Cleaner fuels for ships provide public health benefits with climate tradeoffs

Mikhail Sofiev1, James J. Winebrake2, Lasse Johansson1, Edward W. Carr3, Marje Prank1, Joana Soares1, Julius Vira1, Rostislav Kouznetsov1, Jukka-Pekka Jalkanen1 & James J. Corbett4

We evaluate public health and climate impacts of low-sulphur fuels in global shipping. Using high-resolution emissions inventories, integrated atmospheric models, and health risk functions, we assess ship-related PM2.5 pollution impacts in 2020 with and without the use of low-sulphur fuels. Cleaner marine fuels will reduce ship-related premature mortality and morbidity by 34 and 54%, respectively, representing a -2.6% global reduction in PM2.5 cardiovascular and lung cancer deaths and a -3.6% global reduction in childhood asthma. Despite these reductions, low-sulphur marine fuels will still account for -250k deaths and -6.4 M childhood asthma cases annually, and more stringent standards beyond 2020 may provide additional health benefits. Lower sulphur fuels also reduce radiative cooling from ship aerosols by -80%, equating to a -3% increase in current estimates of total anthropogenic forcing. Therefore, stronger international shipping policies may need to achieve climate and health targets by jointly reducing greenhouse gases and air pollution.

1 Atmospheric Composition Research, Finnish Meteorological Institute, P.O. Box 503 FI-00101 Helsinki, Finland. 2 Rochester Institute of Technology, Rochester, NY 14623, USA. 3 Energy and Environmental Research Associates, LLC, Pittsford, NY 14534, USA. 4 University of Delaware, 305 Robinson Hall, Newark, DE 19711, USA. Correspondence and requests for materials should be addressed to J.-P.J. (email: jukka-pekka.jalkanen@fmi.fi) or to J.J.C. (email: jcorbett@udel.edu)
**Results**

**Overview of approach.** On the basis of 2015 Automatic Identification System (AIS) data on shipping traffic (over 65,000 IMO registered vessels and over 7.6 billion million records), we project 2020 geospatial shipping emissions inventories with and without implementation of the proposed standards. Using global chemical transport models with very high spatial and temporal resolution \((10 \times 10 \text{ km} \times 3 \text{ h})\), we evaluate the public health and climate forcing consequences of low-sulphur marine fuel policy implementation. Atmospheric transport and transformations of ship-emitted pollutants serve as inputs to modelling health effects, and to estimating direct and indirect radiative forcing potential. All computations incorporate existing regulations that limit sulphur emissions from ships in designated Sulphur Emissions Control Areas (SECA)s. We report results of advanced shipping emissions inventories for 2015 and 2020 scenarios, high-resolution spatial and temporal chemistry-transport model runs, health outcomes using linear and log-linear concentration–response functions, and radiative transfer evaluations of direct and indirect aerosol forcing changes.

**Global shipping emissions in 2015 with projections to 2020.** Global ship emissions were calculated using the Ship Traffic Emissions Assessment Model (STEAM) model\(^{19-21}\) and 2015 AIS ship traffic (see Methods). Emissions were projected to 2020 using vessel-type-specific annual growth rates (MEPC 70/5/3\(^{22,23}\)). Table 166). Annual totals for emissions from global shipping for 2015 and 2020 are presented in Table 1, comparable with the 2012 results described in the Third IMO GHG Study\(^{12}\). Projected business-as-usual (BAU) emission results of the global fleet in 2020 are consistent with those of the Third IMO GHG Study, mainly reflecting growth assumptions; as shown in Table 1, emissions reductions associated with implementation of a low-sulphur fuel standard in 2020 (2020 Action) affect both SO\(_2\) and PM (via reduction of sulphate emissions).

Implementation of the 2020 standards results in \(~75\%\) reduction in shipping SO\(_2\) emissions globally, along similar spatial distribution of emissions without the new standards (see Supplementary Figure 1). Supplementary Figure 1 also illustrates the presence of an IMO-approved sulphur emissions control area (SECA) designation for the United States (US) and Canada\(^{24}\); a Chinese domestic emissions control area (DECA) designation for the Bohai Sea, Yangtze River delta, and Pearl River delta in China\(^{25}\); IMO-approved SECA designation in parts of Europe; and additional European Union control requirements under the European Directive\(^{26}\). In the European and US/Canadian SECA,s, ships are limited to burning 0.1\% S fuel within 200 nautical miles of the coast, whereas the limit under the European Directive and in Chinese areas will be 0.5\% S from the beginning of 2020 and 2019, respectively. These SECA,s will be in force independent of the IMO decision to implement global fuel sulphur standards.

**Impact of low-sulphur standards on particulate matter.** Geospatial emissions inventories provide input into the Finnish Meteorological Institute’s (FMI) System for Integrated modelling

| Pollutant (000 tonnes) | Third IMO GHG study estimate for 2012 | 2015 estimate | 2020 BAU without IMO standard | 2020 action with IMO standard | 2010 non-shipping emissions HTAPV2/MEIC\(^{73}\) |
|------------------------|---------------------------------------|---------------|-------------------------------|-----------------------------|---------------------------------|
| NO\(_x\)                | 19,000                                | 20,100        | 21,300                        | 21,300                      | 75,310                          |
| SO\(_x\)               | 10,200                                | 11,500        | 11,000                        | 2500                        | 99,071                          |
| PM\(^a\)               | 1400                                  | 1540          | 1500                          | 770                         | 17,338                          |
| CO\(_2\)               | 938,000                               | 814,000       | 860,000                       | 870,000                     | —                               |
| Fuel Usage\(^b\)       | 254,000 (t-d) 300,000 (b-u)           | 263,000       | 277,000                       | 274,000                     | —                               |

\(^a\) PM estimates in this work include speciated fractions for modelling, including sulphate, elemental carbon, organic carbon, and ash, as appropriate for the source characteristics. For ships, PM includes sulphate formed from gaseous emissions of SO\(_x\). Nearly all ship-emitted PM falls within the PM\(_{2.5}\) size range.

\(^b\) The Third IMO GHG Study reported both top-down (t-d) and bottom-up (b-u) estimations; STEAM methods are activity based, bottom-up.

![Table 1 Summary of emissions (and fuel consumption) of this work using STEAM for 2020 compared with Third IMO GHG Study (Smith et al.)\(^{12}\) and all non-shipping emissions](image-url)
of Atmospheric composition (SILAM). SILAM was run at geospatial resolution of 10 x 10 km and a temporal resolution of 3 h (see Methods section). We used SILAM to predict pollutant concentrations under the following two scenarios: (1) business-as-usual (BAU), which assumes no implementation of a global, low-sulphur fuel standard; and (2) 2020 Action, which assumes on time implementation of the IMO’s 0.5% S low-sulphur fuel standard. Figure 1 shows the spatial distribution of atmospheric PM$_{2.5}$ near-surface concentrations under the BAU scenario; Fig. 2 presents the difference between the BAU scenario and the 2020 Action scenario. Related concentration maps provided in Supplementary Figure 1 shows consistency with previous distributions at a global level.

Considering all PM sources, typical concentrations of PM$_{2.5}$ amount to a few $\mu$g m$^{-3}$, with much higher levels in arid areas and regions with strong fire activity (Fig. 1). Over the ocean, the major component of PM$_{2.5}$ is sea salt (over 50–75%), whereas over land terrestrial emissions are the major contributor. Ship contributions are visible mainly in the open ocean and over the busiest sea-lanes. Stricter limits on sulphur content in ship fuel decrease sulphate concentrations (2–4 $\mu$g SO$_4$ m$^{-3}$ annual mean), leading to significant reduction of PM$_{2.5}$ in the vicinity of busy ship lanes (Fig. 2). However, total aerosol load reduction is limited since sulphates constitute on average less than 15% of total PM in the air (compare concentrations in Figs. 1 and 2).

**Fig. 1** Mean annual PM$_{2.5}$ concentrations from all sources. Model results showing mean annual PM$_{2.5}$ concentrations in micrograms per cubic meter from all sources and with business-as-usual ship emissions in 2020.

**Fig. 2** Reduction in annual PM$_{2.5}$ concentrations due to low-sulphur fuel standards. Model results showing the reduction in annual PM$_{2.5}$ concentrations in micrograms per cubic meter due to the implementation of the International Maritime Organization’s global fuel sulphur standard in 2020.
Chemical links between sulphate and nitrate species resulted in some ammonia becoming available for forming ammonium nitrate, partly offsetting the sulphate aerosol reductions. However, this effect is small and the offset only exceeds 1% around the east coast of China, with a maximum of 15% over the Yellow Sea.

**Health impacts of global shipping.** We apply concentration–response (C–R) functions to 2020 population projections to estimate health impacts due to ship emissions in the BAU case and the 2020 Action case (see Methods section). We calculate adult mortality from lung cancer and cardiovascular disease and childhood asthma morbidity, and report results using a linear C–R function as discussed in Lepeule et al.\textsuperscript{31}, and Zheng et al.\textsuperscript{31, 32}. The vast majority of PM\textsubscript{2.5} exposure concentrations in our study area represent conditions similar to those in the Six Cities Study, indicating that functions derived from that study can also apply to our study.

Total premature mortality due to shipping in the 2020 BAU case is 403,300 per year (range of 212,300–595,400 based on the 95% confidence interval for relative risk); BAU mortality distribution is shown in Fig. 3. Total avoided premature mortality in 2020 with implementation of the low-sulphur fuel standards is expected to be 266,300 per year (range of 138,500–395,700), a reduction of ~34%; avoided mortality distribution is shown in Fig. 4. Childhood asthma morbidity due to shipping declines by 54%, from 14 million children affected in the BAU case, to 6.4 million children in the 2020 Action case (see Supplementary Note 1, Supplementary Figure 2. Detailed results are in Table 2, and Supplementary Table 1 and Supplementary Table 2 report regional results.

More than 97% of the adult mortality benefits from ship emissions reductions will be in Asia (80%), Africa (12%), and Latin America and the Caribbean (5%). More than 98% of the childhood morbidity benefits from ship emissions reductions also occur in Asia (54%), Africa (33%), and Latin America and the Caribbean (12%). The different distributions are primarily due to the different distributions of adult and youth populations among nations. Europe, North America, and Oceania combined will receive <3% and <2% of the mortality and morbidity benefits of the global sulphur standard, respectively. This is primarily due to
Table 2 Estimated annual health impacts of global shipping in 2020 in the BAU case and the 2020 Action case with IMO low-sulphur fuel standard, where parentheses indicate 95% confidence intervals based on relative risk calculations

| Scenario results (linear C–R model) | Mortality estimate (annul premature adult deaths*) | Childhood asthma (million cases) |
|-------------------------------------|--------------------------------------------------|---------------------------------|
| BAU 2020 (No implementation of global 0.5% S fuel standard) | CV mortality 349,000 (CI 95%: 200,300; 501,800) | 14.0 (CI 95%: 7.5; 21.0) |
|                                      | LC mortality 54,300 (CI 95%: 12,000; 93,600) |                                |
|                                      | Combined mortality 403,300 (CI 95%: 212,300; 595,400) |                                |
| 2020 Action (Implementation of global 0.5% S fuel standard in 2020) | CV mortality 226,800 (CI 95%: 129,800; 327,000) | 6.4 (CI 95%: 4.1; 11.5) |
|                                      | LC mortality 39,500 (CI 95%: 8,700; 68,700) |                                |
|                                      | Combined mortality 266,300 (CI 95%: 138,500; 395,700) |                                |
| Health benefit of 2020 Action | Avoided mortality$^a$ CV: 122,200 (CI 95%: 70,500; 174,800) | 7.6 (CI 95%: 3.4; 9.6) |
|                                      | LC: 14,800 (CI 95%: 3,300; 24,900) |                                |
|                                      | Combined: 137,000 (CI 95%: 73,800; 199,700) |                                |

CV cardiovascular disease, LC lung cancer, CI 95% 95% percent confidence interval

* Values for annual premature mortality are rounded to nearest 100; values for annual childhood asthma morbidity rounded to nearest 100,000

$^a$ Differences between avoided health impacts and scenario differences attributed to rounding

Our assessment of the global health burden due to air pollution from ships is much higher than prior assessments primarily due to improved geospatial resolution of global models, updated inputs, and the use of linear C–R functions (see Methods section). First, by applying higher spatial and temporal resolution using the 2020 STEAM inventories and SILAM chemistry transport model, we more precisely assess proximal exposure concentrations, better quantifying peak exposure to at-risk communities (see Supplementary Note 2, Supplementary Figure 3). Second, the updates provided by Lepeule, et al., increase the health risks attributable to PM compared to previous assessments; e.g., the attributable fraction for lung cancer due to PM$_{10}$ exposure increases by a factor of 2.5. Third, we adopt the so-called linear relative risk function by Lepeule, et al., which differs from the log-linear functions of Pope, et al., and Ostro$^b$, used in our previous health burden studies. The linear formulation produces characteristically higher health burden estimates for the range of ship pollution concentrations modelled (see Methods section and Supplementary Note 1). At higher concentrations, the combination of updated β coefficients and C–R functional form can increase the in-cell health burden estimate by orders of magnitude. Supplementary Table 3 presents results using log-linear C–R function. Supplementary Tables 4 and 5 present regional mortality and asthma results using log-linear C–R function, respectively. Lung cancer mortality using the linear function is more than three times the log-linear C–R results in Supplementary Table 3, and more than seven times the lung cancer estimates using coarser resolution inputs in Winebrake et al.$^c$. (Note that Winebrake et al.$^c$ computed premature cardiopulmonary mortality and cannot be compared directly with cardiovascular mortality as discussed in Methods section.)

Our linear function BAU 2020 estimates in the East Asia region are an order of magnitude higher than the health burden estimated for 2013 by Liu et al.$^d$ (238,100 vs. 24,000 air pollution deaths from shipping). This is clearly a result of the different choices of C–R functions, where Liu et al. use an assortment of relative risk functions within their exposure–response model based on Burnett et al.$^e$. For our 2020 Action case, global controls for sulphur standards would reduce the shipping health burden in the East Asia domain of Liu et al. by 25% (with 34% reduction of all Asia as shown in Supplementary Table 1).

Calculated using the log-linear C–R function in Supplementary Note 3 are 45% higher than the health burden estimated by Liu et al., and their estimates fall within the log-linear 95% confidence interval.

Ship pollution matters in the context of total health impacts from ambient (outdoor) air pollution, with or without the new standards. Despite demonstrated health benefits associated with a low-sulphur fuel standard, ship traffic using cleaner fuel will produce air pollution impacting mortality and morbidity in proximal coastal communities. The World Health Organization (WHO) estimates ~3 million deaths in 2012 attributed to ambient air pollution$^{36}$, with 1.48 million attributed to lung cancer and cardiovascular disease; the World Bank estimates 5.5 million deaths in 2013 attributed to both household and ambient air pollution (2.9 million due to ambient PM$_{2.5}$)$.^{37}$ Other scientific peer-reviewed journal papers report similar health burdens due to ambient PM, with estimates ranging from 2.2 to 3.3 million deaths annually$^{38–42}$.

In terms of childhood asthma, the results are also quite significant. Under our BAU case and linear risk function, shipping emissions lead to ~14 million childhood asthma cases annually. Further, even with the low-sulphur fuel standards proposed by IMO, ships are still responsible for ~6.4 M childhood asthma cases annually. The 2014 Global Asthma Study estimated "as many as 334 million people in the world have asthma," and statistics indicate 26% of the world population is 14 years or younger$.^{43}$ Without adjusting for higher prevalence for asthma among young and old persons, direct application of population statistics suggest 86 million children could suffer from asthma, and our linear–function results suggest that shipping is currently responsible for ~16% of these cases (~7.5% if the new IMO standards are implemented).

Most synthesis reports reference sources using log-linear C–R functions and matched β-coefficients$^{35–42}$, and some meta-analyses do not explicitly report underlying C–R functions informing their health assessments. We can make bounding comparisons of the contribution of ship pollution to global cardiovascular and lung cancer mortality and asthma morbidity assuming that our log-linear specifications compare better with other studies (see Supplementary Note 3). Under the BAU scenario, shipping accounts for about 3.6% of WHO and World Bank mortality estimates from ambient air pollution, and nearly 7% of lung and cardiovascular disease mortality; shipping pollution under the BAU scenario contributes to ~3.6% of childhood asthma morbidity. Under the 2020 Action scenario,
shipping would contribute about 2% of WHO and World Bank air pollution total mortality from ambient air pollution, about 4% of the lung cancer and cardiovascular disease mortality, and about 2.5% of childhood asthma. Cleaner ship fuels reduce exposure to ambient PM$_{2.5}$ air pollution, reducing total ambient air pollution deaths from lung cancer and cardiovascular disease by ~2.6% and reducing total childhood asthma by ~3%. Along major shipping lanes and ports near densely populated coastal regions, these health impacts are much greater.

**Radiative forcing due to low-sulphur fuel.** Global low-sulphur fuel standards also impose climate forcing consequences. Sulphates from ship emissions reduce radiative flux at the earth's surface (i.e., contribute a cooling effect), previously estimated in the range of $-47$ to $-8$ mW m$^{-2}$ (direct radiative effect) and $-600$ to $-38$ mW m$^{-2}$ (indirect radiative effects)\textsuperscript{15, 30, 44, 46}. Our analysis covered direct and the first-indirect effects (i.e., impact on the number concentration and size of cloud droplets and, consequently, cloud albedo via the Twomey effect). We estimated these effects for a variety of assumptions regarding the aerosol properties and meteorological conditions. We found that the global average of the direct radiative forcing of ship-emitted sulphates and nitrates strongly depends on the assumption of the single-scattering albedo (SSA) of the particles, consistent with Lohman et al.\textsuperscript{47} and Lesins et al.\textsuperscript{47, 48}. The cooling under the BAU scenario is about $-5.2$ mW m$^{-2}$ and $-1.5$ mW m$^{-2}$ for sulphate and nitrate scattering aerosols, respectively (Table 3). Comparing BAU and 2020 Action scenarios, reduction of sulphates will lead to over $4$ mW m$^{-2}$ of reduced cooling, whereas the nitrate concentrations will increase due to the compensation effect, thereby adding extra cooling of almost $0.2$ mW m$^{-2}$. The net reduced direct cooling will be $3.9$ mW m$^{-2}$ for scattering particles.

Estimation of the indirect effects remains very uncertain and depends on the assumption on the distribution of the ship-induced sulphate aerosols. We quantified the first-indirect effect using empirical relationships between the sulphate and sea salt concentrations derived from several cloud chemistry observation campaigns\textsuperscript{19–51} (see Methods section). This allowed estimating the sensitivity of the forcing in terms of the vertical in-cloud distribution of the sulphate aerosols. In order to compute an envelope of the possible variations, we considered two extreme cases: (1) when the sulphates are well mixed with the cloud water, i.e., the in-cloud convection mixes them perfectly; and, (2) when the in-cloud convection does not affect the sulphate vertical profiles. In the latter case, the sulphates and cloud-water vertical profiles are largely detached, and the upper levels of the clouds are practically not polluted by sulphates. For the well-mixed case, the cooling by ship-induced sulphates amounts to $-86$ mW m$^{-2}$ for our BAU scenario and $-19$ mW m$^{-2}$ for 2020 Action scenario. This means that the reduced cooling due to these first-indirect effects is $67$ mW m$^{-2}$.

Sensitivity simulations for direct and indirect effects highlighted several parameters affecting the final estimates. The direct radiative effect is strongly sensitive to aerosol optical features. The ship-induced sulphates and nitrates in nature are not pure scattering aerosols due to emissions of black carbon as part of PM. Black carbon serves as condensation centres attracting SO$_2$ and creating internally mixed particles with black cores coated by sulphates. Sensitivity computations suggested that at SSA of 0.975 the direct cooling effect of ship plumes is zero, whereas darker aerosols are the warming agents (see Supplementary Figure 5, for results with SSA = 0.999, 0.99, and 0.95). The tipping point at SSA $= 0.975$ leads to all-zero direct forcing practically everywhere\textsuperscript{48}. Therefore, the change of the direct radiative forcing due to low-sulphur standards will strongly depend on the technologies used for reaching the limitations, specifically how control technologies affect black carbon emissions\textsuperscript{2, 33}.

Sensitivity of the indirect effect to the vertical profile of in-cloud sulphates seems to be most important to our estimates. In the detached sulphate profiles, the dominant effect of upper cloud layers, which are practically not polluted by sulphates, determines the overall cloud albedo regardless of the pollution at the cloud bottom, resulting in the lower estimate. Therefore, if most of sulphates are located near the cloud bottom, the reduced cooling is much less than for the well-mixed case: $17$ mW m$^{-2}$ instead of $69$ mW m$^{-2}$ (see Supplementary Note 4, Supplementary Figure 6). With high-resolution simulations, however, the well-mixing assumption is closer to observations, especially in tropical conditions, where horizontal size of the convective clouds is indeed of the order of $10$ km. Therefore, the best estimate from this study is $69$ mW m$^{-2}$.

The above results are generally consistent with other literature\textsuperscript{15, 46, 54, 55}. Thus, Eyring et al.\textsuperscript{54} roughly estimated direct cooling effect of sulphates of $-11$ to $-26$ mW m$^{-2}$ in 2030 depending on traffic scenarios\textsuperscript{54}, which is well in line with our BAU case plus the increase of ship traffic by 2030. Fuglesvedt et al.\textsuperscript{46} estimated indirect negative forcing from ship sulphate aerosols as $-66$ mW m$^{-2}$ compared to preindustrial level (uncertainty range ($-114$, $-38$) mW m$^{-2}$), which also compares well to our range of the first-indirect effect ($-86$, $-22$) mW m$^{-2}$. Capaldo et al.\textsuperscript{36} suggested $-110$ mW m$^{-2}$, but the evaluation against SO$_2$ mixing ratio observations showed almost an order of magnitude of over-estimation. A study of Dalsoren et al.\textsuperscript{57} considered a similar set of ship emission scenarios and estimated the total loss of sulphate-induced cooling at $-70$ mW m$^{-2}$ with 10–15% variability depending on scenario of ship emission in 2030, which is in close agreement with our estimates. However, Dalsoren et al. suggested that direct and indirect effects are similar in amplitude, which differs from our and most other studies, which showed ~10-fold dominance of indirect effect(s).

Our BAU 2020 direct radiative forcing estimates are also consistent with the estimates by Liu et al.\textsuperscript{37} in East Asia\textsuperscript{5}. This work estimates the BAU direct forcing in East Asia averages $-23$ mW m$^{-2}$ (standard deviation of $-14$ mW m$^{-2}$), with a region peak and minimum forcing of $-55$ mW m$^{-2}$ and $-1.5$ mW m$^{-2}$, respectively. We identify two local maxima for sulphate and nitrate radiative forcing (Fig. 3), one in the East Asia domain aligned with Liu et al., and a second region with greater maxima in the South China Sea. Liu et al. evaluated AIS-derived shipping only within their East Asia study domain, which included only a portion of the shipping traffic in the South China Sea. Our global AIS-derived shipping domain extends the major shipping lanes.

### Table 3 Summary of change in radiative forcing due to aerosol effects in BAU and 2020 Action scenarios

| Radiative forcing (mW m$^{-2}$) | Direct effect | Indirect effect | Net aerosol forcing$^a$ |
|-------------------------------|---------------|----------------|------------------------|
| **Scenario**                  |               |                |                        |
| BAU                           | $-6.7$        | $-86$          | $-93$                  |
| 2020 action                   | $-2.8$        | $-19$          | $-22$                  |
| **Net change**                | $3.9$         | $67$           | $71$                   |
| **Percent change in radiative forcing (positive percentage = increased warming)** |               |                |                        |
| **Shipping**                  | 75%           | 81%            | 81%                    |
| **Transport**                 | 2.1%          | 36%            | 38%                    |
| **All human activity$^b$**    | 0.2%          | 3.6%           | 3.8%                   |

$^a$ Net sums may differ from observed sum in table due to rounding
$^b$ 5% of anthropogenic radiative forcing, per Fuglesvedt\textsuperscript{46}
$^c$ Comparative calculation uses $-1.8$ W m$^{-2}$ vs. preindustrial level, per IPCC\textsuperscript{19}
from mainland China to Singapore, and therefore better represents forcing peaks associated with the full extent of regional shipping routes.

A few studies showed much larger effects than the ones above. Tronstad Lund et al.\textsuperscript{15} based on regression between total sulphur emission and radiative forcing suggested the indirect cooling effect as high as −450 mW m\textsuperscript{-2}. However, a relative reduction was estimated as 37–40% in direct and indirect aerosol forcing from fuel changes originally presented by Lauer et al.\textsuperscript{58}, about half of the results reported in this work.\textsuperscript{58} In turn, Lauer et al.\textsuperscript{58} estimated the direct effect at the same −11 mW m\textsuperscript{-2} level as came from our computations but got the indirect effect varying from −190 to −600 mW m\textsuperscript{-2} (about 75% of these attributed to sulphates) depending on ship emission inventory.\textsuperscript{58} A similar study of Righi et al.\textsuperscript{59} showed the range from −280 to −400 mW m\textsuperscript{-2} but used higher ship \textsubscript{SO}\textsubscript{x} emission of 14 Mton per year. Balkansky et al. (2010) reported shipping direct effect of −20 mW m\textsuperscript{-2}, and also pointed out the compensation by direct warming effect of road-traffic carbonaceous aerosols.\textsuperscript{44} For global estimates, including CO\textsubscript{2}, the net reduced cooling of this study amounts to nearly 35% of the total radiative forcing from transport\textsuperscript{46} and over 3% of anthropogenic radiative forcing\textsuperscript{13}. Considering the reasons for the differences from some of the previous work, one can notice a generally lower-sulphur lifetime and global mean burden in the current simulations in comparison with other studies. For example, we estimate a 0.6 Mton sulphur burden (2015), vs. 1.5 Mton sulphur burden (mean 1999–2004) assumed by Lauer, et al. Accounting for the lower emission of SO\textsubscript{x} in 2015 (111 vs. 150 Mton per year in Dentener et al.\textsuperscript{60} used by Lauer et al.), one ends up with the mean atmospheric lifetime of sulphur of about 2.8 days in the current study and 3.6 days in Lauer, et al. Species-specific lifetimes in our runs were 1.5 days for SO\textsubscript{2} (primary sink is oxidation to SO\textsubscript{4}) and 2.8 days for sulphates, i.e. dry and wet deposition of sulphates are the primary mechanisms controlling the total-sulphur burden in SILAM. These lifetimes are within the range of values reported by other models\textsuperscript{61}, with the SILAM estimate being more on the lower end. Evaluation of SILAM is outlined in Supplementary Note 5.

With our results comparing BAU and 2020 Action scenarios, Table 3 presents the estimated change in the radiative forcing from cleaner ship fuels on the following: (a) the shipping sector, (b) the transportation sector, and (c) all anthropogenic activity. The cleaner fuels will also shift the composition of the forcing aerosols. In the BAU scenario, over two-thirds of the cooling is due to sulphates, whereas the contributions of sulphates and nitrates are practically the same under the 2020 Action scenario. Similar to patterns of near-surface PM\textsubscript{2.5} concentrations, highly heterogeneous ship traffic density and meteorological conditions determine the radiative forcing patterns (Fig. 5). In particular, the most-significant regional direct radiative effect is over the Persian Gulf, where very dense emissions coincide with large fraction of cloud-free days. Shipping in Southeast Asia is stronger and leads to higher ship-induced PM\textsubscript{2.5} concentrations covering wider area (Fig. 1). However, the resulting direct radiative forcing is lower than in the Persian Gulf region due to a larger number of cloudy days. For the indirect forcing, the pattern is likely to be the opposite: the effect will be more pronounced in cloudy regions of South-East Asia.

**Discussion**

Global compliance with 2020 marine fuel-sulphur standards will reduce annual premature mortality and morbidity in nations across the globe. We demonstrate significant mortality and morbidity benefits from low-sulphur ship fuels in densely populated, major-trading nations. Additionally, even coastal nations, which are not engaged heavily in international trade will benefit, if these nations are proximal to major shipping routes. The distributional impacts associated with air pollution health burdens and the benefits of new standards should be considered in future research. Health burdens and benefits are accompanied by important changes in the regional distribution and global net radiative forcing due to ship-emitted aerosols, which can itself lead to potential health impacts associated with a warmer planet.

Even with cleaner marine fuels after 2020, shipping activity will continue to produce harmful air emissions and greenhouse gases.
Table 4 Emission factors used for this study\(^{a}\)

| Emission       | Emission factor (g kW\(^{-1}\)h, Normal (80%) load) | Emission factor (g kW\(^{-1}\)h, low (25%) load) |
|----------------|---------------------------------------------------|--------------------------------------------------|
| NO\(_2\)\(^b\) | Tier 1 engine 17 (SSD), 12.9 (MSD)\(^b\), 9.8 (HSD) | 17 (SSD), 12.9 (MSD)\(^b\), 9.8 (HSD)            |
|                | Tier 2 14.4 (SSD), 10.5 (MSD), 7.7 (HSD)          | 14.4 (SSD), 10.5 (MSD), 7.7 (HSD)               |
|                | Tier 3 3.4 (SSD), 2.6 (MSD), 2 (HSD)              | 3.4 (SSD), 2.6 (MSD), 2 (HSD)                   |
| SO\(_x\)\(^c\) | 0.1% sulphur (S) 0.48 (MDO/MGO: SFOC 250 g kW\(^{-1}\)h) | 0.54 (MDO/MGO SFOC 282 g kW\(^{-1}\)h)          |
|                | 0.5% S 2.40 (MDO/MGO: SFOC 250 g kW\(^{-1}\)h)     | 2.7 (MDO/MGO SFOC 282 g kW\(^{-1}\)h)           |
|                | 2.7% S 8.35 (HFO: SFOC 165 g kW\(^{-1}\)h)        | 9.42 (HFO: SFOC 186 g kW\(^{-1}\)h)             |
|                | CO 0.54                                            | 2.18                                             |
| PM            | 0.1% S 0.38                                        | 0.43                                             |
|                | 0.5% S 0.50                                        | 0.57                                             |
|                | 2.7% S 1.19                                        | 1.35                                             |
| CO\(_2\)^d    | HFO 515 (SFOC 165 g kW\(^{-1}\)h)                  | 580 (SFOC 186 g kW\(^{-1}\)h)                    |
|                | MDO/MGO 803 (SFOC 250 g kW\(^{-1}\)h)             | 905 (SFOC 282 g kW\(^{-1}\)h)                    |

\(^{a}\) Values only indicate the range of values applied on case-by-case basis because fuel consumption and emissions depend on engine load and specific fuel oil consumption (SFOC), calculated from vessel-specific automated identification system (AIS) data, as described in published literature for STEAM0, 21, 41-44.

\(^{b}\) As defined in MARPOL Annex VI, Regulation 13. For MSD, crankshaft rpm of 154 is assumed in this example, but engine specific values are used in each case. For Tier 0 engines, 110% of Tier I value is used.

\(^{c}\) Part of sulphur is as gaseous SO\(_2\) and part is in aerosol SO\(_4\). The emission factors listed for SO\(_2\) contain the gaseous emission part, the aerosol sulphur has been subtracted to maintain mass balance of sulphur.

\(^{d}\) SFOC changes as a function of engine load.0 The values listed include this effect and includes the differences in carbon content between HFO and MDO/MGO.

Notwithstanding the first meaningful global fuel-sulphur controls since fleets converted from coal to petroleum byproducts, energy use in global trade is expected to increase, along with air emissions from shipping\(^{12}\). Given that low-sulphur marine fuels still account annually for ~250k deaths (CI 95% ~139k; ~396k) and ~6.4 M childhood asthma cases (CI 95% ~1.4 to 11.5 million), additional reductions beyond 2020 standards may prove beneficial.

Sulphate-specific impacts quantified here cannot be posed strictly as offsetting tradeoffs. Emission reductions of sulphur aerosols by ships using cleaner marine fuels may offer collateral health and climate benefits that merit quantification beyond this work. For example, 2020 compliant fuels may enable or be accompanied with other PM\(_{2.5}\) emission reductions, such as organic carbon particles and black carbon particles which contribute to strongly warming effects\(^{62}\). Moreover, many control technologies for harmful particulates and ozone precursor emissions perform better under low-sulphur combustion conditions. In a broader context, near-term air-quality health implications and longer-term health implications related to climate forcing deserve joint consideration.

Ship emissions impacts reported here will inform evaluation of additional air pollutant health and climate impacts from shipping activities, and future shipping policies to achieve environmental stewardship goals in shipping and related transportation sectors serving a global economy. International policymaking efforts jointly pursuing air pollution health benefits and climate targets may increase the urgency for continued progress to control and reduce greenhouse gases.

Methods

Geospatial emissions inventories. For this research, we constructed geospatial shipping emissions inventories under the following two scenarios: (1) a business-as-usual (BAU) scenario, where we assume shipping emissions are not constrained by a global fuel sulphur standard; and (2) a policy implementation case (2020 Action), where the IMO MARPOL VI fuel sulphur standard goes into effect in 2020. The emissions inventory was facilitated by the use of the Ship Traffic Emission Assessment Model (STEAM), which has been used in similar types of work\(^{12, 19, 21, 41-65}\). The STEAM model combines 2015 AIS data on shipping routes and volume as well as vessel technical data from IHS Fairplay, with peer-reviewed energy use and emissions equations to construct a geospatial emissions inventory for global shipping. Propulsion energy is evaluated with the Hollenbach resistance calculation method, which is based on tank tests\(^{66}\). The STEAM model employs a range of emissions factors, specific to fuel types, engine types, and engine load, and Table 4 presents a range of these values with notes below the table.

Emission factors for NO\(_2\) depend on engine crankshaft speed (rpm) and age. The IMO Tiers are applied for engines, where Tier 0 follows the definitions of the Third IMO GHG Study\(^{12}\), and where Tiers 1–3 follow the functions defined in MARPOL VI.

Emission factors for sulphur oxides (SO\(_x\)) are determined from fuel sulphur content (% by weight) and the amount of fuel consumed at specific engine load. Part of fuel sulphur is emitted as primary PM and the sulphur fraction included in SO\(_x\) is subtracted from sulphur available for SO\(_2\) formation in the atmosphere. All gaseous emissions of sulphur are calculated as SO\(_2\). Sulphur content for residual fuel use outside of ECA regions in the BAU scenario were assigned 2.7% S, similar to the Third IMO GHG Study; in the 2020 Action scenario, sulphur content was adjusted to 0.5% S. Where lower fuel limits will be in place in 2020 due to current legislation, the maximum sulphur content was set accordingly: (i) the IMO ECA regions (0.1% S)\(^{7}\); (ii) the regions covered by the European Directive (0.5% S)\(^{36}\); (iii) China legislation applied to the Pearl River Delta, Yangtze River Delta, and Bohai Sea (0.5% S)\(^{23}\).

Particulate matter is modelled as dry PM without the associated water, which normally accompanies the sulphate aerosol. The mass of associated water depends on the ambient conditions (temperature, humidity) and the consecutive chemical transport modelling step takes the hygroscopicity of PM into account.

The emission factor for CO\(_2\) depends on fuel type and specific fuel oil consumption (SFOC) at a specific load point. The base SFOC (at 80% engine load) depends on engine age, power output and stroke type, as defined in the Second IMO GHG Study 2009\(^{67}\). Table 4 lists the range of values for HFO and MDO/MGO at high and low engine loads. These examples represent the extremes used in the model for diesel engines. This approach necessitates SFOC modelling as a function of engine load and further details can be found in Jalkanen et al.\(^{68}\) and the Third IMO GHG Study\(^{12}\).

Base year inventory adjustments for 2020. Ship inventories for future years were adjusted for the year 2020 using vessel type compound annual growth rates (Table 5), which are consistent with energy use base case growth rates used in the 2016 IMO FAS (seeMEPC 70/5/6\(^{62}\), Table 166). These growth rates produce future year inventories for ship energy demand and emissions that are lower than some other demand estimates submitted to MEPC-79 by observer delegations (MEPC 70/5/5\(^{85}\); if higher energy demand estimates were used, the health impact from uncontrolled sulphur levels would be greater. The projected emission results of the global fleet in 2020 are similar to those of the Third IMO GHG Study. Largest differences are because of reduction of sulphur in marine fuels, which has an impact on SO\(_2\) and PM emissions.

Atmospheric simulations and radiative forcing. The atmospheric simulations have been performed by the chemical transport model SILAM (System for Integrated modelLing of Atmospheric coMposition, http://silam.fmi.fi)\(^{68-72}\) with spatial resolution of 0.1 degree longitude-latitude, 13 uneven stacked layers, terrain-
| Ship type       | Growth rate |
|----------------|-------------|
| Dry bulk       | 1.74%       |
| Liquid bulk    | −190%       |
| Unitised       | 2.79%       |
| Passenger      | −0.55%      |
| Miscellaneous  | 0.00%       |
| Total energy   | 0.95%       |

Table 5 Energy-based growth rates derived from Table 166 of 2016 IMO FAS

following near surface and reaching ~330 hPa at the highest model altitude (see Supplementary Note 5). The output time step was 3 h for all produced quantities: concentrations, dry and wet deposition; and optical thickness at 380 nm, 550 nm, and 1020 nm wavelength.

The non-shipping gridded emissions inventories prepared as input into SILAM include a number of different sources. Anthropogenic source inventories are obtained through HTAP2+MEIC (Multi-resolution Emission Inventory for China)+RESAS (Regional Emission Inventory in Asia)

flame work74 through the saltation mechanism that Martincorena & Bergametti (1995) advanced in several follow-up works4−7,72. DMS from sea surface is embedded in SILAM after Kettle et al. (2000)1; biogenic VOC inventories are from MEGAN–MACC,76 and aircraft emissions are from RETRO (http://retro-archive.iek.fz-juelich.de/data/documents/reports/D1-6_final.pdf).

Simulations were driven with the meteorological fields of the Integrated Forecasting System (IFS) of the ECMWF with spatial resolution of 0.1 × 0.1 degree and temporal resolution of 3 h. Chemical transformations were computed following the set of gas-phase reactions of CBM-IV carbon-bond mechanism2 with temporal resolution of 3 h. Chemical transformations were computed using Zheng (2015). Our gridded population data are from NASA's SEDAC Population of the World, Version 4.

Health impact analysis and uncertainty. The process of calculating the health impact analysis follows the general approach discussed in previous work.10 Prior work applied a mortality risk analysis discussed in Ostro (2004)74, which built on work developed out of the United States Harvard Six Cities study conducted earlier by Pope, et al.32,84.85. The vast majority of PM$_2.5$ exposure concentrations in our study area study area represent conditions similar to those in the Harvard Six Cities study, indicating that premature mortality functions derived from that study can also apply to our study (see Supplementary Figure 9 and Supplementary Figure 10). We conduct a similar assessment (updated with new population data, incidence data, and concentration data) using a recently preferred concentration–response function (C–R) function from Lepeule, et al.31, which updates epidemiology from the Harvard Six Cities study31. Recognising that Marshall et al.86 specifically notes that the proper functional form is as yet unclear, we present the log-linear C–R function following Supplementary Note 3 (see global results in Supplementary Table 3, and regional results in Supplementary Table 4 and Supplementary Table 5). The linear C–R function reflects new understanding about the relationship between health impacts and exposure to increased air pollution concentrations and better describes the global range of exposures to global shipping pollution across a long-tailed distribution of PM$_2.5$ concentrations (see Supplementary Figure 9). Therefore, the linear C–R function (shown below) provides better estimates of health impacts where i) ambient concentrations of PM$_2.5$ are high, (e.g., >20 μg m$^{-3}$), WHO guidelines, and ii) major ship traffic contributes more to ambient concentrations (e.g., >10 μg m$^{-3}$). Supplementary Note 3 provides an alternate, log-linear C–R function and β coefficients, following Barnett et al. Supplementary Figure 4 illustrates that the log-linear function produces higher relative risk under low initial concentration (C$_{0}$) conditions than appears to higher ambient air pollution concentrations that is not evident in air health impact studies. The log-linear C–R construct enables matched comparison with most of the global health assessments, which do not adopt the linear C–R and β recommended by Lepeule’s updates. We focus on cardiovascular and lung cancer mortality responses as identified by Lepeule, et al.31. We also extend the health analysis to include for the first time an assessment of childhood asthma morbidity, which uses similar concentration–response equations based on reported asthma incidence rates by nation74.

Concentration–response functions for chronic exposure adult mortality correspond to impacts for a population cohort aged 30 years or more using Lepeule, et al.31. Concentration–response functions for childhood asthma morbidity correspond to impacts for a population cohort aged 14 years or younger using Zheng (2015). Our gridded population data are from NASA’s Socioeconomic Data and Applications Center (SEDAC) Population of the World, Version 4.

We calculated country-specific incidence rates for cardiovascular disease and lung cancer using data from the World Health Organization’s Global Health Observatory (GHO), which publishes mortality by age cohort and country, and GLOBALCAN, respectively30,94. In particular, we divided reported deaths from WHO by WHO population estimates for age cohorts 30–99 to calculate an age-standardised death rate (ASR) per 100,000. These data were converted to crude (non-ASR) values using a non-ASR:ASR ratio. For lung cancer mortality incidence rates, we use data from GLOBOCAN 201279. To determine overall health impacts associated with ship emissions and the potential effects of IMO policy delays, we calculate expected exposure from the BAU and 2020 Action scenarios and compare concentration results of these scenarios with a no-shipping case, whereby emissions from ships are removed from the geospatial inventory prior to SILAM processing.

We obtained country-specific incidence rates for childhood asthma using data from the 2014 Global Asthma Report31. Specifically, we used the current wheeze data from the ISAAC world map data as reported in the 2014 Global Asthma Report. Zheng et al.32 provide relative risk (RR) factors for childhood asthma from exposure to PM2.5 pollution (Table 2 of Zheng), which we convert to β coefficients.
We calculated expected mortality and morbidity due to changes in total particulate matter concentrations using approaches mentioned above, consistent with other recent work in this area. The total effect (E) of changes for each grid cell is given as:

\[ E = AF \cdot B \cdot P, \]

where B represents the incidence rate of the given health effect; P is the relevant exposed population; and AF is the attributable fraction of mortality or morbidity to the shipping-related PM pollution, and is given by:

\[ AF = \frac{RR - 1}{RR}. \]

For a linear C-R model, the response RR is given by the function:

\[ RR = e^\left( \beta \cdot \left( C - C_0 \right) \right), \]

And therefore,

\[ AF = 1 - e^\left( \beta \cdot \left( C - C_0 \right) \right), \]

which leads to

\[ E = \left( 1 - e^\left( \beta \cdot \left( C - C_0 \right) \right) \right) \cdot B \cdot P. \]

where \( \beta = 0.032311 \) (95% CI = 0.031303, 0.033647) for cardiovascular mortality; \( \beta = 0.031481 \) (95% CI = 0.031666, 0.035962) for lung cancer related mortality, and \( \beta = 0.002469 \) (95% CI = 0.001291, 0.003633) for childhood asthma morbidity.

We apply mortality C–R functions in a way that is consistent with previous work published by Winebrake, et al. There are several important differences in this health analysis compared with previous work; however, similar types of uncertainties affect our health results (some reducing and some increasing uncertainty ranges).

One important uncertainty involves global ship emissions and inventories. Inventory uncertainties using the AIS data and bottom-up models are both reduced and more transparent than previous work that used AMVER and ICOADS data in top-down geospatial inventories, similar to rigorous bottom-up inventory uncertainty analyses reported by the IMO, which also employed an approach similar to the STEAM model. STEAM also applies a full dynamic spatial-temporal variation of global ship emissions that reduces uncertainty further. Conducting our analysis at a 10 × 10 km resolution reduces spatial uncertainty that existed in previous work. This analysis uses a geospatial resolution considerably more resolved than other comparable work in the literature. To understand how the geospatial resolution affects results, we compare our 10 × 10 km gridded results with results from an analysis done at a 50 × 50 km resolution (Supplementary Figure 7). This comparison demonstrates the heterogeneous regional effects that can result due to geospatial resolution, with notable under-estimation of emissions over large parts of Southeast Asia and Western Europe when using more coarse model outputs.

Another uncertainty is in the specification of a linear C–R function results in higher health impact estimates where shipping pollution concentrations are higher. Based on Lepeule et al., we use a linear C–R function. Comparison of linear and log-linear C–R functions using different concentration differences demonstrate that the linear model in-grid-cell estimates can be more than an order of magnitude greater than health impacts estimated for the same concentration differences using the log-linear model. Direct comparison of Lepeule and Ostro lung cancer beta-coefficients indicate Lepeule coefficients produce higher estimated mortality impacts. Ostro (2004) estimates the relative risk for lung cancer due to a 1 μg m⁻³ concentration increase (using the so called linear formulation) to be 1.012751, corresponding to an attributable fraction (AF) of 0.01239, and beta coefficient of 0.01207. Lepeule (2012) estimates lung cancer relative risk to be 1.033182, corresponding to an AF of 0.030991 and beta coefficient of 0.031481 Ostro’s 2004 best or central estimate for lung cancer beta-coefficients falls within Lepeule’s 95th percentile confidence range. In addition to lung cancer mortality, this work evaluates cardiovascular mortality, where our prior mortality work evaluated cardiopulmonary mortality. Cardiovascular disease (CVD) is a broad term for a range of diseases affecting the heart and blood vessels that include cardiopulmonary conditions (cardiovascular conditions involving the lungs). Therefore, we observe as expected that Lepeule’s beta coefficients for CVD are not directly comparable with Ostro’s beta coefficients for cardiopulmonary disease, but produce higher estimated mortality impacts.

Lepeule uses additional years’ data to evaluate a broader uncertainty range than Ostro et al., providing a quantitative and consistent update to earlier assessments of regional and global health burden from shipping. Note that Lepeule et al. not only has a higher value for the coefficient mid-point, but also a wider 95% confidence interval for the beta coefficient compared to previous work.

Employment of best-available incidence rates for each country in a common set of C–R functions applied to the global population assumes pollution response similarity and recognises uncertainty where in-country incidence rates are poorly known or under reported. Where unavailable, global averages incidence rates are used. This source of uncertainty appears to affect asthma morbidity estimates more so than mortality estimates. We observe, for example, that nations with more advanced healthcare systems (i.e., typically wealthier nations) report higher underlying incidence rates for childhood asthma. For mortality, the uncertainty in underlying incidence rates is somewhat mitigated based on results of epidemiological studies that show similar concentration–response patterns across different countries.

A final area of uncertainty includes assumptions about the geographic distribution of populations geospatially. We distribute uniformly across each country the fraction of population under the age of 14 and between the ages of 30 and 99. This may not account for in-country spatial differences in age cohort distributions (e.g., age distributions in coastal communities that may differ from national statistics). Also, this use Knowing that “the burden of asthma, measured by disability and premature death, is greatest in children approaching adolescence (ages 10–14) and the elderly (ages 75–79)” assigning age cohorts uniformly may produce conservatively low estimates low for childhood asthma morbidity.

**Code availability.** The STEAM model is intellectual property of the Finnish Meteorological Institute (FMI) and is not publicly available. This code uses a real-time connection to IHS Fairplay ship register, requiring an annual subscription. Bilateral contracts between data providers and FMI govern access to input data sets that are required for the modelling, such as vessel activity and ship technical data. The work described in this manuscript did not contain modifications of the STEAM model; model description can be found in the references. The SILAM model is an open-code system and is free for research applications. The model can be obtained by contacting persons responsible for the SILAM model development (visit silam.fmi.fi for more information). The health risk functions are fully described in the manuscript and do not require code to replicate; data obtained for this work are also publicly available and cited.

**Data availability.** FMI may share STEAM outputs upon request as long as the data are in aggregated form, individual vessels cannot be identified, and the commercial input data cannot be reverse engineered from model outputs. The gridded ship emission files used in this work are available from FMI ftp site upon request to authors. The gridded population data used in this work were taken from NASA’s Socioeconomic Data and Applications Center (SEDAC) Population of the World, Version 4. This data set is available at http://sedac.ciesin.columbia.edu/data/collection/gpw-v4. The Population by Age Group data set can be downloaded at https://esa.un.org/unpd/wpp/Download/Standard/Population/.

The cardiovascular disease mortality data using an age-standardised death rate published by WHO were downloaded from http://www.who.int/healthinfo/global_burden_disease/estimates/en/index1.html. Asthma data reporting incidence rates are provided by The Global Asthma Report 2014 at http://globalasthmareport.org/.

Received: 24 July 2017 Accepted: 22 December 2017 Published online: 06 February 2018

**References**

1. Anderson, C., Bergström, R. & Johansson, C. Population exposure and mortality due to regional background PM in Europe—Long-term simulations of source region and shipping contributions. Atmos. Environ. 43, 3614–3620 (2009).

2. Broome, R. A. et al. The health benefits of reducing air pollution in Sydney, Australia. Environ. Res. 143, 19–25 (2015).

3. Caiazzo, F., Ashok, A., Wiatz, I. A., Yim, S. H. & Barrett, S. R. Air pollution and early deaths in the United States. Part I: Quantifying the impact of major sectors in 2005. Atmos. Environ. 79, 198–208 (2013).

4. Corbett, J. L. et al. Mortality from ship emissions: a global assessment. Environ. Sci. Technol. 41, 8512 (2007).

5. Environmental Protection Agency. Regulatory Impact Analysis: Control of Emissions of Air Pollution from Category 3 Marine Diesel Engines (U.S. Environmental Protection Agency, Washington, DC, 2009).

6. Jonson, J., Jalkanen, J., Johansson, L., Gauss, M. & Denier van der Gon, H. Model calculations of the effects of present and future emissions of air pollutants from shipping in the Baltic Sea and the North Sea. Atmos. Chem. Phys. 15, 783–798 (2015).

7. Liu, H. et al. Health and climate impacts of ocean-going vessels in East Asia. Nat. Clim. Change 6, 1037–1041 (2016).

8. Moldanová, J. et al. Physical and chemical characterisation of PM emissions from two ships operating in European Emission Control Areas. Atmos. Meas. Tech. 6, 3577–3596 (2013).

9. Vutukuru, S. & Dadub, D. Modeling the effects of ship emissions on coastal air quality: a case study of southern California. Atmos. Environ. 42, 3751–3764 (2008).

10. Reference citations of the World, Version 4. This data set is available athttp://sedac.ciesin.columbia.edu/data/collection/gpw-v4. The Population by Age Group data set can be downloaded at https://esa.un.org/unpd/wpp/Download/Standard/Population/.
67. Buhaug, Ø. et al. Second IMO Greenhouse Gas Study 2009. (International Maritime Organization, London, 2009).
68. BIMCO and IPIECA, Baltic International Maritime Council & International Petroleum Industry Environmental Conservation Association. Review of Fuel Oil Availability as Required by Regulation 14.8 of MARPOL Annex VI. (International Maritime Organization, London, 2016).
69. Kouznetsov, R. & Sofiev, M. A methodology for evaluation of vertical dispersion and dry deposition. J. Geophys. Res. Atmos. https://doi.org/10.1002/2011JD016366 (2012).
70. Marécal, V. et al. A regional air quality forecasting system over Europe: the MACC-II daily ensemble production. 8, 2777–2813 (2015).
71. Sofiev, M. Extended resistance analogy for construction of the vertical diffusion scheme for dispersion models. J. Geophys. Res. Atmos. 107, 1–8 (2002).
72. Sofiev, M. et al. An operational system for the assimilation of the satellite information on wild-land fires for the needs of air quality modelling and forecasting. Atmos. Chem. Phys. 9, 6833–6847 (2009).
73. Janssens-Maenhout, G. et al. HTAP_v2. 2: a mosaic of regional and global emission grid maps for 2008 and 2010 to study hemispheric transport of air pollution. Atmos. Chem. Phys. 15, 11411–11432 (2015).
74. Fécan, F., Marticorena, B. & Bergametti, G. Parametrization of the increase of wind speed at emission? Atmos. Chem. Phys. 11, 10149–10156 (2011).
75. Marticorena, B. & Bergametti, G. Modeling the atmospheric dust cycle: 1. Design of a soil-derived dust emission scheme. J. Geophys. Res. Atmospheres 100, 16415–16430 (1995).
76. Zender, C. S., Bian, H. & Newman, D. Mineral Dust Entrainment and Deposition (DEAD) model: Description and 1990s dust climatology. J. Geophys. Res. Atmos. https://doi.org/10.1029/2002JD002775 (2003).
77. Sindelarova, K. et al. Global data set of biogenic VOC emissions calculated by the MEGAN model over the last 30 years. Atmos. Chem. Phys. 14, 9317–9341 (2014).
78. Shrivastava, M. et al. Modeling organic aerosols in a megacity: Comparison of simple and complex representations of the volatility basis set approach. Atmos. Chem. Phys. 11, 6639–6662 (2011).
79. Tsimpidi, A. P. et al. Evaluation of the volatility basis-set approach for the simulation of organic aerosol formation in the Mexico City metropolitan area. Atmos. Chem. Phys. 10, 525–546 (2010).
80. Glavas, A. & Käyhä, A. Technical note: The libRadtran software package for radiative transfer calculations-description and examples of use. Atmos. Chem. Phys. 5, 1855–1877 (2005).
81. Soares, J., Sofiev, M. & Jalkanen, J. P. in NATO Science for Peace and Security Series C: Environmental Security Vol. 137 (ed. Prüss-Üstün, A., Campbell-Brown, M. & Martines, C.) 6639 (2013).
82. Dockery, D. W. et al. An association between air pollution and mortality in Six U.S. cities. New Engl. J. Med. 329, 1753–1759 (1993).
83. Laden, F., Schwartz, J., Speizer, F. E. & Dockery, D. W. Reduction in particulate air pollution and mortality: extended follow-up of the Harvard Six Cities study. Am. J. Respir. Crit. Care Med. 173, 667–672 (2006).
84. Marshall, J. D., Apt, J., Coggins, J. S. & Goodkind, A. L. Blue skies blue? Environ. Sci. Technol. 49, 13929–13936 (2015).
85. Center for International Earth Science Information Network (CIESIN) Columbia University. Gridded Population of the World (GPW), v4. NASA Socioeconomic Data and Applications Center (SEDAC). http://sedac.ciesin.columbia.edu/data/set/gpw-v4-population-count (2016).
86. United Nations. World Population Prospects: The 2015 Revision, Key Findings and Advance Tables. (United Nations, Department of Economic and Social Affairs, Population Division, New York, 2015).
87. BIMCO and IPIECA, Baltic International Maritime Council & International Petroleum Industry Environmental Conservation Association. Review of Fuel Oil Availability as Required by Regulation 14.8 of MARPOL Annex VI. (International Maritime Organization, London, 2016).
88. World Health Organization. Global Health Observatory Data Repository: Adult Mortality Data by Country. World Health Organization http://apps.who.int/gho/data/node.main.11 (Geneva, Switzerland, 2016).
89. Davis, D. L. Short-term improvements in public health from global-climate policies on fossil-fuel combustion: an interim report. Lancet 350, 1341–1349 (1997).

Acknowledgements
We are grateful for the financial support from the ClimateWorks Foundation, support of the APTA project of Academy of Finland, and in-kind support from the Finnish Meteorological Institute and Energy and Environmental Research Associates, LLC. We also acknowledge Samuel G. Winebrake for help in data collection and processing, and Doug Lowenthal of the Desert Research Institute for the observational data for the model of indirect aerosol effects.

Author contributions
M.S., I.J., M.P., I.S., F.C., R.W. and P.J. contributed to the emissions inventory and radiative forcing modelling. J.E.W., E.W.C., and I.J.C. contributed to the health analysis; all authors contributed to the overall preparation and editing of the manuscript; I.J.C. led the initial analysis, coordinated the collaboration submittal efforts, and is corresponding author.

Additional information
Supplementary Information accompanies this paper at https://doi.org/10.1038/s41467-017-02774-9.

Competing interests: The authors declare no competing financial interests.

Reprints and permission information is available online at http://npg.nature.com/reprintsandpermissions/

Publisher's note: Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.