools are also available (e.g., WHO's AirQ+; http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/activities/airq-software-tool-for-health-risk-assessment-of-air-pollution), BenMap was selected as it has been previously used for similar purposes (Boldo et al., 2011; Broome et al., 2015; Fann et al., 2018) and in US-EPA Regulatory Impact Analyses (www.epa.gov/ttn/ecas/ria.html). The BenMap default inputs were modified to reflect the specific geographic locations, incidence rates and concentration-response functions for each city.

The following input data (Tables 1, 2, 3 and S1), aggregated at city level, were collected for the health impact assessment (HIA) from national statistical and health registries in each country:

- Population data statistics: statistics for the year(s) for which source apportionment data were available were obtained for total population (ages 0–99), aggregated at city level, from national statistical databases in each country.
- Incidence data: incidence data were collected for WHO International Classification of Diseases (ICD) codes ICD-10 I00–I99 (diseases of the circulatory system), ICD-10 J00–J99 (diseases of the respiratory system), and ICD-10 A00–R99 (all-cause mortality). Incidences were considered to be distributed homogeneously in each city. The specific data sources are reported in the Acknowledgements section.
- Health impact functions: impact functions were selected aiming to maximise the geographical representativity for the study region, i.e., the functions selected were considered representative of European populations (Table 1). Data were available from research projects carried out in the Mediterranean basin (APHEA-2, MedParticles) and also from meta-analyses (Héroux et al., 2015). In the specific case of the meta-analysis used for mortality calculations, the study (Beelen et al., 2014) was carried out for an adult population, whereas in the present analysis ages 0–99 were considered. It should thus be acknowledged that risk estimates in Beelen et al. (2014) were applied to a larger population, in this work. Whenever more than one study was available for a given endpoint (e.g., Beelen et al., 2014; Hoek et al., 2013, for long-term mortality from exposure to $PM_{2.5}$), the burden of disease was calculated with both CRFs and the results are shown as a range.
- Shipping pollution, baseline and air pollution reduction scenarios: mean annual contributions from shipping emissions to ambient $PM_{2.5}$ concentrations, as well as annual mean $PM_{2.5}$ concentrations, were obtained for each city from the literature review, based on receptor modelling. A minimum of one study was available for each city, and therefore an average shipping contribution concentration was used for each city. As will be discussed in the Results section, different models (PCA-MLRA, PMF, ME, etc.) were used by authors in each city, which limits the comparability between results. The health burden of shipping emissions was estimated by running a policy scenario in BenMap whereby $PM_{2.5}$ concentrations were reduced by the total shipping contribution, in each city. A second scenario was run to model the reductions expected for the year 2020, when the sulphur content of fuels used on-board ships must be 0.5% or lower. Expected $PM_{2.5}$ reductions (including primary and secondary aerosols) under this scenario were obtained for each of the study cities from Sofiev et al. (2018).

The BenMap health impact assessment was run for 8 coastal cities (Nicosia, Brindisi, Genoa, Venice, Msida, Barcelona, Melilla, Athens), as described below. The selection was complex and required making compromises regarding the different types of data available, and as a result the following issues should be considered as limitations of this work: (a) receptor models reported shipping emissions in 4 of the cities (Venice, Msida, Barcelona and Melilla), while in the remaining 4 (Nicosia, Brindisi, Genoa and Athens) the source reported was "oil combustion" (identified by the authors as referring mainly to shipping), and therefore the source types are not directly comparable; (b) the monitoring locations within the cities ranged between urban, traffic and harbour locations, and were considered representative of city-wide contributions for the purpose of comparison with population and health incidence statistics; (c) the source apportionment data spanned between years 2007 and 2016, and therefore different years were assessed for the different cities; (d) hospital admission data were not available for Athens or Barcelona due to administrative constraints; (e) the all-cause mortality data obtained for Msida from official statistics (72 deaths in the year 2016) was unexpectedly low, and (f) due to the different study years selected (ranging from 2007 to 2016) different shipping legislation may have been in place in the different study locations.

3. Results

3.1. Contributions from shipping emissions to coastal air quality degradation

A total of 26 studies was identified which reported shipping, oil combustion and/or harbour contributions to PM concentrations for coastal Mediterranean cities and sampling locations in 7 countries (Spain, Italy, Greece, Malta, Turkey, Cyprus and France; see Table S1 in Supporting Information). As previously reported (Viana et al., 2014), one of the major difficulties found in these works was the discrimination between oil combustion sources, and as a result many of the publications reviewed which reported mixed sources (e.g., including shipping and refinery emissions) were finally included in the review. In total, 15 of the 26 studies reviewed reported shipping emissions while the remaining 11 referred to oil combustion sources. The review was also extended to include also contributions to PM_{10} mass concentrations (14 of the 26 studies), but contributions to other size fractions (as reported, e.g., by Karanasiou et al., 2009; Vecchi et al., 2008) were excluded. The observational data in these studies was collected between the years 2003–2015, and analysed using receptor models (PMF, CMB, PCA, ME; Amato et al., 2009; Cesari et al., 2014; Diapouli et al., 2017; Karanasiou et al., 2009; Merico et al., 2016; Viana et al., 2008; Perez

Table 2
Shipping and mixed oil combustion contributions to $PM_{2.5}$ concentrations (in μg/m^3 and %) obtained from literature and quantified by receptor modelling tools, for the 8 cities selected and as reported by the original publications. Ship-$PM_{2.5}$: ship-sourced $PM_{2.5}$ concentrations (by receptor modelling).

		Ship-$PM_{2.5}$ (μg/m^3)	Ship-$PM_{2.5}$ (%)	Aerosol contribution	Emission source	Study year	Monitoring site	Model	Reference
Nicosia	(Cyprus)	1.2	8	Primary + secondary	Oil combustion	2007–2009	Urban	PMF	Koçak et al. (2011)
Brindisi	(Italy)	2.3	15.3 ± 1.3	Primary + secondary	Mixed oil combustion	2010	Harbour-industrial	PMF	Cesari et al. (2014)
Genoa	(Italy)	0.26	15 ± 2	Primary + secondary	Oil combustion	2011	Urban	PMF	Bove et al. (2014)
Venice	(Italy)	0.1–0.7	1.0–8.0	Primary + secondary	Shipping	2007	Cruise terminal	Wind analysis	Contini et al. (2011)
Msida	(Malta)	0.8	5	Primary + secondary	Shipping	2016	Traffic	PMF	Scerri et al. (2018)
Melilla	(Spain)	2.6	14	Primary + secondary	Shipping	2007–2008	Urban	Wind analysis	Viana et al. (2009)
Athens	(Greece)	1.0 ± 0.3	6	Primary + secondary	Oil combustion	2011–2012	Urban	PMF	Diapouli et al. (2017)
Barcelona	(Spain)	1.0	6**	Primary	Harbour-shipping	2011	Urban	PMF	Pérez et al. (2016*)

* A previous study (Amato et al., 2009) reported a mean annual contribution of 1.8 μg/m^3 to $PM_{2.5}$ (6%) from oil combustion, including shipping, in Barcelona.
** Shipping contribution to primary aerosols; secondary aerosol formation derived from shipping accounted for 5–9% of $PM_{2.5}$ (Pérez et al., 2016).

et al., 2016), hybrid models (e.g., Argyropoulos et al., 2017; Bove et al., 2014) and alternative methods (e.g., high temporal resolution PM and wind direction data (Contini et al., 2011). Results evidenced that shipping/oil combustion emissions (mostly primary) contributed with 1–10% of PM_{10} mass (up to 19% for oil combustion sources) and 2–17% of $PM_{2.5}$ mass (up to 20% for oil combustion) in Mediterranean coastal cities. It should be noted that the cities were characterised by different sizes and ship-traffic volumes, and that the emission sources identified occasionally showed mixed profiles (e.g., harbour + industrial; Cesari et al., 2016, 2014) ***(Cesari et al., 2016). In addition, while the majority of studies reported shipping contributions in terms total shipping contributions, some reported primary and secondary contributions separately. For the present work, when in doubt, the contribution to primary aerosols was the only one reported, following a conservative approach (aiming to avoid overestimating the cumulative health effects). The use of only primary contributions, which would underestimate the adverse health outcomes, might compensate for the potential overestimations arising from extrapolating data from a single location to the entire city and from including impacts from other sources such as oil combustion (instead of strictly considering shipping emissions).

The data reported (Fig. 1, Tables 2 and S1) update those reviewed in 2014 for a more limited number of locations, also based only on receptor models (not dispersion modelling tools), which reported slightly lower contributions from shipping in European coastal areas (1–7% of ambient PM_{10}, 1–14% of $PM_{2.5}$, and at least 11% of PM_1; Viana et al., 2014). In the present review, shipping contributions to mean annual $PM_{2.5}$ concentrations reported in the different coastal cities were 5–14% in Spain, 1–14% in Italy, 2–10% in Greece (4–17% for oil combustion), 8% in Cyprus, 7% in France (Marseille region) and 5% in Malta (and 11–18% of PM_{10} in Turkey; Tables 2 and S1). While certain of the studies selected for this work were carried out in harbour stations (e.g., Venice; Table 2) and could thus be especially impacted by ship emissions, the majority refer to urban background and traffic sites. The relevance of maritime transport as a contributor to air quality degradation in Mediterranean coastal areas becomes, thus, evident.

The mean contributions of shipping emissions to air quality in European coastal cities were compared to contributions in other regions around the globe, for reference. Across the Yangtze River Delta in China (Feng et al., 2019), on average, ships contributed 0.36 μg/m^3 in January and 0.75 μg/m^3 in June to ambient $PM_{2.5}$ concentrations, and accounted for 5.2% of $PM_{2.5}$ in Shanghai in 2015 (3.6% from inland-water ships, 1.6% from coastal ships). Also in Shanghai, in the port area, primary $PM_{2.5}$ shipping emissions contributed with 4.2% of the average $PM_{2.5}$, reaching maxima of 12.8% (Zhao et al., 2013). More recently, shipping was reported to account for 2.4% of mean $PM_{2.5}$ concentrations in Yangshan Harbor (Shanghai; Mamoudou et al., 2018). Finally, high contributions were reported for the Pearl River Delta region (Tao et al., 2017): ship emissions, a source previously ignored in making emission control policies in this region, were among the top contributors to $PM_{2.5}$ in Guangzhou and Zhuhai, accounting for > 17% of $PM_{2.5}$ mass concentrations. Comparable results are obtained in certain US regions: 18.8% of mean daily modelled $PM_{2.5}$ in central Los Angeles (Vutukuru and Dabdub, 2008), between 8.8% (at the monitoring location closest to the port, West Long Beach) and 1.4% of $PM_{2.5}$ (at a monitoring location 80 km inland; Agrawal et al., 2009) within the Southern California Air Basin, and between 5% and 43% in the remote Northwestern United States (Cheeka Peak Atmospheric Observatory; Hadley, 2017).

Health impact assessment was carried out in a final selection of 8 European cities, based on data availability for the variables described in the Methodology section ($PM_{2.5}$ source apportionment, population, and health incidence data): Nicosia (Cyprus), Brindisi, Genoa and Venice (Italy), Msida (Malta), Barcelona and Melilla (Spain), and Athens (Greece) (Table 2 and Fig. 1). In the 8 final study cities, absolute contributions from shipping or oil combustion sources to $PM_{2.5}$ were

Table 3
Estimated health outcomes (all-cause mortality, and hospital admissions, with confidence intervals) due to long-term exposure to shipping emissions, for cities and years: Nicosia 2007–2009, Brindisi 2010, Genoa 2011, Venice 2007, Msida 2016, Melilla 2007–2008, Athens 2011–2012, Barcelona 2011. Results are presented as annual averages when > 1 year of data was available. Population (number of inhabitants).

Input data			HIA model output				
City	Population	Shipping $PM_{2.5}$**	Health endpoint	Baseline (N)*	Incidence rate	Outcomes due to shipping/oil (N)***	% of baseline
Nicosia	335,345	1.2 μg/m^3	All-cause mortality	4715	1.4%	22–66 (95% CI: 2–129)	0.5%-1.4%
			Respiratory hospital admissions	1770	0.5%	6 (95% CI: −5–18)	0.4%
			Cardiovascular hospital admissions	2925	0.9%	20 (95% CI: 14–27)	0.7%
Brindisi	88,068	2.3 μg/m^3	All-cause mortality	828	0.9%	7–22 (95% CI: 1–43)	0.9%-2.7%
			Respiratory hospital admissions	1331	1.5%	9 (95% CI: −7–25)	0.7%
			Cardiovascular hospital admissions	2350	2.7%	31 (95% CI: 21–41)	1.3%
Genoa	619,250	0.3 μg/m^3	All-cause mortality	8222	1.3%	8–25 (95% CI: 1–49)	0.1%-0.3%
			Respiratory hospital admissions	543	0.1%	0.4 (95% CI: −0.3–1)	0.1%
			Cardiovascular hospital admissions	3040	0.5%	5 (95% CI: 3–6)	0.2%
Venice	520,014	0.4 μg/m^3	All-cause mortality	4727	0.9%	7–22 (95% CI: 1–43)	0.1%-0.5%
			Respiratory hospital admissions	304	0.1%	0.4 (95% CI: −0.3–1)	0.1%
			Cardiovascular hospital admissions	1798	0.3%	4 (95% CI: 3–6)	0.2%
Msida	10,889	0.8 μg/m^3	All-cause mortality	72	0.7%	0.2–0.7 (95% CI: 0.02–1)	0.3%-1.0%
			Respiratory hospital admissions	485	4.5%	1 (95% CI: −1–3)	0.2%
			Cardiovascular hospital admissions	641	5.9%	3 (95% CI: 2–4)	0.5%
Melilla	70,444	2.6 μg/m^3	All-cause mortality	1251	1.8%	13–37 (95% CI: 1–73)	1.0%-3.0%
			Respiratory hospital admissions	127	0.2%	1 (95% CI: −1–3)	0.8%
			Cardiovascular hospital admissions	392	0.6%	6 (95% CI: 4–8)	1.5%
Athens	664,046	1.0 μg/m^3	All-cause mortality	25,938	3.9%	102–302 (95% CI: 8–591)	0.4%-1.2%
			Respiratory hospital admissions	N.A.	N.A.	N.A.	N.A.
			Cardiovascular hospital admissions	N.A.	N.A.	N.A.	N.A.
Barcelona	5,529,099	1.0 μg/m^3	All-cause mortality	15,219	0.3%	60–177 (95% CI: 5–347)	0.4%-1.2%
			Respiratory hospital admissions	N.A.	N.A.	N.A.	N.A.
			Cardiovascular hospital admissions	N.A.	N.A.	N.A.	N.A.

* Baseline (N): number of outcomes attributable to $PM_{2.5}$ from all emission sources (including shipping).
** Shipping $PM_{2.5}$: source apportionment results obtained from literature (see Table 2 for the references). Baseline: outcomes attributable to PM2.5 from all emission sources (including shipping).
*** Outcomes due to shipping/oil (N): number of outcomes attributable to shipping/oil emissions, including the confidence interval (CI).

highest in Brindisi (2.3 μg/m^3) and Melilla (2.3 μg/m^3), followed by Nicosia (1.2 μg/m^3), Barcelona and Athens (1.0 μg/m^3, each), and finally Msida (0.8 μg/m^3), Venice (0.1–0.7 μg/m^3, averaged to 0.4 μg/m^3) and Genoa (0.26 μg/m^3) (Table 2 and Fig. 1). No correlation was found between total population and shipping contributions to $PM_{2.5}$, nor any clear spatial pattern across the Mediterranean basin. Contributions were coincidentally the same (1.0 μg/m^3) in the two largest cities in the study (Barcelona and Athens). It should be noted that two studies were found for Barcelona (Amato et al., 2009; Pérez et al., 2016), with higher contributions reported during 2003–2007 (1.8 μg$PM_{2.5}$/m^3, 6% of $PM_{2.5}$, oil combustion source; Amato et al., 2009) than in 2011 (1.0 μg$PM_{2.5}$/m^3, 6% of primary $PM_{2.5}$, fuel oil combustion source; Pérez et al., 2016). The latter results (Pérez et al., 2016), for the year 2011, were used for Barcelona in the following sections due to the availability of concurrent health data. The health impacts estimated for the years 2003–2007 are reported in Table S2. Source contributions from mixed oil combustion sources in Barcelona and Athens are reported by Amato et al. (2016).

3.2. Health impact assessment: long-term exposure

The health incidence data collected for the 8 study cities from the scientific literature and from national statistical registries are summarised in Table 2, together with shipping and oil combustion contributions to $PM_{2.5}$. The endpoints assessed and the concentration-response functions (CRFs) applied for the health impact assessment (HIA) are summarised in Table 1. Whenever more than one risk estimate was available (e.g., Beelen et al., 2014; Hoek et al., 2013), results are shown as a range (Table 3). Confidence intervals (CI 95%) are reported in parenthesis.

All-cause (non-accidental) mortality for the baseline scenario, i.e., the cases attributable to $PM_{2.5}$ from all emission sources (including

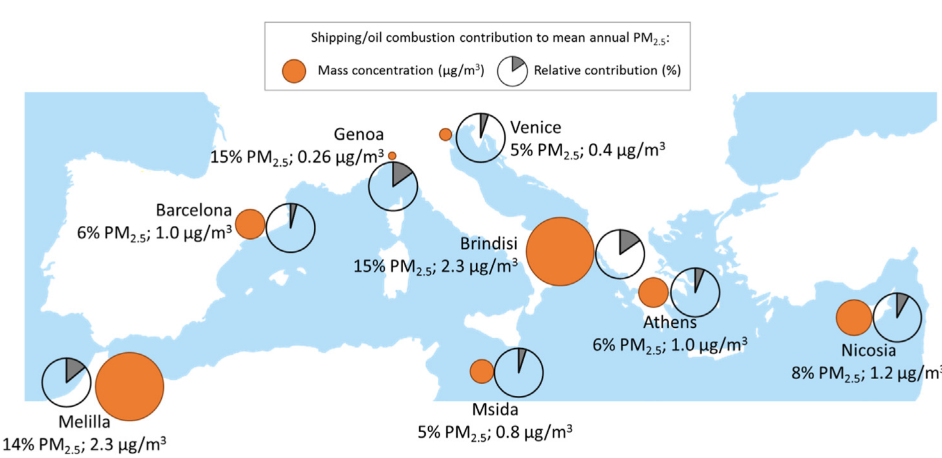

Fig. 1. Shipping/oil combustion contributions to mean annual $PM_{2.5}$ concentrations in the 8 cities selected for the health impact assessment (Nicosia for years 2007–2009, Brindisi 2010, Genoa 2011, Venice 2007, Msida 2016, Melilla 2007–2008, Athens 2011–2012, Barcelona 2011), in terms of mass concentrations (μg/m^3) and relative contributions (%). The size of the mass concentration dot is proportional to the concentration. Percentages refer to mean annual $PM_{2.5}$ mass concentrations at the study location and during the study period.

shipping), for each study year (or average for 2 years in Nicosia, Athens and Melilla) for ages 0–99 ranged between 828 (Brindisi) and 25,938 (Athens), with the exception of Msida (72) (Table 3). In relative terms (compared to the total population), exposure to $PM_{2.5}$ from all sources accounted for 0.3–1.8% of the baseline mortality in most of the cities, with a higher incidence rate (3.9%) in Athens. The incidence rates obtained for the study cities were higher than those calculated at country level (0.06% for Cyprus, Malta and Spain, 0.10% for Italy, and 0.11 for Athens; EEA, 2019), which could be related to the higher $PM_{2.5}$ concentrations (and thus, health impacts) in major cities when compared to the national average. However, this could also be linked to different exposures (population distribution at national level being different from that at urban level) and/or from different risk factors used in the different studies. Hospital admissions due to diseases of the respiratory system (attributable to $PM_{2.5}$ from all sources) for ages 0–99 varied between 127 (Melilla) and 1770 (Nicosia), and for the cardiovascular system between 392 (Melilla) and 3040 (Genoa).

The health impacts of long-term exposure to shipping and oil combustion contributions are reported in Table 3. They were quantified by applying the HIA model to a policy scenario in which the $PM_{2.5}$ reduction estimated was the contribution from shipping or oil combustion quantified by source apportionment (Table 2). It is necessary to highlight here the relevance of health impact assessment as a tool for comparative analysis between scenarios (in this case, cities), and not for quantification of health impacts in absolute terms. In this framework, long-term exposure to $PM_{2.5}$ sourcing from shipping or oil combustion was estimated to account for between 7 and 302 premature deaths annually (all-cause mortality) in each of the 8 study cities (with the exception of Msida, < 1) (Table 3, Fig. 2). In total for the 8 cities, this would account for 432 (95% CI: 219–652) premature deaths per year, which adds up to 5.5 premature deaths per year for every 100.000 inhabitants (Fann et al., 2019). The estimated burden of shipping emissions may be compared to that of other emission sources, for example vehicular traffic: in Malmö (Sweden), exhaust-free transport could avoid between 55 and 93 premature deaths annually in the city (Malmqvist et al., 2018), by reducing mean annual $PM_{2.5}$ (from 0.1 to 1.7 $\mu g/m^3$) and NO_2 (from 0.6 to 11.8 $\mu g/m^3$). In relative terms, vehicular traffic in Malmö would thus account for a higher number of premature deaths (28.5 premature deaths/100.000 inhabitants) than shipping emissions in the 8 study cities. Similarly, Castro et al. (2017) quantified the health benefits of implementing a clean air plan in the agglomeration of Lausanne-Morges (Switzerland), which would prevent 26 premature deaths annually (with a reduction of 3.3 $\mu g/m^3$ of PM_{10}). Once again, the impact of the sources targeted by this plan (vehicular transport, energy and industry) would account for a higher relative number of premature deaths per 100.000 inhabitants (8.9) than shipping in the 8 Mediterranean cities under study. In absolute terms, all of these results show a comparable order of magnitude.

In the Mediterranean, results evidenced two clearly distinct groups of cities: Barcelona and Athens, with > 100 premature deaths/year, and Nicosia, Brindisi, Genoa, Venice, Msida and Melilla, with one order of magnitude lower impacts on all-cause mortality. As evidenced in Fig. 2, health impacts are not directly related with source contributions in terms of $\mu g PM_{2.5}/m^3$ of shipping emissions, but are instead

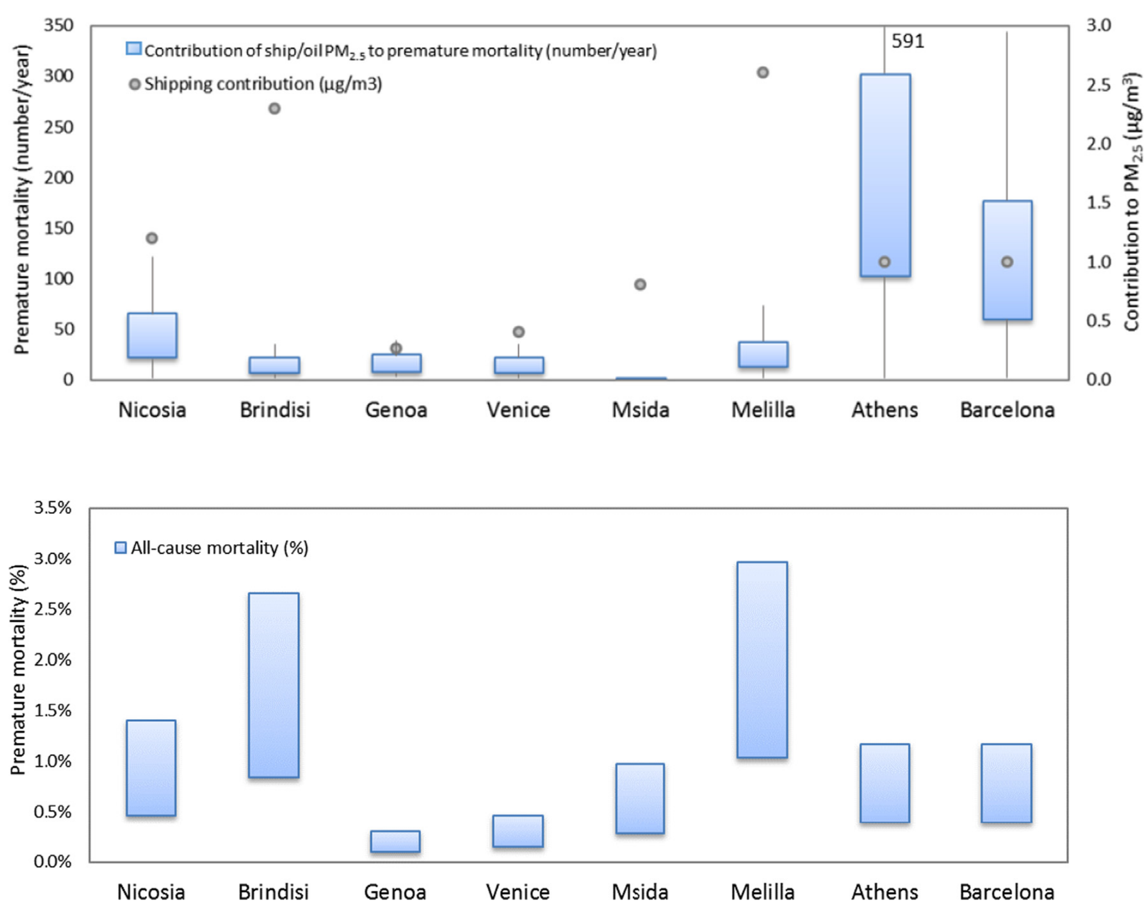

Fig. 2. Top: estimated impact (premature mortality, with 95% confidence interval) of shipping and oil combustion emissions per year in each city (left axis); contribution of shipping and oil combustion emissions to annual mean $PM_{2.5}$ concentrations ($\mu g/m^3$) in each city (right axis). Bottom: contribution of premature mortality from long-term exposure to shipping/oil combustion emissions, relative to all-cause mortality for the baseline scenario, i.e., cases attributable to $PM_{2.5}$ from all emission sources (including shipping). Results for the following cities and time periods: Nicosia for years 2007–2009, Brindisi 2010, Genoa 2011, Venice 2007, Msida 2016, Melilla 2007–2008, Athens 2011–2012, Barcelona 2011.

influenced by population and incidence rates (which are influenced by other emission sources, e.g., vehicular traffic) and this, in turn, results in the highest overall premature mortality in the larger cities despite them not reporting the highest shipping or oil combustion contributions (Fig. 2, top). Conversely, in relative terms (Fig. 2, bottom), premature mortality due to exposure to $PM_{2.5}$ from oil combustion sources was highest in Brindisi and Melilla, accounting for 2.7% and 3.0% respectively of the $PM_{2.5}$ baseline all-cause mortality. This probably reflects the influence of other emission sources on premature mortality in the larger cities (e.g., vehicular traffic in Barcelona or Athens) and highlights the fact that the influence of shipping emissions is higher, in relative terms, in smaller cities (e.g., Melilla and Brindisi).

Hospital admissions due to diseases of the respiratory and cardiovascular systems as a result of exposure to $PM_{2.5}$ from shipping or oil combustion sources are also reported in Table 3 (hospital admission data not available for Barcelona or Athens due to administrative constraints). Exposure to shipping/oil emissions had a lower impact on the number of hospitalisations in each city when compared to premature mortality, ranging between < 1 and 9 respiratory and < 1 and 3 cardiovascular hospital admissions, per year, in Nicosia, Brindisi, Genoa, Venice, Msida and Melilla (Fig. 3). The largest numbers of hospital admissions per year were observed for Nicosia and Brindisi, for both endpoints (Fig. 3 top, respiratory; bottom, cardiovascular admissions). Relative differences were observed between Genoa/Venice/Melilla and Msida: respiratory admissions were higher in Msida, while cardiovascular admissions were higher in the former three cities (Genoa/Venice/Melilla). Finally, data on cardiovascular and respiratory hospital admissions were not available for Athens or Barcelona for the study years selected, but they were available for Barcelona for 2003–2007 (Table S2).

The variability in health outcomes observed between cities, in terms of premature mortality as well as of hospital admissions, could be a reflection of differences in age distributions of the populations. Population demographics, i.e., skewed towards adult or infant groups, would impact the outcomes of the HIA. However, in order to account for these differences age-specific risk estimates would need to be used (e.g., Beelen et al., 2014, for adult populations), which were not found in the literature for this study region. Further research would be necessary to address this issue.

3.3. Scenario analysis

In view of the reduction in the sulphur content of fuels used onboard ships after the year 2020 established by MARPOL Annex VI (0.5% or lower), Sofiev et al. (2018) modelled the expected reduction in ship-sourced $PM_{2.5}$ concentrations at global scale. It should be noted that this reduction will mostly affect secondary aerosol formation from the lower S content in fuels, as opposed to primary emissions. The $PM_{2.5}$ reductions expected for each of the 8 study cities, which were extracted from Sofiev et al. (2018) and were used as input for the HIA model, are reported in Table 4. It is evident that the modelled reductions are low in terms of absolute concentrations (< 1 μg/m^3 for most cities), lower than the detection limit of most conventional particle monitoring instrumentation, and therefore their uncertainty should be taken into consideration.

The health benefits in each city derived from reducing the mean annual ship-sourced $PM_{2.5}$ concentrations by the concentrations reported in Table 4 (modelled by Sofiev et al., 2018), were estimated for

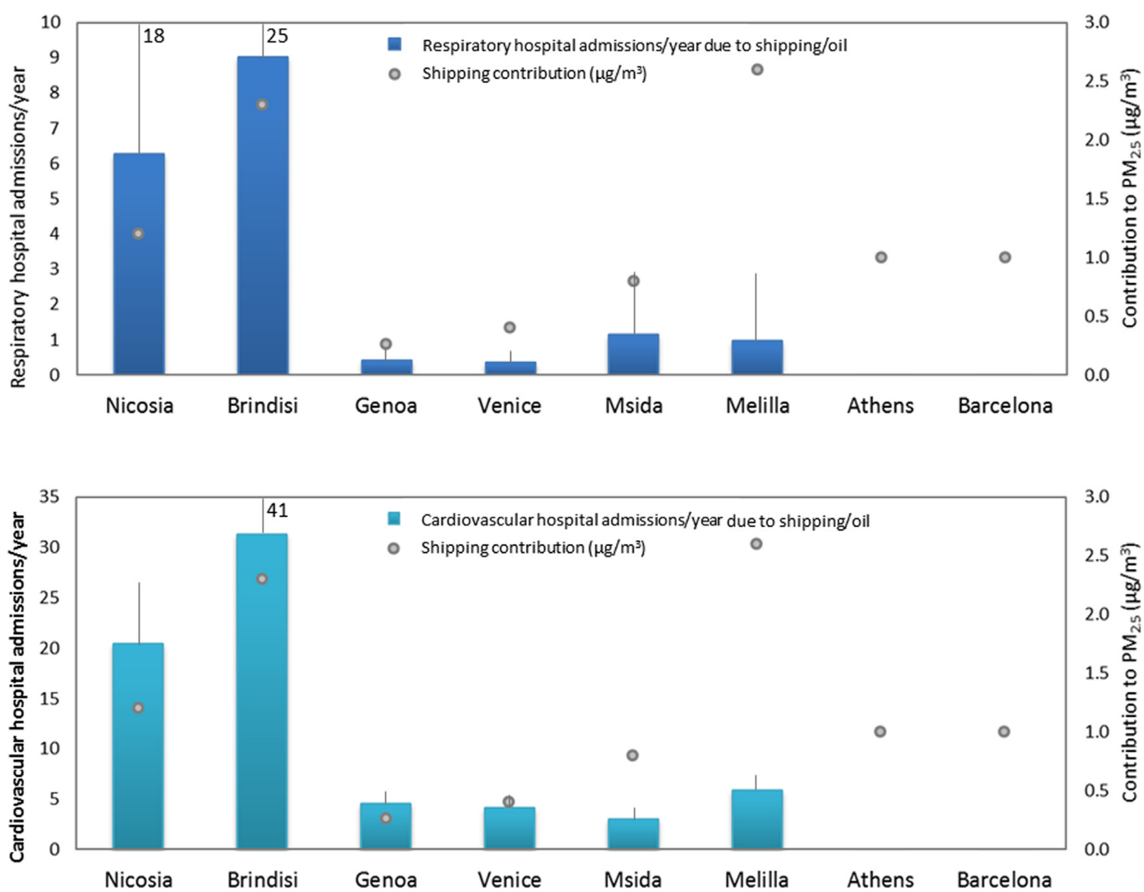

Fig. 3. Estimated impact on hospital admissions (with 95% confidence interval) due to respiratory disease (top) and to cardiovascular disease (bottom) of shipping and oil combustion emissions per year in each city (left axis); contribution of shipping and oil combustion emissions to annual mean $PM_{2.5}$ concentrations (μg/m^3) in each city (right axis). Results for the following cities and time periods: Nicosia for years 2007–2009, Brindisi 2010, Genoa 2011, Venice 2007, Msida 2016, Melilla 2007–2008. Hospital admission data were not available for Athens or Barcelona due to administrative constraints.

Table 4
Scenario analysis: comparison for the current scenario (based on literature data) and for the 2020 scenario (with more stringent sulphur content standards, 0.5% S mass content) between the estimated health outcomes (all-cause mortality, and hospital admissions, with 95% confidence intervals) due to long-term exposure to shipping emissions. Hospital admission data were not available for Athens or Barcelona due to administrative constraints.

Input data		HIA model output			
City	$PM_{2.5}$ reduction*	Endpoint	Outcomes due to shipping/oil source (N)**	Reduction due to 2020 scenario (N)***	% reduction
Nicosia	0.27 μg/m^3	All-cause mortality	22–66	9.9 (4.9–14.8)	23%
		Respiratory hospital admissions	6.3	1.4	23%
		Cardiovascular hospital admissions	20.4	4.6	23%
Brindisi	0.02 μg/m^3	All-cause mortality	7–22	0.1 (< 0.1–0.2)	1%
		Respiratory hospital admissions	9.0	0.1	1%
		Cardiovascular hospital admissions	31.3	0.3	1%
Genoa	0.02 μg/m^3	All-cause mortality	8–25	1.3 (0.7–1.9)	8%
		Respiratory hospital admissions	0.4	< 0.1	8%
		Cardiovascular hospital admissions	4.6	0.4	8%
Venice	0.01 μg/m^3	All-cause mortality	7–22	0.4 (0.2–0.6)	2%
		Respiratory hospital admissions	0.4	< 0.1	3%
		Cardiovascular hospital admissions	4.2	0.4	8%
Msida	0.23 μg/m^3	All-cause mortality	0.2–0.7	0.1 (< 0.1–0.2)	26%
		Respiratory hospital admissions	1.1	0.3	29%
		Cardiovascular hospital admissions	3.0	0.9	29%
Melilla	0.07 μg/m^3	All-cause mortality	13–37	10.7 (5.4–16.0)	45%
		Respiratory hospital admissions	1.0	0.4	42%
		Cardiovascular hospital admissions	5.9	2.5	42%
Athens	1.10 μg/m^3	All-cause mortality	102–302	16.2 (8.1–24.3)	8%
		Respiratory hospital admissions	N.A.	N.A.	N.A.
		Cardiovascular hospital admissions	N.A.	N.A.	N.A.
Barcelona	0.08 μg/m^3	All-cause mortality	60–177	8.3 (4.2–12.5)	7%
		Respiratory hospital admissions	N.A.	N.A.	N.A.
		Cardiovascular hospital admissions	N.A.	N.A.	N.A.

* From Sofiev et al. (2018).
** Outcomes due to shipping/oil (N): number of outcomes attributable to shipping/oil emissions, including the confidence interval (CI), for the current scenario.
*** Reduction due to 2020 scenario (N): reduction in the number of outcomes (including the confidence interval), for the 2020 scenario.

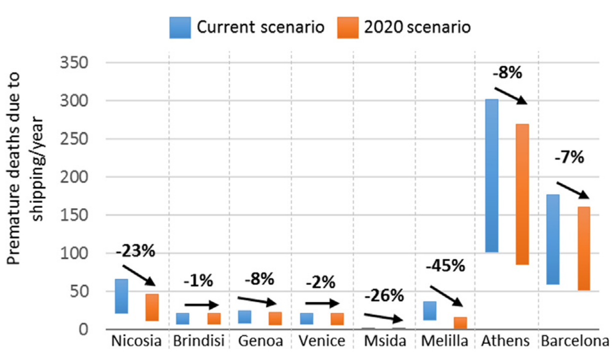

Fig. 4. Estimated premature mortality (annual) from long-term exposure to shipping and oil combustion emissions: current scenario (shipping contributions to $PM_{2.5}$ extracted from literature source apportionment studies) and scenario for the year 2020 after implementation of cleaner fuels by MARPOL Annex VI (from Sofiev et al., 2018). Percentages indicate the reduction in premature deaths due to shipping/year from the current to the 2020 scenario.

the same endpoints described above: all-cause mortality, respiratory and cardiovascular hospital admissions. Similarly to the previous section, premature mortality avoided was calculated based on two different CRFs (Beelen et al., 2014; Hoek et al., 2013) and therefore a range of potentially avoided premature mortality is presented in Table 4 and Fig. 4. Results showed reductions in premature all-cause mortality ranging from < 1 (Brindisi, Venice, Msida) to close to 10 (Nicosia, Melilla, Barcelona) and 16 (Athens) cases per year, while reductions in hospital admissions were considerably lower. In total, for the 8 cities under study, it was estimated that 47 (CI 95%: 24–71) premature deaths would be avoided as a result of the reduction in sulphur content of shipping fuels implemented in the year 2020, from a total of 432 (CI 95%: 219–652). These results are of the same order of magnitude as those reported by Rouïl et al. (2019), who estimated that the potential implementation of an emission control area (ECA) in the entire Mediterranean region could avoid nearly 1730 premature deaths each year. On average for all the cities, the implementation of the MARPOL Annex VI regulations would result in a 15% reduction in estimated premature mortality per year. Percentages varied from 1% in Brindisi to 45% in Melilla, once again evidencing the larger health benefits expected in smaller-sized cities with a higher relative impact of shipping emissions (in comparison to cities such as Barcelona or Athens, where a 7–8% reduction of premature mortality was estimated). In absolute terms, the total reduction in premature mortality would be larger in cities with larger populations.

4. Discussion

Health impact assessment (HIA) is a versatile tool which may be applied for different purposes (Wernham, 2011): in this work, it was applied to obtain a comparative assessment of the health burden of shipping emissions across Mediterranean coastal cities. The present assessment concluded that, in total for the 8 European Mediterranean coastal cities studied, exposure to shipping (or oil combustion) emissions could account for a sum of 432 premature deaths per year. On average, this implies 5.5 premature deaths per year for every 100.000 inhabitants in those cities. The impact of this emission source is lower than that of the most typical urban source, vehicular traffic, with 28.5 and 8.9 premature deaths/100.000 inhabitants in other European cities (Malmö, Sweden, and Lausanne-Morges, Switzerland; Castro et al., 2017; Malmqvist et al., 2018), but comparable in magnitude. Similarly, according to a study referring to the year 2010 (Lelieveld et al., 2015), $PM_{2.5}$ and ozone emitted by land traffic sources would have accounted for 5.8 premature deaths annually in Italy (at country level; 3519 deaths over 60.3 million inhabitants). The implementation of the IMO scenario in the year 2020, with lower sulphur-content fuels, could prevent 47 of the 432 premature deaths, annually, as a sum in the 8 cities. As also concluded by Hänninen et al. (2014), these results evidence that current methods and data allow to produce targeted assessments to support policy evaluation and resource allocation aiming to reduce population exposure to key air pollution emission sources, such as in this case maritime transport.

The main strengths of this study are the estimation of health impacts using a single, comparable model across all 8 cities (BenMap CE), and the fact that receptor modelling results were reviewed from the literature to estimate comparable baseline ship-sourced aerosol concentrations.

Key limitations and uncertainties of this approach must be taken into account. The main issue is the fact that health estimates are sensitive to the dose-response functions used. In this work this was dealt with by: (1) using the two most geographically-relevant studies (Beelen et al., 2014; Hoek et al., 2013), referring to European populations, and (2) reporting the health impacts as ranges including the impacts calculated with both dose-response functions. It should be noted that this was not a sensitivity analysis. How these results could compare with others obtained using integrated exposure-response functions (GBD, GEMM functions; (Burnett et al., 2018; Fantke et al., 2019) is a relevant topic for future research. Specifically for the Beelen et al. (2014) study, it should be taken into account that it refers to an adult population whereas in the present work the risk estimates were applied to the total population (ages 0–99), therefore introducing a bias in the analysis.

Exposure classification by means of different models should also be further explored. In this work we chose to focus on receptor modelling studies to quantify shipping contributions to ambient air quality, but dispersion models could also have been used. Using receptor models favoured the more specific source identification based on experimental data collected at each study location, whereas using a dispersion model would have increased the spatial density of the air pollution and health impact results. Going one step further, the accuracy and representativity of HIA would also increase by using exposure concentrations as input (de Hoogh et al., 2018, 2014). The complexity in discriminating between shipping emissions and oil combustion should also be taken into account.

To conclude, the HIA presented in this work estimated that long-term exposure to ship-sourced $PM_{2.5}$ accounted for 430 (95% CI: 220–650) premature deaths per year, in the 8 cities assessed. The more stringent standards in 2020 would reduce the number of $PM_{2.5}$-attributable premature deaths by 15% on average.

CRediT authorship contribution statement

M. Viana: Conceptualization, Formal analysis, Methodology, Writing - original draft, Writing - review & editing, Supervision. **V. Rizza:** Data curation, Formal analysis. **A. Tobías:** Writing - review & editing. **E. Carr:** Writing - review & editing. **J. Corbett:** Conceptualization, Writing - review & editing. **M. Sofiev:** Resources. **A. Karanasiou:** Writing - review & editing. **G. Buonanno:** Writing - review & editing, Funding acquisition. **N. Fann:** Writing - review & editing.

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.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2020.105670.

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