Valuating environmental impacts from ship emissions – The marine perspective

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Research article

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ABSTRACT

Shipping is an activity responsible for a range of different pressures affecting the marine environment, air quality and human welfare. The methodology on how ship emissions impact air quality and human health are comparatively well established and used in cost-benefit analysis of policy proposals. However, the knowledge base is not the same for impacts on the marine environment and a coherent environmental and socio-economic impact assessment of shipping has not yet been made. This risk policies to be biased towards air pollution whilst trading off impacts on the marine environment. The aim of the current study was to develop a comprehensive framework on how different pressures from shipping degrade marine ecosystems, air quality and human welfare. A secondary aim was to quantify the societal damage costs of shipping due to the degradation of human welfare in a Baltic Sea case study. By adding knowledge from marine ecotoxicology and life-cycle analysis to the existing knowledge from climate, air pollution and environmental economics we were able to establish a more comprehensive conceptual framework that allows for valuation of environmental impacts from shipping, but it still omits economic values for biological pollution, littering and underwater noise. The results for the Baltic Sea case showed the total annual damage costs of Baltic Sea shipping to be 2.9 billion €$_{2010}$ (95% CI 2.0–3.9 billion €$_{2010}$). The damage costs due to impacts on marine eutrophication (768 million €$_{2010}$) and marine ecotoxicity (582 million €$_{2010}$) were in the same range as the total damage costs associated with reduced air quality (816 million €$_{2010}$) and climate change (737 million €$_{2010}$). The framework and the results from the current study can be used in future socio-economic assessments of ship emissions to prioritize cost efficient measures. The framework can be used globally but the damage costs presented on the marine environment are restricted to emissions on the Baltic Sea and Kattegat region as they are based on willingness to pay studies conducted on citizens around the Baltic Sea where eutrophication and emissions of chemicals are particularly threats to the state of the Baltic Sea.

1. Introduction

The utilization of the marine environment is today wide, ranging from oil and natural gas extraction, to fishing and aquaculture to renewable energy installations and finally shipping and leisure boating. To ensure sustainable use of marine resources, there is a need to understand what unintended impacts these activities have on ecosystems and human health. One of the more interesting sectors is shipping which has shown to affect the marine environment in many different ways via discharges of contaminants from grey water (Ytreberg et al., 2020), sewage (ADEC Alaska Department of Environmental Conservation, 2018), bilge water (Tiselius and Magnusson, 2017), scrubber water

(Koski et al., 2017) and antifouling paints (Thomas and Brooks, 2010); emission of nutrients from sewage, grey water, food waste and deposition of nitrogen oxides (NO_X) (Raudsepp et al., 2019); emissions of acidifying compounds from scrubber washwater and deposition of sulfur oxides (SO_X) (Endres et al., 2018); spread of invasive species from hulls or ballast water (Havel et al., 2015); and finally, underwater noise (Weilgart, 2007). Shipping also affect terrestrial ecosystems through eutrophication and acidification as well as human health by emission of air pollutants such as fine particulate matter (PM2.5), non-methane volatile organic compounds (NMVOCs), nitrogen oxides (NO_X) and sulfur oxides (SO_X) (Jalkanen et al., 2012, 2014). Climate change is also affected via emissions to air of the greenhouse gases carbon dioxide

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(CO_2) the short-lived climate pollutants (SLCP) methane (CH_4) and black carbon (BC), NO_X and in many sea regions sulfur dioxide (SO_2) (Eyring et al., 2010; Fuglestvedt et al., 2014).

The knowledge about terrestrial ecosystem and human health impacts of air pollution is comparatively well developed, much because land-based emissions of air pollutants have been regulated in international and national regulations in a tight science-policy regime. Since the 80′s, analytical progress has been substantial in areas such as air pollution inventories and monitoring, emission dispersion modelling, integrated assessment modelling and cost-benefit analysis of international air pollution control (Maas and Grennfelt, 2016; Reis et al., 2012). Also, the climate change sciences have experienced similar development, as summarized in the IPCC assessment reports.

The methodology and methods of environmental economics is used to quantify environmental, human health and climate change impacts in monetary terms, which enables a single-unit comparison of costs and benefits of emission reductions, i.e. cost-benefit analysis. The concept of market externalities is central, which can basically be considered as effects of a traded good or service not already accounted for in the market price; environmental impacts being one of the more well recognized since Ayres and Kneese (1969). Given the absence of a real market for such negative externalities (henceforth referred to as damage costs), some sort of valuation is needed. In these valuations, environmental economists try to establish a price, or willingness to pay (WTP), for the good or service in focus for the valuation (Costanza et al., 1997; Nieminen et al., 2019). There are two main branches of the valuation methods. Either one tries to reveal the willingness to pay from price variations in existing markets for other goods such as houses in which the good in focus, such as air quality, varies in supply. Alternatively, one tries to establish a credible hypothetical market situation and let respondents engage in hypothetical market exchanges of the good or service in focus. Both branches have several sub-categories which ultimately in the context of this study enables a valuation of the damage costs associated with specific types of environmental and human health degradation (Boardman et al., 2001).

Currently, the effect of large-scale air pollution emission changes on ecosystem-, human health-, and economic impacts can be modelled with reasonable accuracy, and analysis of these impacts are done with established methods and models. These methods and models feeds in to the air pollution & shipping policy processes, such as the revised EU Sulfur-in-fuels/Fuel Quality Directive (Directive No, 1999/32/EC & 2009/30/EC) and the International Maritime Organization (IMO) use of sulfur and nitrogen emission control areas (SECA and NECA respectively) and the global sulfur cap which from January 1, 2020 requires the maximum sulfur content of marine fuels to be reduced from 3.5% to 0.5% (IMO, 2017) (Amann et al., 2013; Åström et al., 2018; Bosch et al., 2009; Cofala et al., 2018). The same can be said for climate change, where the regular IPCC reports provides influential input to several policy processes, such as the 1.5° special report (IPCC, 2018).

However, the knowledge base is not the same for impacts on the marine environment and a coherent environmental and socio-economic impact assessment of shipping has not yet been made. This risk policies to be biased towards air pollution and climate change whilst trading off impacts on the marine environment. Further, European legislation, via the Marine Strategy Framework Directive (MSFD; Directive, 2008/56/EC), requires member states to assess the benefits of improving the conditions of the sea to a good environmental status. And finally, the pressure on marine resources and the demand for marine ecosystem services are too high in many marine water bodies (Culhane et al., 2019). For these reasons it is important to establish methods that comprehensively cover the environmental and human health damages that shipping (and other human marine activities) have on the socio-economic system including impact on the marine environment. To adhere to these needs, whilst recognizing the different levels of pre-existing knowledge on damages, the aim of this study is to develop a framework allowing to determine how different pressures from shipping

affect the contribution to the socio-economy of primarily marine ecosystem services, but also freshwater and terrestrial ecosystems, human health and climate change. A secondary aim is to quantify the societal costs of shipping due to the degradation of ecosystem services and human welfare in a Baltic Sea case study.

2. Concretizing a framework to value damage costs for the shipping sector

Given the multipe environmental and human health damages, as well as the multipe drivers of the damages associated with shipping, it is important for any coherent framework to maintain a systems approach. The DPSIR (*Drivers, Pressures, State, Impact* and *Response*) framework is a structured theoretical framework aiming to analyze environmental problems and to identify and propose adequate measures to reduce the problem as such (Atkins et al., 2011; Borja et al., 2006; Relvas and Miranda, 2018). DPSIR starts with identifying the driving force (**Drivers**) that causes specific environmental pressures. The **Pressure** on the environment can in turn change the **State** of the environment. This change in State may cause an **Impact** on ecosystems and human health as well as the way human can use the ecosystem (i.e. ecosystem services). Society can then act in different ways to reduce the Pressure by the specific Driver. The latter is termed **Response**. In a recent study by Elliott et al. (2017) the DPSIR framework was proposed to be extended to DAPSI(W) R(M) in which **Drivers** of basic human needs require different **Activities** which leads to environmental **Pressures**. The pressures will lead to a change in environmental **State** which subsequently lead to **Impacts** (on human **Welfare**). This will then require **Responses** (of **Measures**) to reduce different environmental pressures.

In the current study, we have developed a conceptual framework on how the pressure of different emission sources from shipping can be structured to assess the environmental impact and how this impact can be translated to losses in human welfare (Fig. 1). The framework is built on the DAPSI(W)R(M) concept but includes Life Cycle Impact Assessment (LCIA) to be able to compare the impact of different emissions sources and to assess how shipping in monetary terms impacts human welfare. LCIA is used as it allows for weighting of different pollutants in terms of their contribution to a specific environmental theme via characterization factors (de Bruyn et al., 2018). The higher the characterization factor, the greater potential impacts. Characterization are particularly useful when comparing the impact of many different emission sources or pollutants. With the use of characterization factors, the complexity of assessing e.g. the ecotoxicological impacts of a large number of contaminants on the marine environment can be reduced to a single indicator representing the toxicity potential (TP) (Huijbregts et al., 2016). Damages to human health, terrestrial eutrophication and acidification and climate change induced by emissions to air is already systematically addressed with established methods (Amann et al., 2013; Amann et al., 2011; Holland, 2014; IPCC, 2014; Nordhaus, 2017). Correspondingly, this study focuses primarily on the marine environment and associated emissions and discharges.

3. Valuating damage costs for the shipping sector in the Baltic Sea

The conceptual framework in Fig. 1 was designed to comprehensively address all pressures from shipping and its corresponding impacts on human welfare. We used activity data for Baltic Sea shipping for the year 2018 and calculated the corresponding pressures (loads) on the marine environment and to the atmosphere. However, due to lack of data, the pressures from biofouling, stern tube oil, garbage, anchoring and mooring and underwater noise are excluded in the present study (shown as dotted lined in Fig. 1). Hence, the estimated damage cost of emissions from commercial shipping in the Baltic Sea are based on the pressures from antifouling paints, ballast water, bilge water, sewage, grey water, food waste, scrubber water and engine exhaust. The damage
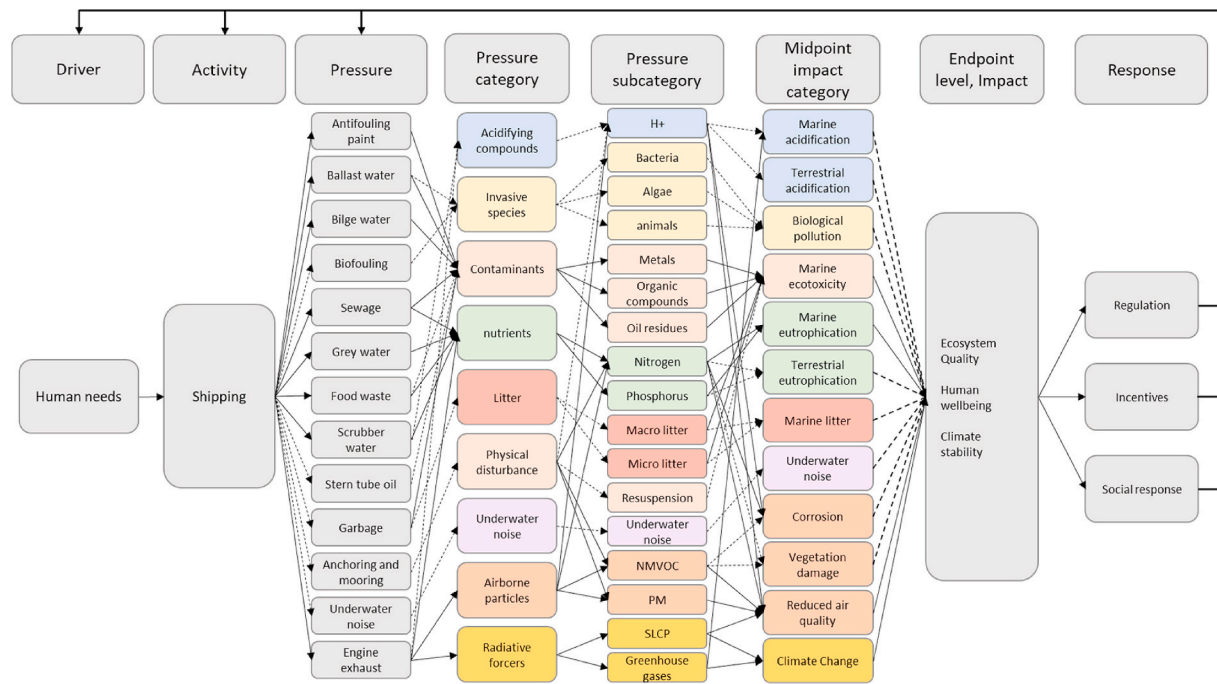
Fig. 1. Conceptual framework on how shipping affects marine and terrestrial ecosystem quality, impacts on human welfare, and climate change. Pressure sources include antifouling paints, ballast water, bilge water, biofouling, sewage, grey water, food waste, scrubber water, stern tube oil, garbage, anchoring and mooring, underwater noise and engine exhaust. The dotted lines represent flows not monetised in the following Baltic sea case study.

costs were calculated for the midpoint impact categories marine eco-toxicity, marine eutrophication, reduced air quality as well as climate change. The damage costs were calculated taken into account un-certainties from all individual aspects, i.e. the annual pressure (loads or volumes) from the different pressure categories, concentration of con-taminants, nutrients, airborne particles and radiative forcers as well as the uncertainties in the impact valuation. The emissions of NO_X, SO_2, PM2.5, CO, NMVOC and CO_2 to the atmosphere were used to calculate damage costs due to climate change and reduced air quality. For NO_X, also N deposition on the Baltic Sea were used to determine the external cost resulting from marine eutrophication. Deposition of the other airborne pollutants (SO_2, PM2.5, CO, NMVOC and CO_2) on the seafloor were not included since no external cost (resulting from damage on the marine environment) exist for these pollutants. To fully reflect the damage from shipping on the marine environment, also the midpoint impact categories biological pollution, marine acidification, marine litter and underwater noise should be included. However, this was not possible due to the lack of damage cost estimations of these impact categories for the Baltic Sea. For the same reason, effects of air pollution on freshwater and terrestrial acidification, freshwater and terrestrial eutrophication and corrosion was excluded. We omitted effects from air pollution on crop productivity (vegetation damage) since previous analysis have shown that these costs are not even 1% of the costs from health effects of air pollution (Holland, 2014).

3.1. Drivers

The Driver is in the current framework referred to as "basic human needs", and comprise food, shelter, security, goods and services (Patrício et al., 2016). In order to obtain this, society carries out activities, including e.g. fishing, shipping, agriculture, leisure activities, all causing pressures on the environment. In the current framework, only the link between drivers and the activity shipping is assessed.

3.2. Activity

Shipping is a complex industry comprising different ship categories ranging from passenger ships to oil tankers and container ships. For the year 2018, 7914 unique IMO registered ships visited the Baltic Sea (Jalkanen and Johansson, 2019a). The vast majority of the ships were Cargo ships (4011) followed by Tankers (1911). The volume (and pressure) of different emission sources from shipping varies depending on ship category. Grey water and sewage produced on ships are, for example, correlated to the number of passenger whereas release of biocides from antifouling paints is a function of the painted wetted hull surface area. Hence, the activity of shipping is in this framework split into certain ship classes (RoPax vessels, Vehicle carriers, Cargo ships, Container ships, Tankers, Passenger ships, Cruisers, Fishing vessels and Service ships).

3.3. Pressures (volumes and loads) to the marine environment and to the atmosphere

As shown in Fig. 1, only the pressure categories antifouling paints, ballast water, bilge water, sewage, grey water, food waste, scrubber water and engine exhaust are included in the present Baltic Sea case study. The yearly pressures (volumes of discharge water) from ballast water, bilge water, sewage and grey water and biocides from antifouling paints on the Baltic Sea were obtained from Jalkanen and Johansson (2019b). The data is derived from the EU BONUS project *SHEBA* where a new water emission application to the STEAM (Ship Traffic Exhaust Emission) model originally developed by Jalkanen et al. (2014) were used. Briefly, emission factors (Liters per person per day (sewage and grey water), m^3/MWh (scrubber water), m^3 per loading/unloading event (ballast water), and $\mu g/cm^2/d$ (leaching rate of biocides from antifouling paints) where coupled with Baltic shipping AIS data for 2018 via the STEAM model. The STEAM model uses AIS data positions from all IMO-specified vessels and contain detailed information about e.g. the vessels passenger capacity, main engine power, gross tonnage, vessel size, hull surface area etc., and can hence produce activity and pressure

maps of individual vessel and for all Baltic Sea shipping. We used the latest STEAM model run, available in Jalkanen and Johansson (2019b), which contains total discharged volumes of sewage, grey water, scrubber water (open- and closed loop), treated ballast water (assuming all ballast water to be treated with an IMO approved ballast water management system) and treated bilge water from all IMO-registered vessels trafficking the Baltic Sea in 2018. The loads were determined considering different operations; cruise, maneuvering and hoteling in harbors. However, the STEAM pressure data (volumes and loads) do not contain any estimates on data uncertainties. Therefore, to calculate uncertainties, we used the same background material as was used in the STEAM model (DNV, 2009), where production rates (L/passenger/day) of grey water and sewage have been reported for different vessels within specific ship classes. This DNV data was used in the present study to calculate uncertainties in discharges rates of grey water and sewage as well as to determine uncertainties of the total yearly volumes emitted to the Baltic Sea (see Supporting Material A and Table S2, Table S3, Table S7 and Table S8).

For open loop and closed loop scrubbers, the discharge rate in the STEAM model are assumed to equal $45 m^3/MWh$ and $0.3 m^3/MWh$, respectively, as recommended by IMO (2008). These assumptions may however be out of date, and based on 41 recently on-board measured discharge rates (MEPC 73/INF.5, 2018) we updated the open loop and closed loop discharge rate and applied them to calculate the total annual discharge volume and uncertainties (see detailed description in Supporting Material A). We were however not able to calculate uncertainties for ballast water or bilge water discharge rates.

The annual load of biocides from antifouling coatings was also obtained from Jalkanen and Johansson (2019b) where four different antifouling paint categories are used depending on where the ship operates (see detailed description in Supporting Material A). The pressure of biocides (copper, zinc and zineb), in $\mu g/cm2/d$, from the different antifouling paint categories are also shown in Supporting Material A Table S5.

To calculate damage costs of nutrient supply to the Baltic Sea, a higher resolution at Baltic Sea sub-basin scale was needed for greywater, sewage and N deposition. The yearly load of nitrogen and phosphorus from greywater and sewage per Baltic Sea sub-basin was obtained from Ytreberg et al. (2020). The study also uses the STEAM model, but emissions are estimated for the year 2012. However, since the total yearly discharges of grey water in 2012 (5.5 million m^3) is similar as determined for 2018 (5.4 million m^3), the 2012 sub-basin emission data was decided to be appropriate to also represent emissions for 2018.

The corresponding annual total load of contaminants and nutrients to the Baltic Sea during 2018 was determined by multiplying the total discharged volumes with the average concentration of contaminants and nutrients present in the respective emission source. Concentrations of contaminants in grey water was obtained from Ytreberg et al. (2020) while concentrations in ballast water, bilge water, sewage, grey water and scrubber water (open and closed loop mode) were obtained from Jalkanen et al. (2020). Both studies have conducted an extensive literature review to characterize the different waste streams. The average concentrations of contaminants and calculated uncertainties are shown in Supporting Material B and are based on samples from 40 different Ballast Water Management Systems, 49 samples of bilge water, 95 samples of sewage, 69 samples of grey water, 56 samples of open-loop scrubber water, 14 samples of closed loop scrubber water and 145 measurements of biocidal release rates from antifouling coatings.

The concentrations of nitrogen 0.029 kg N/m^3 (0.004–0.054 95% CI) and phosphorous 0.0048 kg N/m^3 (0.0018–0.0078 95% CI) in grey water were derived from Ytreberg et al. (2020). For sewage, 0.43 kg N/m^3 and 0.028 kg P/m^3 was used (Ytreberg et al., 2020).

Annual ship engine exhaust emissions of NO_X, SO_2, PM2.5, CO, NMVOC and CO_2 in the Baltic Sea were obtained from Jalkanen and Johansson (2019a) and consider both cruise, maneuvering and hoteling in harbors. Uncertainties are not given in the reference; however, the same model was used in the third IMO GHG study (International Maritime Organization, 2014) where the uncertainty is given as 13% (SD of mean) which is used here. No information was available for emissions of CH_4. For N deposition, we used the 2018 NO_X emissions from Baltic shipping reported by Jalkanen and Johansson (2019a) and assumed that 18% is deposited in the same sea-basin as the NO_X were emitted. The latter assumption is calculated based on an average of the most recent EMEP reports on emissions and deposition of nitrogen on the Baltic Sea region (EMEP, 2014, 2016, 2018, 2019).

3.4. Impact at midpoint and endpoint level

The midpoint impact categories marine ecotoxicity, marine eutrophication, reduced air quality and climate change were used in the damage cost valuation (Fig. 1). For the impact category marine ecotoxicity, characterization factors from ReCiPe were used to determine the cumulative toxicity potential of all contaminants present in the specific pressure (e.g. in ballast water). ReCiPe, which is the most recent and harmonized indicator approach available in LCIA, have produced characterization factors for over 3000 organic substances and 20 metals for different environmental compartments (freshwater, marine waters, air etc.) (Huijbregts et al., 2016). The ReCiPe characterization factors for the hierarchist marine perspective was used for the midpoint category marine ecotoxicity, expressed as1,4-dichlorobenzene equivalents (1, 4-DCB eq). To avoid double counting, individual PAHs was used and PAH sum16 was excluded in the analysis. For oil index, which is determined for bilge water, the ReCiPe characterization factor "Hydrocarbons, aliphatic, alkanes, cyclic" was used. To avoid double counting, the petroleum fractions "C10–C12", "C12–C16", "C16–C35" and "C35–C40" were excluded in the analysis. The average concentration, in $\mu g/L$ (and 95% CI), of the different contaminants identified in each pressure (ballast water, bilge water, sewage, grey water and scrubber water) was multiplied with the ReCiPe characterization factor of the specific contaminants to derive the total TP per m^3, expressed as kg 1, 4-DCB/m^3 (see Supporting Material B). The average load, in kg, (and 95% CI) of the different contaminants identified in each emission source (pressure) was multiplied with the characterization factor of the specific contaminants to derive the annual (year 2018) toxicity potential each emission source pose to the Baltic Sea. This was calculated according to the following equation:

$$Toxicity\ potential_i = \sum_j (conc_{i,j} * CF_j * V_i)$$

where:

$i,j,$: emission source, substance.

CF: ReCiPe characterization factor for the hierarchist marine perspective (1,4-DCB eq).

V: annual volume discharged to the Baltic Sea.

For antifouling paints, the total yearly (2018) load of copper, zinc and zineb was multiplied with the corresponding characterization factor to obtain the annual toxicity potential.

In a study by Noring et al. (2016) the valuation of ecotoxicological impacts from the organotin compound tributyltin (TBT) was assessed in Sweden. The study, based on peoples' willingness-to-pay, concluded that households in Sweden are willing to pay on average 108 USD per year (95% CI = 74–129 USD) to prevent any release of paint flakes containing TBT to the marine environment. The total willingness-to-pay per year was obtained by multiplying 108 USD with the number of households in Sweden. The total willingness-to-pay was divided with the total amount of TBT released to the environment during the period 1965–2001 to generate a damage price for TBT in USD/kg TBT. A generic damage price for "marine ecotoxicity", expressed in kg/1, 4-DCB-eq, was developed by Noring (2014) by dividing the damage price for TBT with the ReCiPe characterization factor for TBT. The damage cost of marine ecotoxicity was calculated to 1 €/kg 1,4-DCB-eq

([Noring, 2014](#)) in 2012 price level. This corresponds to 0.94 €$_{2010}$ per kg 1,4-DCB-eq and based on the variation in the households' willingness-to-pay, a 95% CI of 0.64–1.13 €$_{2010}$/kg 1,4-DCB-eq was used in the current study as damage price for marine ecotoxicity.

Eutrophication is a severe problem in the Baltic Sea and Maximum Allowable Inputs (MAI) for both nitrogen and phosphorus for the entire Baltic Sea as well as per Baltic Sea sub basins have been set by HELCOM and the Baltic Sea Action Plan (BSAP) to reduce the pressure of nutrient ([Svendsen et al., 2015](#)). On a Baltic Sea level, 62,621 tonne of nitrogen and 10,640 tonne of phosphorus needs to be reduced to follow the MAI according to BSAP and the reductions are required in the sea basins: Bothnian Bay (phosphorus), Baltic Proper (nitrogen and phosphorus), Gulf of Finland (nitrogen and phosphorus) and Gulf of Riga (nitrogen and phosphorus) (Supporting material A [Table S6](#)).

There are a few studies where losses in welfare due to the deterioration of the Baltic Sea environment have been assessed. In a study by [Ahtiainen et al. (2014)](#), citizens from all nine littoral countries of the Baltic Sea were in 2012 asked how much they are willing to pay annually to improve the eutrophication level as set by the HELCOM BSAP. The aggregated willingness to pay for fulfilling the BSAP scenario amounts to 4000 million Euros annually ([Ahtiainen et al., 2014](#)). In the [HELCOM (2017)](#) report the cost of degradation from eutrophication were adjusted to the 2015 purchase power and calculated to be 3.8–4.4 billion Euros. Hence, the exceedance of the MAI to the Baltic Sea (i.e. 62,621 tonne of nitrogen and 10,640 tonne of phosphorus) could be translated to losses in human welfare corresponding to 3.8 to 4.4. billion Euros annually. Hence, the damage cost of N and P to the Baltic Sea was calculated with the following equation

$$Damage\ \cos t\ Eutrophication = \frac{Exceedance\ MAI}{Tot\ WTP}$$

where: Damage cost eutrophication: damage cost in €/kg of N and P. Exceedance MAI: The total exceedance of MAI with respect to both N and P, in kg/year Tot WTP: Total willingness to pay to follow the BSAP.

The calculated damage cost of eutrophication for not fulfilling BSAP is hence 52–60 €$_{2010}$/kg N (when nitrogen is discharged to the Baltic Proper, Gulf of Finland and Gulf of Riga) and 52–60 €$_{2010}$/kg P (when phosphorus is discharged to the Bothnian Bay, Baltic Proper, Gulf of Finland and Gulf of Riga).

The GAINS and ARP-models were used to estimate human health impact and damage costs from engine exhaust air pollutants. The GAINS model ([Amann et al., 2011](#); [Cofala et al., 2018](#)), is an air pollution Integrated Assessment Model which integrates modelling of emissions, emission abatement costs, regional emission dispersion, human health effects, and ecosystem impacts to provide support to policy makers. The European version covers all larger European countries and sea regions, the time span 1990–2050, and emissions of the air pollutants SO_2, NO_X, NH_3, PM2.5, BC, OC, NMVOC and the GHG:s CO_2, CH_4, N_2O, HFC, PFC and SF_6. The online version of the model is openly available after registration.

The ARP model ([Holland et al., 2013](#)) is a tool for health impact assessment and monetary evaluation of air pollution emissions. By using age group specific population data projections from the [United Nations (2011)](#), together with data on health impact incidence rates and data on concentration-response functions from [WHO et al. (2013)](#), and economic valuation of human health endpoints ([Holland et al., 2005](#)), the ARP model calculates the health impacts from air pollution and the corresponding damage costs.

For the purpose of this study we used the reduced complexity emission dispersion module of the GAINS model (a linearized version of the EMEP emission dispersion model ([Simpson et al., 2012](#))) to calculate Baltic Sea shipping emission dispersion and corresponding effects on population-weighted concentration of PM2.5 and O_3 in the countries affected by emissions from the Baltic Sea in 2020 (the model year closest to 2018). By adjusting Baltic Sea emission levels pollutant by pollutant

for SO_2, NO_X, PM2.5, and NMVOC, we could estimate the effect on population-weighted concentration of PM2.5 and O_3 in the surrounding countries per unit pollutant emitted in the Baltic Sea. These GAINS-results were then transferred to the ARP model to allow for valuation of emissions from shipping in the Baltic Sea. The results could then be expressed as €/kg pollutant. For the Baltic sea 2018 emissions of the main air pollutants to the atmosphere, the damage costs for NO_X then becomes 1–6 €$_{2010}$ per kg, and for PM2.5 6–30, SO_2 4–19 and NMVOC 0.3–0.4 €$_{2010}$ per kg. Given that much of the variation is associated with different ethical standpoints regarding the value of a life lost near end-of-life, we split these perspectives into the following ethical components: VSL (value of statistical life) and VOLY (value of life year lost), as presented in [Table 1](#).

The economic value of radiative forcing used in this study was taken from the global average social cost of carbon in 2018 from [Nordhaus (2017)](#) for varying discount rates and climate targets: US$$_{2010}$ 34/tonne CO_{2eq} assuming a baseline projection and 3% discount rate (22 for a baseline projection and 5% discount rate and 201 for a 2.5 max degree scenario and a 2.5% discount rate), corresponding to 26 €$_{2010}$ (16–154). Given that radiative forcing comes from greenhouse gases and air pollutants and that the equivalent effect on radiative forcing depends on inter alia ethical standpoints regarding future generations, the choice of climate metric is important for the results. In this study we used the GWP_{100} metric since it is most common in the climate policy discussions. The metric values were based on the specification in [Jalkanen and Johansson (2019a)](#). We assumed that PM2.5 emissions from Baltic Sea shipping are composed of 11% black carbon, 30% organic carbon, 8% ash and 51% hydrated SO_4. There are no literature estimates on the radiative forcing of non-carbonaceous PM2.5-fractions. We therefore assumed that ash and hydrated SO_4 have the same radiative forcing properties as SO_2 in gaseous form.

3.5. Model ship

As a further illustration of the impacts and damage costs resulting from shipping, a single RoPax ship operating in the Baltic Proper was used as an illustrative model ship in the assessment. The model ship has characteristics typical for the ships operating in the Baltic Sea transporting passengers and cargo between Finland and Sweden. Ships of the same type as the RoPax model ship operate all year round in the Baltic Sea and typically make stops on the eastern and western side, respectively, of the Baltic Sea every two days. The total fuel consumption for all RoPax ships in the Baltic Sea has been estimated to be 1050 ktonne annually which is about 29% of the total annual fuel consumption for shipping in the Baltic Sea ([Jalkanen and Johansson, 2019a](#)). For the model ship we determined damage cost in different scenarios, representing various emission control strategies (in one scenario the ship does not have an open-loop scrubber, in the next it follows Tier III NO_X regulations and in the final it uses biocide-free paint). The characteristics of the RoPax model ship and the different scenarios are described in detail in Supporting Material A [Table S1](#).

3.6. Statistical analysis

The uncertainty analysis in this work was performed with the Excel add-in program @RISK. The software uses Mote Carlo simulations to determine the combined uncertainties when several parameters are included. The distributions for the parameters are assumed to be normal

Table 1
Estimated health-related damage costs (€$_{2010}$/kg emission) associated with 2018 Baltic Sea shipping emissions of NO_X, SO_2, PM2.5 and VOC.

	NO_X	SO_2	PM2.5	VOC
VOLY	2 (1–3)	5 (4–10)	7 (6–15)	0.3 (0.3–0.4)
VSL	5 (3–6)	15 (8–19)	24 (13–30)	0.3 (0.3–0.4)

with the exception of the emission factors for NO_X and PM for the RoPax ship which are assumed to be log-normal and the unit cost values, assumed to be triangular. Note that the distributions for the latter are significantly skewed with the central values differing from the mean values (see Table 1). The uncertainty values used are given throughout the text.

4. Results

The conceptual framework outlined in Fig. 1 was developed to monetise damages to human health and the environment resulting from emissions from shipping. However, due to lack of data and knowledge gaps only the pressure categories antifouling paints, ballast water, bilge water, sewage, grey water, food waste, scrubber water and engine exhaust could be used in the Baltic Sea case study.

4.1. Pressures on the Baltic Sea

The total volumes of greywater, bilge water, ballast water and sewage discharged to the Baltic Sea in 2018 are shown in Table 2 and are derived from Jalkanen and Johansson (2019b) while the uncertainties have been determined in this work. Discharged washwater volumes from open- and closed loop scrubbers are also shown in Table 2. The latter data are also based on the study by Jalkanen and Johansson (2019b) but with the use of the new estimates for discharge of scrubber water (90.0 m^3/MWh for open loop and 0.44 m^3/MWh for closed loop) and calculated uncertainties (Supporting material A Table S4).

The total annual pressure of nitrogen due to nitrogen deposition to the Baltic Sea was 18.1 ktonne, with 13.1 ktonne being deposited on the sea basins where nitrogen reductions are required to reach MAI, i.e. Baltic Proper, Gulf of Finland and Gulf of Riga. For grey water, sewage and food waste the corresponding nitrogen loads to the Baltic Sea were 0.16, 0.58 and 0.08 ktonne, respectively, with 0.11, 0.42 and 0.06 ktonne being emitted to Baltic Proper, Gulf of Finland and Gulf of Riga (Supporting material A Table S9).

The pressures to the atmosphere from Baltic Sea shipping correspond to 330 ktonne NO_X, 10 ktonne SO_2, 10 ktonne PM2.5, 24 ktonne CO, 3 ktonne NMVOC and 15.7 Mtonne CO_2 (Jalkanen and Johansson, 2019a). No information has been given for emissions of CH_4. Although relevant when comparing different specific engine types, CH_4 emissions from shipping in the entire Baltic sea are still in 2018 low due to the insignificant use of LNG engines in Baltic sea shipping.

4.2. Impact at midpoint and endpoint level

A detailed description on the concentrations of the contaminants present in the different pressures, the corresponding characterization factor and calculated marine toxicity potential (TP) are shown in Supporting Material B. The toxicity potential of the different pressure sources, normalized to kg 1,4-DCB-eq/m^2 (antifouling paints) or kg 1,4-DCB-eq/m^3 (all other emission sources), are shown in Table 3. Of the antifouling coatings, category D had the highest toxicity potential (0.4 kg 1,4-DCB-eq/m^2) followed by category C (0.26 kg 1,4-DCB-eq/m^2), category B (0.13 kg 1,4-DCB-eq/m^2) and category A (0.05 kg 1,4-DCB-

Table 2

Discharged volumes from ships during 2018 to the Baltic Sea. Data from Jalkanen and Johansson (2019b), but with updated emission factors (scrubber, open and closed) and calculated 95% confidence intervals (CI).

Emission source	average volume (m^3/year)	95% CI Lower	95% CI Upper
Greywater	5,400,000	3,600,000	7,200,000
Bilge water	360,000	n/a	n/a
Scrubber open	153,000,000	123,000,000	184,000,000
Scrubber closed	177,000	80,000	273,000
Ballast water	314,000,000	n/a	n/a
Sewage	1,400,000	660,000	2,000,000

Table 3

Toxicity potential of different emission sources from shipping.

Pressure	kg 1,4-DCB-eq./m^2/d
Antifouling paint category	
A	0.05 (0.04–0.07)
B	0.13 (0.09–0.16)
C	0.26 (0.18–0.34)
D	0.40 (0.37–0.45)
Pressure	**kg 1,4-DCB-eq./m^3**
Ballast water	0.003 (−0.001–0.006)
Bilge water	0.49 (0.19–0.79)
Grey water	0.62 (0.41–0.82)
Scrubber water (open loop)	0.23 (0.12–0.35)
Scrubber water (closed loop)	7.8 (3.3–12.3)
Sewage	0.66 (0.29–1.04)

eq/m^2). Copper contributed to 96.3% of the TP from the antifouling paint category D, whereas zinc (3.7%) and zineb (0.01%) only had a minor impact. A similar pattern was observed for the antifouling category A, B and C were copper contributed between 91.3 and 93.9% of the TP.

When the emission sources were normalized to TP per m^3 discharge water, closed loop scrubber water showed the highest average TP (7.8 kg 1,4-DCB-eq/m^3) followed by sewage (0.66 kg 1,4-DCB-eq/m^3), grey water (0.62 kg 1,4-DCB-eq/m^3), bilge water (0.49 kg 1,4-DCB-eq/m^3), open loop scrubber water (0.23 kg 1,4-DCB-eq/m^3) and ballast water (0.003 kg 1,4-DCB-eq/m^3) (Table 3). For ballast water, in total 40 different BWMS were assessed in the current study. The TP in the effluent water from the different BWMS varied between 4E-7 to 0.08 kg 1,4-DCB-eq/m^3. Copper and zinc were responsible for the main TP in both grey water (68.0% and 28.6%, respectively) and sewage (75.2% and 20.5%, respectively). Vanadium was the key pollutant in both open- and closed loop scrubber water and contributed to 36.5% and 69.5% of the TP, respectively. The total TP from the different emission sources discharged to the Baltic Sea during 2018 are shown in Fig. 2. Antifouling paints had the highest yearly TP (600,000,000 kg 1,4-DCB-eq.), followed by open-loop scrubber water (35,000,000 kg 1,4-DCB-eq) and greywater (3,400,000 kg 1,4-DCB-eq).

The 2018 damage cost due to ship emissions of contaminants to the Baltic Sea and the corresponding impact on marine ecotoxicity was calculated to 582 (279–886) million €$_{2010}$, where emissions from antifouling paints contributed to 545 million €$_{2010}$ (Table 4).

The 2018 emissions from engine exhaust from shipping in the Baltic Sea, as described in 3.3, reduce air quality and lead to numerous adverse
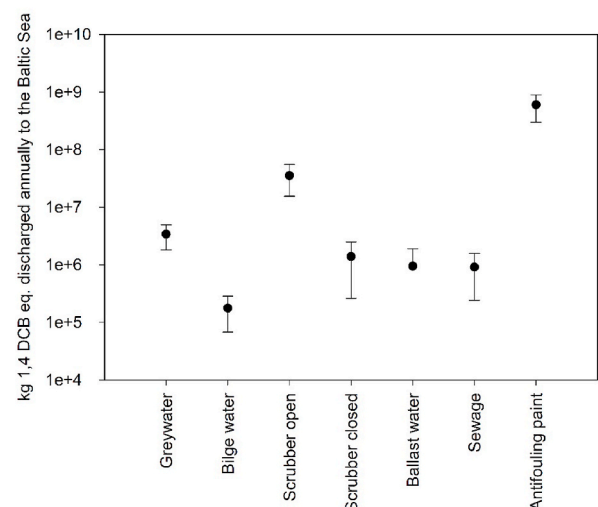
Fig. 2. Toxicity potential (kg 1,4-DCB eq), in log scale, from different pressures to the Baltic Sea during 2018.

Table 4

Damage costs of marine ecotoxicity due to discharge of contaminants from shipping to the Baltic Sea. Valuation in million $€_{2010}$ for 2018.

Emission source	average damage cost, million $€_{2010}$	95% CI Lower	95% CI Upper
Greywater	3.06	1.52	4.6
Bilge water	0.16	0.05	0.27
Scrubber open	32.1	12.7	51.4
Scrubber closed	1.2	0.20	2.3
Ballast water	0.9	−0.025	1.7
Sewage	0.82	0.19	1.4
Antifouling	545	246	843
Sum million € per year	582	279	886

health impacts in the Baltic sea region (Supporting material A Table S10) as well as impacts on climate change. Due to differences in tradition, excess mortality attributable to air pollution is either expressed as life-years lost or as loss of statistical life's, both are presented in Table S10 but are complementary in the economic valuation. In the present study we present results based on life-years lost, which is in line with the way economic valuations are currently used for European air pollution policies, and with the method used in most epidemiological studies. These health effects lead to substantial damage costs and for 2018 it was the emissions of NO_X that caused to largest costs (Fig. 3). In addition, the engine exhaust has radiative forcing properties, and the associated climate change damage costs were in the same order of magnitude as the external costs due to health effects. The most important forcers were CO_2 and NO_X.

4.3. Total valuation from shipping for the Baltic Sea area

The total annual damage costs of Baltic Sea shipping, as calculated in this study, is 2.9 billion $€_{2010}$ (95%-CI 2.0–3.9 billion $€_{2010}$). The distribution in damage costs between the different impact categories are shown in Fig. 4 (with uncertainties presented in supporting material A Table S11). The damage costs due to impacts on marine eutrophication (768 million €) and marine ecotoxicity (582 million €) are in the same range as the total damage costs associated with reduced air quality (816 million €) and climate change (737 million €).

4.4. External costs for a model ship

The total annual damage costs due to emissions to water and air from the model ship were calculated using the ship characteristics in Supporting Material A Table S1, the pressure of nutrients from nitrogen deposition, grey water, sewage and food waste, the TP of different emission sources as described in Table 3, and the associated damage costs due to marine eutrophication, marine ecotoxicity and health
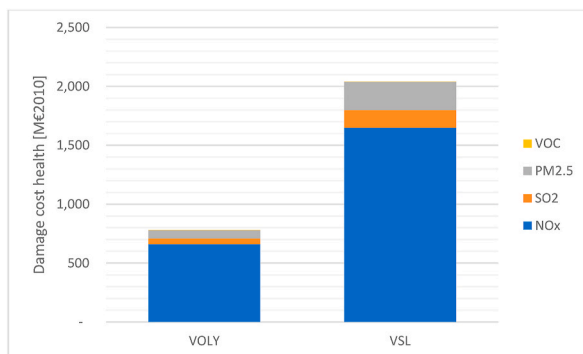

Legend:
- Marine ecotoxicity: 582 million €
- Marine eutrophication (N): 768 million €
- Marine eutrophication (P): 3 million €
- Reduced air quality (NO_x): 660 million €
- Reduced air quality (PM2.5): 93 million $
- Reduced air quality (SO_2): 63 million €
- Climate change (CO_2 eq): 737 million €

Fig. 4. 2018 damage costs (mean values) caused by shipping in the Baltic Sea divided into the impact categories marine ecotoxicity, marine eutrophication, reduced air quality and climate change.

impacts as described in section 3.4. The 2018 damage costs were calculated for the model ship as well as three alternative scenarios of emission reducing technologies. In one scenario the ship does not have an open-loop scrubber, in the next it follows Tier III and in the final it uses biocide-free paint (Table 5). The highest damage costs were related to reduced air quality and marine eutrophication and the most important factor here was emissions to air of NO_X (and nitrogen deposition) which can be seen through the large reduction in damage costs when the ship was assumed to follow the Tier III NO_X standard. The damage cost for marine ecotoxicity was reduced if no scrubber is used as well as for biocide-free paint. The results illustrate that the external impact of any individual ship can vary significantly depending on the environmental performance. That the model ship has negative damage costs in the Climate change category whilst the entire Baltic Sea fleet has positive damage costs is because the model ship being equipped with old Tier I NO_X emission control technology, leading to much higher cooling NO_X emissions per kg CO_2 emission than for the average Baltic sea fleet. The damage costs for a one-way journey for one passenger is $€_{2010}$ 26 for this model ship and reduced to $€_{2010}$ 12 if applying Tier III engine exhaust technology.

5. Discussion

To our knowledge, this is the first integrated framework to assess multi-sphere external costs of ship emissions. Based on the framework, we have assessed the damage costs of shipping in the Baltic Sea considering impacts on the marine environment, reduced air quality and climate change. Shipping is crucial for trade and for many of our worlds' economics but it also create impacts on human welfare through air emissions and pressures on the marine environment. Up till now,

Fig. 3. Central value damage costs due to reduced air quality and associated health impacts of Baltic Sea shipping 2018 engine exhaust, expressed either as value of life-years lost (VOLY) or as value of a statistical life (VSL).
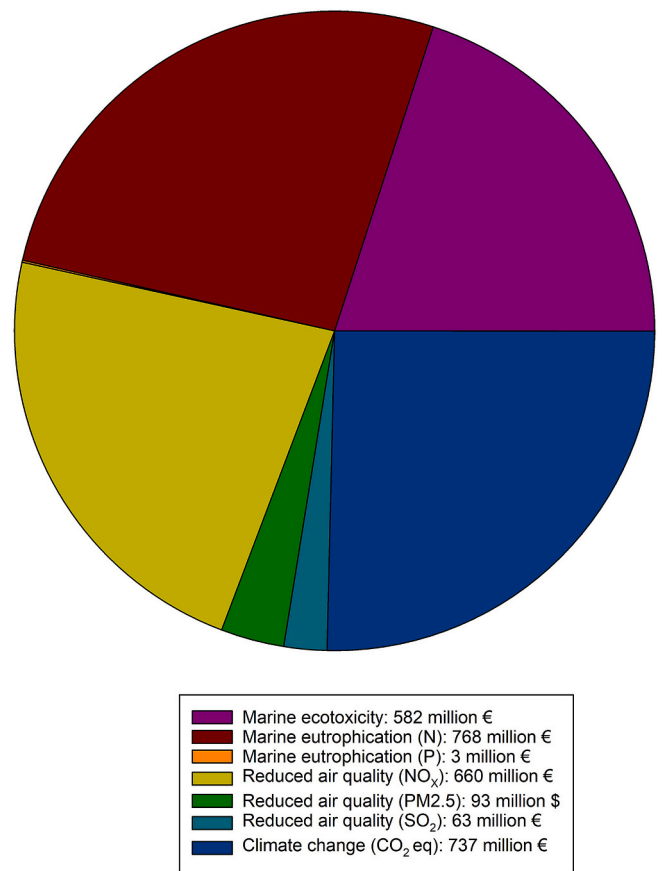
Table 5

Annual (2018) damage costs for the model ship in thousand €$_{2010}$ per year. Range with two standard deviations in brackets.

Impact category	Model ship	Model ship w/o scrubber	Model ship Tier III	Model ship biocide-free paint
Marine ecotoxicity	2094 (1038–3318)	246 (181–330)	2094 (1038–3318)	1878 (837–3074)
Marine eutrophication	4705 (1972–6293)	4705 (1972–6293)	1336 (691–1714)	4705 (1972–6293)
Reduced air quality	3361 (1331–4397)	3361 (1331 - 4397	1121 (509–1321)	3361 (1331 - 4397
Climate change	−286 (-1327–1220)	−286 (-1327–1220)	59 (-909–1030)	−286 (-1327 - 1220))
SUM	9874 (5817–12450)	8026 (4142–10258)	4610 (2699–6036)	9658 (5599–12215)

economic and social analyses of shipping have primarily focused on air pollution assessments. However, the results from the current study show emissions of contaminants and nutrients to the marine environment to contribute to 48% of the total damage cost from Baltic Sea shipping. Deposition of nitrogen on the Baltic sub-basins Baltic Proper, Gulf of Finland and Gulf of Riga contributed to about 25% of the total damage costs from Baltic Sea shipping. Hence, the designation of the Baltic Sea as an emission control area for nitrogen oxides (NECA), will be even more beneficial than what has previously been assessed. In a study by Åström et al. (2018) it was shown that the benefits, in terms of improved air quality and reduced impacts on human health, due to the introduction of a NECA in the Baltic Sea would be 139 million €$_{2010}$ in 2030 while the costs to conform to the regulation will be 111 million €$_{2010}$. If considering also the marine perspective and the benefits of reducing nitrogen deposition to the Baltic Sea, the net benefits would increase from 28 million €$_{2010}$ to ~135 million €$_{2010}$. It must however be emphasized that the calculated damage cost of nitrogen input (marine eutrophication) to the Baltic Sea basins Baltic Proper, Gulf of Finland and Gulf of Riga can be applied to any human activity emitting nitrogen to these Sea basins. In addition, the load of nitrogen to the Baltic Sea from diffusive sources (mainly from agriculture) was in 2014 estimated to be 246 ktonne (HELCOM, 2018), which is 13 times higher than the total load from shipping calculated in this study (18.9 ktonne). Nonetheless, it is important that all sectors, including shipping, contribute to reduce the supply of nitrogen to improve the environmental state of the Baltic Sea.

As shown in Fig. 1, shipping generates multiple waste streams responsible for emissions of hundreds of contaminants to the marine environment. Hence, it is a huge challenge to predict and compare the waste streams' environmental impacts due to emissions of contaminants.

The advantage with the proposed framework is that LCIA allows for a comparison of the toxicity potential between the different waste streams. With the use of LCIA and CFs, we have described the toxicity potential both per volume discharge water (or m^2 for antifouling paints) (Table 3) as well as the total 2018 toxicity potential per pressure (Table 4). This is important knowledge both for the shipping sector as well as for policy makers to prioritize efficient measures for reduced environmental impact from shipping. The damage cost due to emissions of contaminants from shipping to the Baltic Sea was calculated to be 582 million €$_{2010}$ annually, where antifouling paints, due to emissions of primarily copper, caused the highest damage cost, 545 million €$_{2010}$ annually. The emission of copper to the Baltics Sea from ships coated with antifouling paints is significant, 366 tonne annually (Jalkanen and Johansson, 2019b). That can be compared to the total annual input to the Baltic Sea from all waterborne sources (natural and anthropogenic) which have been estimated to be 886 tonne (HELCOM, 2011). Hence, switching to biocide-free antifouling coatings would reduce the total load of copper substantially to the Baltic Sea. Another increasing pressure from shipping is the use of scrubbers where only 85 ships operating in open-loop mode discharged 153 million m^3 to the Baltic Sea in 2018 (Table 2). These ships' scrubber washwater caused a damage cost of 32.1 million €$_{2010}$ (Table 4). The impact of open loop scrubbers' is also shown for the model ship where the damage cost was 1.85 million €$_{2010}$ (Table 5, compare model ship w/and w/o scrubber). It should also be

emphasized that recent statistics from DNV GL suggest that over 4263 ships globally are fitted or on order to be with a scrubber in 2020 (DNV GL, 2020). This is a rapid 10-fold increase as 2017 data suggest only about 401 vessels where equipped with scrubbers globally. If the discharge volume to the Baltic Sea from open loop scrubbers' also increases in a similar rate (10-fold), the damage costs due to marine ecotoxicity would be in the range of 320 million €$_{2010}$. The damage cost of open loop scrubber water is most likely underestimated as the discharge water has a low pH (2–5) and hence impacts on marine acidification, an impact category unaccounted for in this study.

The effects of engine exhaust emissions on air quality and climate change is as mentioned overall better known than the knowledge on the effects on the marine environment and have been discussed extensively (Dessens et al., 2014; Fuglestvedt et al., 2014; Jonson et al., 2014). For air quality and climate change the main challenges relates to future scenarios, where it can be anticipated that the emission mix from shipping will be different in the future due to new air quality and climate change legislations coming into place. These legislations have already decreased SO$_2$ and PM emissions in the Baltic Sea and will after 2021 also reduce NO$_X$ emissions, but through increased use of LNG engines there is a risk that CH$_4$ emissions will increase and offset some of the CO$_2$ and air quality benefits. It is therefore the case the specific values in the present study will need to be recalculated for assessments of future scenarios.

5.1. Limitations and data uncertainties

The developed framework can be used to assess pressures and impact on midpoint level on a global scale. However, damage cost predictions require site-specific pressure and impact assessments and valuation studies, and the damage costs determined here cannot be extrapolated to other sea areas. For the Baltic Sea, we were only able to calculate damage costs for the midpoint impact categories climate change, reduced air quality, marine eutrophication and marine ecotoxicity as characterization factors and socio-economic assessments were absent for the midpoint impact categories biological pollution, marine acidification, marine litter and underwater noise. However, when the scientific knowledge increases within the related scientific disciplines, these impact categories can be included in a more comprehensive assessment of shipping externalities. The cumulative uncertainties in the valuation of marine ecotoxicity and marine eutrophication is a function of uncertainties in a) the prediction of discharge volumes and nitrate deposition, b) concentrations of contaminants and nutrients present in the different waste streams, c) characterization factors and d) valuation (WTP) studies. The predicted discharge volumes of open loop scrubber water, closed loop scrubber water, grey water and sewage are based on 34, 7, 12 and 12 on-board measurements, respectively, while the uncertainty for bilge water and ballast water are unaccounted for. The concentrations of contaminants are based on samples from 40 different Ballast Water Management Systems, 49 samples of bilge water, 95 samples of sewage, 69 samples of grey water, 56 samples of open-loop scrubber waters, 14 samples of closed loop scrubber waters and 145 measurements of biocidal release rates from antifouling coatings. Hence, extensive data have been used to predict volumes and loads of contaminants from different waste streams. However, the damage cost

prediction for marine ecotoxicity and marine eutrophication is based on one respective study only (Ahtiainen et al., 2014; Noring, 2014). Hence, WTP studies should be prioritized to decrease the uncertainties in the damage cost predictions.

Another more ethical uncertainty relates to the choice of whether to consider the value of a statistical life or the value of life years lost in the valuation of human health impacts from air pollution. In the present study we chose to base the valuation on life-years lost from poor air quality. This is in line with how epidemiological studies estimate health effects of air pollution, and as mentioned in line with how the European Commission reasoned when proposing the Clean Air Policy Package in 2013. It does however imply the ethical standpoint that old and vulnerable persons are worth less for society than an average person, a position that deserves a longer discussion elsewhere. For the future studies it is however easy to change the ethical standpoint with basis in the figures presented in Fig. 3.

When valuing climate effects of emissions to air the choice of climate metric has a large effect on the outcome. As an example, due to the inverse short- and long-term radiative forcing properties of NO_X emissions (particle forming vs. ozone precursor), the net climate-related damage cost of NO_X emissions varies substantially as a function of climate metric chosen for the analysis. If using the climate metric Global Temperature Potential (GTP) with a 20-year time horizon, one tonne of NO_X emission would cause climate damages corresponding $-1250 \, €_{2010}$, but if considering emissions accumulated over 20 years (GWP_{20}) one tonne of NO_X emissions would cause climate damage costs of $2200 \, €_{2010}$. Again, as for the discussion on whether to use VOLY or VSL when valuing health damage costs, the choice of climate metric when comparing SLCPs and long-lived greenhouse gases has an ethical dimension (Tanaka et al., 2014).

6. Conclusion

Economic valuation has been used extensively to compare impacts from different transportation modes. These valuation studies have mainly addressed impacts on air pollution and climate change where other impacts on e.g. the marine environment seldom are included. This risks policies to be biased. Open loop scrubbers are a perfect example where atmospheric emissions of SO_2, on the one hand, can be reduced substantially. On the other hand, it also creates a new waste stream where the discharged effluent water is highly acidic and contains a cocktail of organic contaminants (mainly PAHs) and heavy metals. The results from the current study did indeed show the damage costs for the Baltic Sea marine environment to be in the same range as the combined damage costs associated with reduced air quality and climate change. Therefore, it is strongly recommended that legislators on a global (IMO), EU and national level include the marine perspective in future socio-economic assessments of ship emissions. It should however be pointed out that the damage cost for the marine environment determined in this study are underestimated as it includes marine eutrophication and marine ecotoxicity only. Therefore, we recommend that more research should be directed towards other impact categories, mainly underwater noise, marine acidification and biological pollution to be able to comprehensively assess external costs of ship emissions. In addition, the damage costs associated with marine eutrophication and marine ecotoxicity are site-specific for the Baltic Sea and can hence not be used directly to predict impacts and damage costs in other marine water bodies.

Credit author statement

Erik Ytreberg, Conceptualization, Writing – original draft, Formal analysis, Investigation. Stefan Åström, Writing – review & editing, Formal analysis, Investigation. Erik Fridell, Writing – review & editing, Formal analysis, Investigation, Funding acquisition

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 data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.111958.

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