THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

# Holistic assessment of ship scrubbers, with emphasis on the marine environment

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Cover:
Shipping shown from the marine perspective. Illustrating the shift in primary recipient, from air to water, when scrubbers are used to “clean” the exhaust. Illustration by Anna Lunde Hermansson.

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# ABSTRACT

The ocean is essential, providing life support, food and recreation and ocean economy is projected to grow, implying more human activities and ocean claims. A key player in ocean economy is the shipping sector, both as an industry on its own and as a prerequisite for the development of other ocean-based sectors. The daily operations of a ship exert multiple pressures on the environment, affecting climate change, human welfare and ocean health.

As shipping can trigger change in environmental state which may cause negative impact on both the environment and human welfare, there is a need to get a complete perspective of the environmental impacts associated to ship activities. The DAPSIR framework offers a structured approach to assess the cause-effect relationship of society (*Drivers*, *Pressures*), environment and human welfare (*State*, *Impact*) and policy (*Response*). The overall aim of this thesis is to assess pressures, changes in environmental state and impacts on the marine environment following the wide-scale use of scrubbers, an aftertreatment technology to reduce sulphur oxide emissions to the atmosphere by spraying the exhaust with (sea)water, producing large volumes of heavily acidified and contaminated water that is discharged to the marine environment.

The results show that, in comparison to the use of distillate fuels, ships running on residual heavy fuel oil with a scrubber have much higher emissions of metals and polycyclic aromatic hydrocarbons (PAHs) (Paper I) and the discharge of scrubber water can result in one of the largest anthropogenic pressures of certain metals and PAHs on the marine environment (Paper II). When multiple shipping activities are included in the impact assessment, results show that three out of four ports are subject to unacceptable cumulative environmental risk and that scrubber water discharge and leakage of copper from antifouling paint have the largest contribution (Paper III). By applying computational models to fill the experimental ecotoxicological data gap, the risk and impact assessment of substances commonly found in scrubber water could be extended. Examples of substances, previously unaccounted for, are alkylated PAHs that show substantial contribution to the toxicity and environmental risk associated to scrubber water (Paper IV). The economic perspective shows strong economic incentive to install scrubbers, with potentially high socio-economic costs related to the discharge of hazardous substances in scrubber water discharge (Paper V).

It is possible to quantitatively assess activity, pressure and state, while extending the analysis to impact is a challenging task. Despite the challenges, the results from this thesis show that the use of scrubbers is not a sustainable technology and that the use of scrubbers could favour lock-ins in unsustainable patterns of development, enabling the continued use of residual heavy fuel oil.



**Keywords:** ship pollution, marine environment, DAPSIR, scrubbers, metals, polycyclic aromatic hydrocarbons

# SAMMANFATTNING

Havet är viktigt, det ger liv, mat och rekreation och samtidigt förväntas havsekonomin växa, vilket innebär fler mänskliga aktiviteter och havsanspråk. En nyckelaktör i havsekonomin är sjöfartssektorn, både som en egen industri och som en förutsättning för utvecklingen av andra havsbaserade sektorer. Den dagliga driften av ett fartyg innebär belastning på miljön, vilket påverkar klimatförändringar, människors välmående och havets hälsa.

Då sjöfarten kan innebära förändringar i miljötillståndet, med negativ påverkan på både miljön och människors välbefinnande, finns det ett behov av att få ett helhetsperspektiv på miljöpåverkan förknippad med fartygsverksamhet. DAPSIR-ramverket erbjuder ett strukturerat tillvägagångssätt för att bedöma orsak-verkanförhållandet mellan samhället (drivkrafter/*Drivers;* belastningar/*Pressures*), miljö och mänsklig välfärd (tillstånd/*State*; påverkan/*Impact*) och policy (respons/*Response*). Det övergripande syftet med denna avhandling är att bedöma belastningar, förändringar i miljötillstånd och påverkan på den marina miljön till följd av storskalig användning av skrubbrar, en rökgasreningsteknik för att minska svaveloxidutsläppen till luft genom att spreja avgaserna med (havs)vatten, vilket resulterar i stora volymer kraftigt försurat och förorenat vatten som släpps ut i den marina miljön.

Resultaten visar att, jämfört med användningen av destillatbränslen, fartyg som körs på tjockolja med en skrubber harmycket högre utsläpp av metaller och polycykliska aromatiska kolväten (PAHer) (Paper I) och att utsläpp av skrubbervatten innebär en av de största antropogena belastningarna av vissa metaller och PAHer på den marina miljön (Paper II). När flera sjöfartsaktiviteter ingår i konsekvensanalysen visar resultaten att tre av fyra hamnar är utsatta för oacceptabel kumulativ miljörisk och att de största bidragen kommer från skrubbervattenutsläpp och läckage av koppar från bottenfärg (Paper III). Genom att använda beräkningsmodeller för att fylla på den experimentella ekotoxikologiska informationen kan risk- och konsekvensbedömningen av ämnen som återfinns i skrubbervatten utökas. Exempel på ämnen som tidigare inte inkluderats är alkylerade PAHer som visar ett betydande bidrag till toxiciteten och miljörisken förknippad med skrubbervatten (Paper IV). Det ekonomiska perspektivet visar på starka ekonomiska incitament att installera skrubbrar, med potentiellt höga samhällsekonomiska kostnader relaterade till utsläpp av farliga ämnen i samband med skrubbervattenutsläpp (Paper V).

Det är möjligt att kvantitativt bedöma aktivitet, belastning och miljötillstånd, samtidigt som det är en utmanande uppgift att utvidga analysen till påverkan. Trots utmaningarna visar resultaten från denna avhandling att användningen av skrubbrar inte är en hållbar teknik och att användningen av skrubbrar möjliggör fortsatt användning av tjockolja vilket innebär fortsatt inlåsning i ohållbara utvecklingsmönster.

**Nyckelord:** föroreningar från sjöfart, marin miljö, DAPSIR, skrubbrar, metaller, polycykliska aromatiska kolväten

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# LIST OF PUBLICATIONS

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# CONTENTS

## ABBREVIATIONS

| | |
|---|---|
| 1,4-DCB | 1,4-dichlorobenzene |
| AFS | International Convention on the Control of Harmful Antifouling Systems on Ships |
| AI | Artificial Intelligence |
| AIS | Automatic Identification System |
| AM | Arithmetic mean |
| BBNJ | Biodiversity of Areas Beyond National Jurisdiction |
| BLG | Sub-Committee on Bulk Liquids and Gases. Precursor to PPR. |
| BSAP | Baltic Sea Action Plan |
| BSPA | Baltic Sea Protected Areas |
| BWMC | International Convention for the Control and Management of Ships' Ballast Water and Sediments |
| CA | Concentration addition |
| CI | Confidence interval |
| $CO_2$ | Carbon dioxide |
| CV | Coefficient of variation |
| DAPSIR | Driver-Activity-Pressure-State-Impact-Response |
| DNV | Det Norske Veritas |
| EC | European Commission |
| EC10 | Effect concentration at which 10% of the organisms tested exhibit a statistically significant effect |
| ECA | Emission Control Area |
| EEA | European Environmental Agency |
| EEZ | Exclusive Economic Zone |
| EF | Emission factor |
| EGCS | Exhaust Gas Cleaning System |
| EIHA | Environmental Impacts of Human Activities Committee (OSPAR) |
| EMSA | European Maritime Safety Agency |
| EU | European Union |
| EQS | Environmental Quality Standards |
| FAO | Food and Agriculture Organization |
| GES | Good Environmental Status |
| GM | Geometric mean |
| HELCOM | Helsinki Commission, The Baltic Marine Environment Protection Commission |
| HFO | Heavy Fuel Oil |
| HOLAS | HELCOM Holistic Assessment |
| ICCT | International Council on Clean Transportation |
| IEA | International Energy Agency |
| IMO | International Maritime Organization |
| LOD | Limit of detection |
| MAMPEC | Marine antifoulant model to predict environmental concentrations |
| MARPOL | International Convention for the Prevention of Pollution from Ships |
| MEC | Measured Environmental Concentration |
| MEPC | Marine Environment Protection Committee |
| MDO | Marine Diesel Oil |
| MGO | Marine Gas Oil |

| | |
|---|---|
| MSFD | Marine Strategy Framework Directive (2008/56/EC) |
| MWh | Megawatt hour |
| NaOH | Sodium hydroxide |
| NEAES | North-East Atlantic Environment Strategy |
| NECA | Nitrogen Emission Control Area |
| NM | Nautical mile |
| NOEC | No Observed Effect Concentration |
| $NO_X$ | Nitrogen oxides |
| OECD | Organisation for Economic Co-operation and Development |
| OSPAR | The Oslo-Paris Convention, The Convention for the Protection of the Marine Environment of the North-East Atlantic |
| PAH | Polycyclic Aromatic Hydrocarbon |
| PEC | Predicted Environmental Concentration |
| PM | Particulate matter |
| PNEC | Predicted No Effect Concentration |
| PPR | Sub-Committee on Pollution, Prevention Response |
| PSSA | Particularly Sensitive Sea Area |
| QSAR | Quantitative Structure-Activity Relationship model |
| RBSP | River Basin Specific Pollutants |
| RCR | Risk Characterization Ratio |
| RoPAX | Roll-on/roll-off passenger vessel |
| ROS | Regression on Statisics |
| S | Sulphur |
| SDG | Sustainable Development Goal |
| SECA | Sulphur Emission Control Areas |
| $SO_X$ | Sulphur oxides |
| STEAM | Ship Traffic Emission Assessment Model |
| TBT | Tributyltin |
| ULSFO | Ultra Low Sulphur Fuel Oil |
| UN | United Nations |
| UNCLOS | The United Nations Convention on the Law of the Sea |
| UNCTAD | United Nations Trade and Development |
| UNOC | United Nation Ocean Conference |
| VLSFO | Very Low Sulphur Fuel Oil |
| WET | Whole Effluent Toxicity |
| WFD | Water Framework Directive |
| WTP | Willingness to pay |

# 1 INTRODUCTION

The ocean is essential, providing fundamental functions and benefits often taken for granted. Examples of ecosystem services and societal benefits provided by the marine environment include gas and climate regulation, where the ocean absorbs both carbon dioxide ($CO_2$) and heat (https://www.un.org/en/climatechange/science/climate-issues/ocean), and food provision, where the global production from marine areas was 115 million tonnes in 2022 and aquatic animal foods accounting for 7% of all proteins consumed in 2021 (FAO 2024). With more than 2.1 billion people living in the near-coastal zone, i.e. within 100 km from the coast (Reimann et al. 2023), the ocean is an important space for recreational activities and leisure. The ocean also provides overarching support services such as resilience and resistance, withstanding natural and human perturbations while continuing to provide life support and protection (Beaumont et al. 2007, Atkins et al. 2011).

At the same time, ocean economy is projected to grow in the upcoming years, contributing significantly to the global economy and potentially even outperforming the global economic growth rate by 2030 (OECD 2016, Bennett et al. 2019). While efforts are made in understanding and quantifying the cumulative impacts of human activities and ocean claims (e.g. Halpern et al. 2019, Hammar et al. 2020, Jouffray et al. 2020), the interactions of existing and new pressures remain poorly understood (Bennett et al. 2019). With almost 60% of the ocean being subject to significant increase of cumulative impact (Halpern et al. 2019), all sectors must take responsibility in ensuring sustainable ocean development. The shipping industry is a key player in ocean economy, both as an industry on its own and as a prerequisite for the development of other ocean-based sectors (Virdin et al. 2021), and the maritime traffic is forecasted to increase between 240-1200% by 2050 as compared to 2014 levels (Sardain et al. 2019). Hence, the development in ocean economy places the responsibility with policy makers to take (bold) actions based on interdisciplinary science to ensure that ocean industrialization does not conflict with the ecological and social dimensions of the sustainable development goals (Bennett et al. 2019, Jouffray et al. 2020, Virdin et al. 2021).

The shipping industry causes emissions and discharges that adversely affect the (marine) environment, e.g. ecotoxicity in biota, acidification and eutrophication, as well as affecting human welfare (Andersson et al. 2016, Walker et al. 2019, Jalkanen et al. 2021, Moldanová et al. 2022, Ytreberg et al. 2022). The need to get a complete perspective of the environmental risks associated to ship activities was identified by Moldanová et al. (2022) whom proposed a structured approach, following the classical DPSIR framework, to collectively assess different pressures related to shipping. DPSIR is a widely accepted conceptual framework (EEA 1999) to describe the cause-effect relationships between society (*Drivers*, *Pressures*), environment and human welfare (*State*, *Impact*) and policy (*Response*). The framework is also applied within European marine management legislations (e.g. the Water Framework Directive (WFD, (EC 2000)) and the Marine Strategy Framework Directive (MSFD, (EC 2008))) as well as regional sea conventions (e.g. HELCOM for the Baltic Sea area and OSPAR for the North East Atlantic area) to work towards ecosystem-based management (Borja et al. 2010). There are several different alternative forms of the initial DPSIR concept (Elliott et al. 2017), and DAPSIR, where an additional A (for human *Activities*) has been added to nuance the distinction between drivers and pressures (Borja et al. 2006, Atkins et al. 2011), was selected to support the structuring of this thesis.

Ship operations entail numerous pressures on the environment (Figure 1). Examples of pressures that arise from ship activities include introduction of invasive species through biofouling on the ship hull and via ballast water exchange (Sardain et al. 2019), physical damage of seafloor due to anchoring (Watson et al. 2022), leakages of biocides from antifouling paints (Ytreberg et al. 2022), emission of harmful particles from combustion (Corbett et al. 2007), discharge of oily residues in bilge water (Tiselius and Magnusson 2017) and discharge of chemicals from tank cleaning operations (Lunde Hermansson and Hassellöv 2020). Despite the growing collection of research showing several environmentally important aspects of shipping (e.g. Andersson et al. 2016, Jalkanen et al. 2021, Moldanová et al. 2022), efforts are mainly targeted towards climate change and human health issues. For example, the International Maritime Organization (IMO) recently decided to implement drastic measures to reduce $CO_2$ emissions from shipping to nearly zero by the end of 2050 (MEPC 2023a). Consequently, the shipping sector have concentrated their efforts on continuing to achieve this goal, with the risk of overlooking other ship activities and pressures.

A relatively new activity that has been identified as a major concern is the use of exhaust gas cleaning systems (EGCS), commonly known as scrubbers (Turner et al. 2017, Endres et al. 2018, Hassellöv et al. 2020a, Hassellöv et al. 2020b, Tronczynski et al. 2022, Marin-Enriquez et al. 2023). Scrubbers were introduced to the market as an alternative measure to comply with the stricter global and regional sulphur emission regulations by the IMO, from 4.5% m/m (mass sulphur by mass fuel) in 2005 to today's global sulphur cap of 0.5% m/m (since 1 January 2020) with even stricter regulations in sulphur emission control areas (SECAs) (0.1% m/m since 2015)(IMO 2020). The IMO started to limit sulphur oxide ($SO_X$) emissions in ship exhaust and sulphur content in marine fuels as a response to scientific findings from the late 1990s, showing that the global share of emissions of $SO_X$ and nitrogen oxides ($NO_X$) from ship emissions were substantial both with respect to acidic precipitation and human health issues (e.g. Corbett and Fischbeck 1997, Corbett and Winebrake 2008). The 2020 global sulphur cap meant that the 70-80% of the commercial shippers that operate on heavy fuel oil (HFO) (Corbett and Fischbeck 1997), being cheaper with higher sulphur, ash and metal content (Uhler et al. 2016), had to shift to more expensive low sulphur fuels (e.g. marine gas oil (MGO) and very low sulphur fuel oils (VLSFO)) or , alternatively, they could opt for scrubbers and continue to use HFO.

There are three different types of scrubber systems: open loop, closed loop and hybrid scrubbers. The concept of the three systems is similar, where the exhausts are led through a fine spray of water allowing $SO_X$ to dissolve in the water, i.e. forming sulphuric acid, to reduce $SO_X$ emissions to the atmosphere (Karle and Turner 2007). The main difference of the systems is the process and handling of residual products, e.g. discharge water and sludge. In addition of removing $SO_X$ from the exhaust, all scrubbers will scavenge other substances, such as metals and polycyclic aromatic hydrocarbons (PAHs), that will either be discharged to the marine environment and/or collected in the sludge (Turner et al. 2017, Teuchies et al. 2020, Lunde Hermansson et al. 2021, Marin-Enriquez et al. 2023).

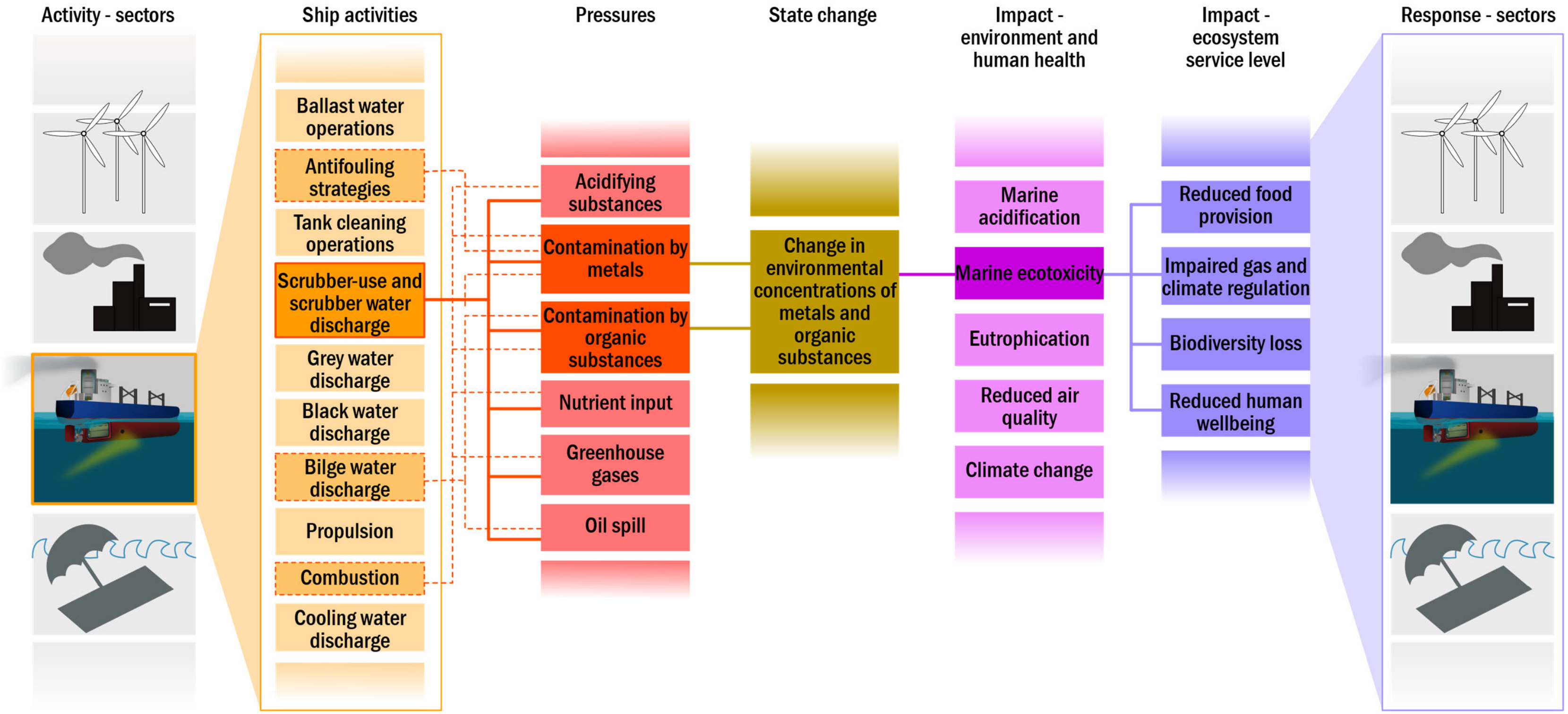


Figure 1: Flow chart illustrating how DAPSIR can be used to structure the cause-effect relationships between ship activities and scrubber-use to pressures, state change and direct impacts and subsequent impact on ecosystem services that may result in responses targeting the shipping sector and other sectors. The lists and connections are non-exhaustive but indicate the focus areas of this thesis (highlighted by the darker boxes).

In the open loop scrubber system (Figure 2A), with >80% of the market share (Ytreberg et al. 2022, DNV 2024), seawater is continuously pumped through the system to act as the scrubber agent where the seawater alkalinity, i.e. buffer capacity, buffers the acid formation and enables sufficient uptake of $SO_X$ (Karle and Turner 2007). In the closed loop scrubber system (Figure 2B) the water is instead recirculated under continuous addition of a strong alkali (often sodium hydroxide ($NaOH$)) to buffer the acid formation and ensure efficient uptake of $SO_X$. For closed loop systems, as the water is recirculated, the sludge (containing a large fraction of the contaminants) is separated and can be collected for onshore disposal. Dedicated closed loop systems are less common with 1% of the market shares (DNV 2024) but hybrid systems exist (approximately 16% of the market share (DNV 2024)) that can switch between open and closed loop mode to enable compliance with different local regulations.

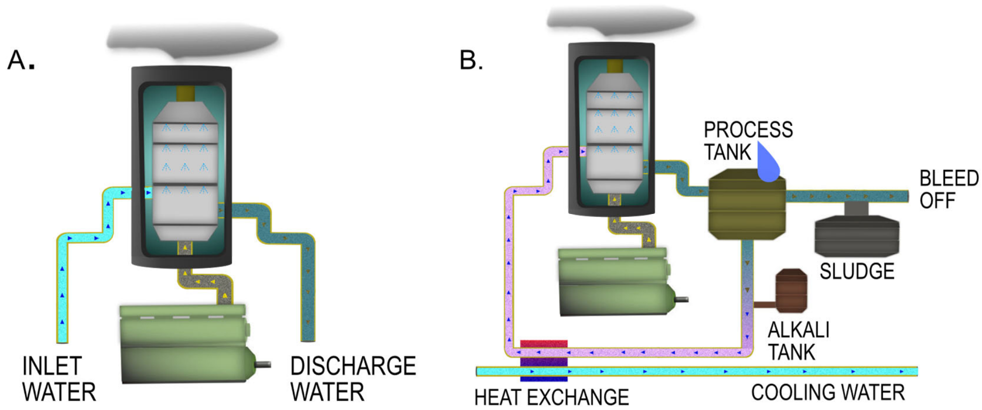


Figure 2: Illustration of open (A) and closed (B) loop scrubber systems. The arrows mark the direction of the water (blue) or exhaust (yellow) through the system. Not to scale.

The use of scrubbers, particularly operated in open loop mode, results in the production of large volumes (hundreds to thousands of cubic meters per hour) of heavily acidified (pH around 3-4) and contaminated wash water that is discharged directly to the marine environment (Turner et al. 2017, Teuchies et al. 2020, Lunde Hermansson et al. 2021). Despite its name, closed loop systems have a "bleed off", i.e. water that needs to be removed from the system, and the discharge volumes are estimated to approximately 1-5 $m^3/h$ (Schmolke et al. 2020, Lunde Hermansson et al. 2021). Globally, Osipova et al. (2021) estimated that the total volumes of discharged scrubber water amounted to more than 10 billion tonnes annually in 2019, which is on par with the volumes of the world seaborn trade of the same year reaching 11.1 billion tonnes (UNCTAD 2020).

As of last year (2023), more than 3500 ships were registered as being equipped with a scrubber (Lunde Hermansson et al. 2024a) and according to DNV, the total number of ships with scrubbers in operation today exceeds 5000 (Figure 3) (DNV 2024). Although ships with scrubbers only represent approximately 5% of the current fleet with respect to number of ships (DNV 2024), the latest report by the International Energy Agency (IEA 2021), forecasted that HFO, in combination with scrubbers, will represent approximately 25% of the total bunker fuel demand in year 2026. Similar conclusions, that ships equipped with scrubbers belong to the larger size-categories with the higher fuel oil consumption, are also confirmed by other studies (Dulière et al. 2020, Teuchies et al. 2020, Jalkanen et al. 2022).

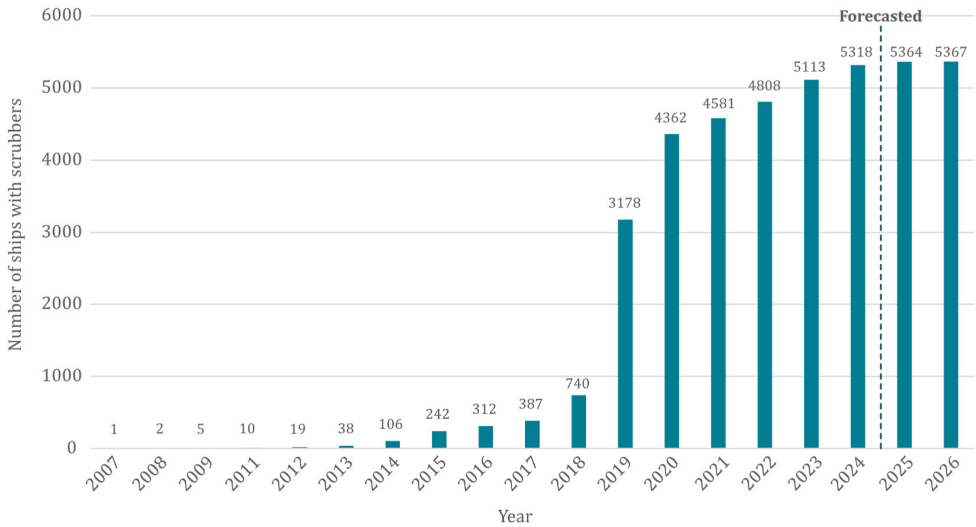


Figure 3: Number of ships with scrubbers installed or on order (DNV 2024).

With an increasing share of the global fleet installing or opting to install scrubbers (Hassellöv et al. 2020a, Ytreberg et al. 2022, DNV 2024), it is crucial to investigate the implications that the scrubber-use and discharges of scrubber water will have on the marine environment. Some of the pressures related to the use of scrubbers include addition of strong acids, nutrients (when nitrogen oxides are scavenged) and, which is the main focus of this thesis, the input of several toxic substances, such as metals and PAHs, to the marine environment (Lunde Hermansson et al. 2021). Several studies show that the impact of scrubber water discharge include adverse effects on marine organisms, e.g. reduced growth and increased mortality (Koski et al. 2017, Thor et al. 2021, Picone et al. 2023), changes in planktonic communities of zooplankton (Jönander et al. 2023), phytoplankton (Ytreberg et al. 2019, Ytreberg et al. 2021a, Genitsaris et al. 2023) and bacteria (Genitsaris et al. 2023, Genitsaris et al. 2024), and acidification effects and consumption of alkalinity on local and regional levels (Hassellöv et al. 2013, Claremar et al. 2017, Turner et al. 2018, Dulière et al. 2020). In addition, the operation of a scrubber entails a fuel penalty at approximately 2-3% due to the energy requirements for e.g. pumping water through the system, resulting in higher $CO_2$ emissions per travelled distance as compared to a ship not having a scrubber installed (Brynolf et al. 2014, Yaramenka et al. 2018). With the implementation of the global sulphur cap, the sulphur content in the ship exhaust has decreased (Claremar et al. 2017, Van Roy et al. 2023), but the scrubbers introduce new problems, shifting the primary recipient from air to water, implying increased pressure on the marine environment. The increased pressure can trigger change in environmental state and may cause negative impact on both the environment and human welfare.

# 2 AIMS AND RESEARCH APPROACH

The overall aim of this thesis is to assess pressures, changes in environmental state and impacts that the wide-scale use of scrubbers has on the marine environment. Several aspects of scrubber-use and environmental impacts are investigated to address the following research questions:

1. What contaminants are being emitted and discharged from HFO combustion and scrubber activities and how do the resulting environmental pressures compare to:
   a) ships using low sulphur MGO?
   b) other waste streams from ships (i.e. antifouling paint, bilge water, near-ship atmospheric deposition)?
   c) other natural and anthropogenic sources (i.e. riverine input, atmospheric deposition and coastal point sources)?
2. How can scrubber water discharge affect the state change and impact in the marine environment and how does that compare to the state change and impact of other ship induced pressures?
3. What are the costs and savings from the shipowner perspective associated to the use of scrubbers and what is the impact on human welfare, as potential societal damage cost, for not taking action to restrict the use of scrubbers as a response?

The overall structure of the thesis follows the DAPSIR framework, where different components are covered in the papers (Figure 4). The work is both interdisciplinary, where concepts and methods from different scientific disciplines are utilized and combined to gain new knowledge, and transdisciplinary, i.e., closely linked to policymakers and stakeholders, to connect the cause-effect relationship of Activity, Pressure, State, Impact and Response components of DAPSIR (Figure 4).

The concentrations and emission factors (i.e. amount of substances emitted per megawatt hour (MWh)) of metals and PAHs from ships operating with different marine bunker fuels, with and without scrubbers, are characterised and compared in Paper I (Lunde Hermansson et al. 2021). In Paper II, the focus is to compare the pressure of metals and PAHs from different human activities, including shipping and the discharge of scrubber water, in the Baltic Sea region (Ytreberg et al. 2022). Paper I and II lay the foundation for the assessment of the state change in port environments, in terms of predicted environmental concentration (PEC) of metals and PAHs from near-ship atmospheric deposition, bilge water discharge, scrubber water discharge and the release of copper and zinc from antifouling paints in four European ports (Paper III, Lunde Hermansson et al. 2023). Further, the derived PECs are used in impact assessment, as cumulative risk, in the same four ports (Paper III). In Paper IV (Lunde Hermansson et al. (in review)), the use of computational methods, in combination with ecotoxicological tests, is explored to identify previously overlooked substances of concern in scrubber water and to improve the prediction and understanding of scrubber water toxicity. Finally, Paper V presents the economic perspective where the reasons for the pending response, regarding new regulations on scrubber water discharge, are discussed and the impact on human welfare is assessed as the cost of not restricting the use of scrubbers.

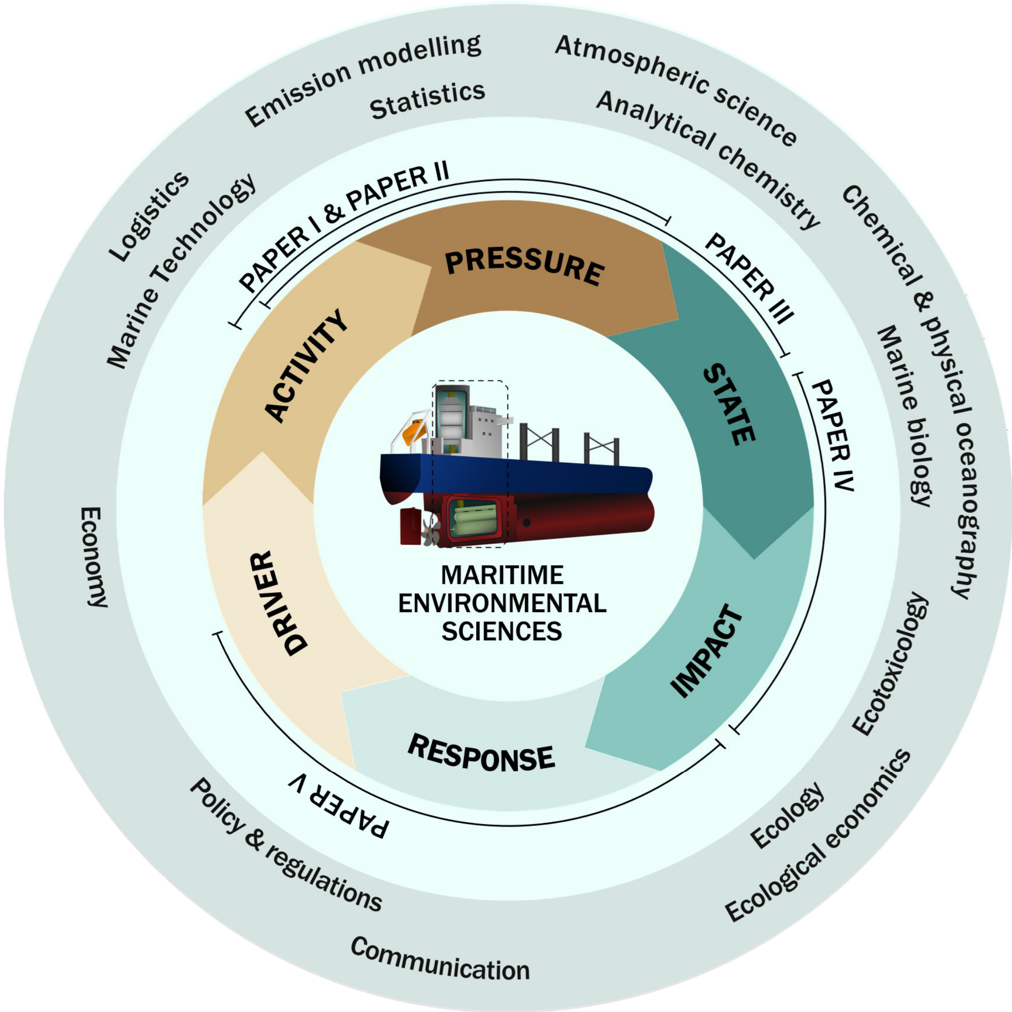


Figure 4: How the five papers of this thesis correlate to the different components of the DAPSIR framework and how the different scientific disciplines in the outer circle encompass the work.

As a response to the current IMO discussions regarding emission factors for scrubbers (PPR 2024), this thesis includes an updated set of concentrations and emission factors of metals and organic substances found in scrubber water. The original dataset (from Paper I and Ytreberg et al. (2020)) has been extended with the addition of more ships and more substances. As part of the updated dataset, the methodology for handling values reported as being below limit of detection is also developed. Due to the dramatic increase of scrubbers over the last years, the spatiotemporal development in scrubber-use is explored and discussed in relation to ocean claims and regulations at different geographic levels.

# 3 BACKGROUND

Shipping is a global industry and the world's merchant fleet now consist of more than 100,000 vessels (UNCTAD 2022). Although efforts are being made to decarbonize the maritime sector (e.g. IMO 2023), over 99% of the ships in operation, and over 80% of the ships on order, operate on conventional, fossil based, fuels (DNV 2024). As a consequence of installing scrubbers, high sulphur HFO still remain a major bunker fuel on the market after the implementation of the global sulphur cap (IEA 2021). At the same time, global, regional and local initiatives envision a future with healthy and resilient oceans, by expanding marine protected areas and reduce pollution as concrete examples (e.g. UNOC 2022, UN 2023). Although scrubbers have been discussed at global (IMO) level, no consensus have been reached on restrictions as some countries and stakeholders claim that more research on scrubbers is needed until any policy measures can be implemented (PPR 2024). However, several countries (e.g. Denmark, Germany, Belgium, China) have already acted within their mandate to put a ban on the discharge of (open loop) scrubber water (ICCT 2023). The case with scrubbers exemplifies the dynamic relationship between environmental policy and industry, where the shipping sector must be able to adjust to the local and regional restrictions while acting on a global market.

## 3.1 POLICY LANDSCAPE CONNECTING SHIPPING AND THE ENVIRONMENT

"Understand and beat marine pollution" is listed as the number one challenge of the United Nations Decade of Ocean Science for Sustainable Development, i.e. the Ocean decade (UNOC 2022). Similar targets are listed under the sustainable development goal (SDG) number 14 (Life Below Water), stating that we should aim "to conserve and sustainably use the oceans, seas and marine resources", and where target 14.1 refer to the prevention and reduction of "marine pollution of all kinds" (UN 2015).

Different organisations and legislations act on different levels of the geographical scope (Figure 5). Shipping operates on all levels and will be influenced by local as well as global regulations. On a global level, the UN and IMO can define goals, introduce regulations and issue guidelines for Member States to ratify and implement locally (on a national level), sometimes via regional unions (e.g. through the European Union (EU)). Nations can also formulate their own regulations, but these should not conflict with the regional or global standards and legislations.

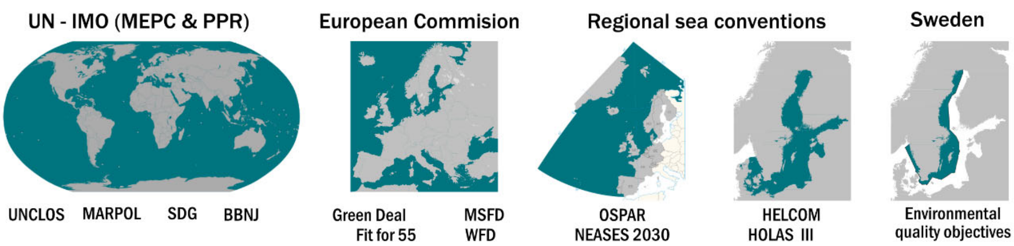


Figure 5: Policy landscape of shipping and the marine environment where different organizations, including Member States, can act within their mandate to establish new regulations, implement new and old regulations, coordinate measures and/or define guidelines and directives to try to achieve a sustainable use of ocean. Non-exhaustive list.

The United Nations Convention on the Law of the Sea (UNCLOS) establishes the global legal framework for international marine and maritime activities. UNCLOS applies to shipping and define the rights and obligations within national jurisdictions of the sea (Figure 6), e.g. territorial water and exclusive economic zones (EEZ). The coastal State have sovereignty over their territorial sea, extending a maximum of 12 nautical miles from the baseline (Figure 6), where the laws and regulations for protection, conservation and/or preservation may be adopted as long as they do not conflict with the global or regional sea conventions. Within EEZ (Figure 6), the coastal State still have sovereign rights of exploring, exploiting and conserving natural resources but other States can enjoy e.g. freedom of navigation and overflight and laying of submarine cables. Part XII of UNCLOS covers the protection and preservation of the marine environment, and the general obligation (Article 192) reads: "States have the obligation to protect and preserve the marine environment" (UN 1982). UNCLOS also covers the areas beyond national jurisdiction, also known as the High Seas (UN 1982, UN 2023). In 2023, the High Seas Treaty (formally known as the agreement on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction (BBNJ)) was adopted under UNCLOS, providing the legal framework to preserve the marine environment and to support the establishment of marine protected areas to help achieve the goal of protecting 30% of the ocean by 2030 (UN 1982, UN 2023). Marine protected areas are designated sites with high nature values where human activities can be partly or fully restricted to ensure conservation. Article 195 of UNCLOS ("Duty not to transfer damage or hazards or transform one type of pollution into another") is also picked up by the High Sea Treaty in Article 7.

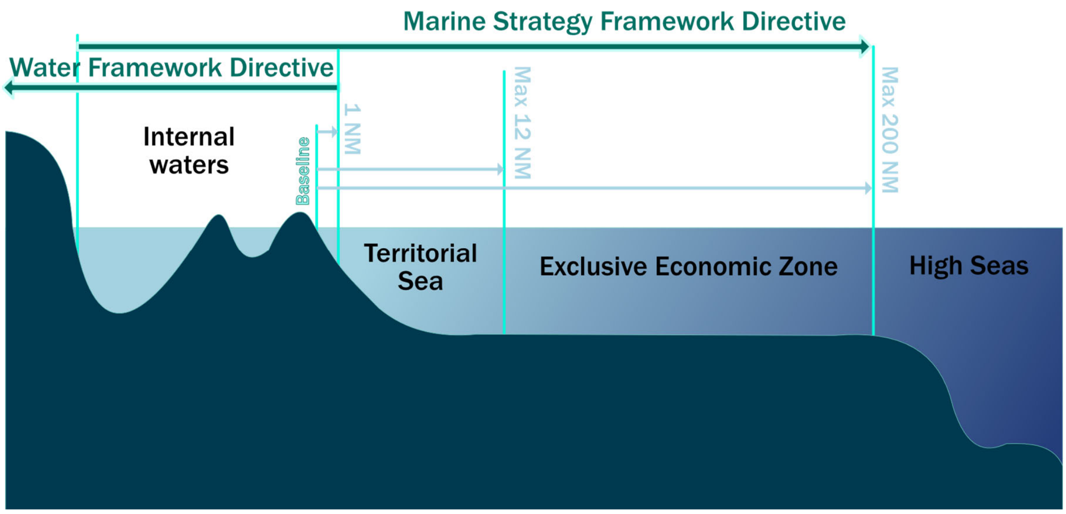


Figure 6: Schematics showing the UNCLOS Maritime zones, definition of water bodies and the spatial coverage and overlap of the European commissioned Water Framework Directive and Marine Strategy Framework Directive.

For shipping, the most comprehensive environmental regulatory framework is the International Convention for the Prevention of Pollution from Ships (MARPOL), to prevent, or at least to minimise, pollution from specific ship operations and accidents. MARPOL consist of six annexes (Annex I-VI), covering different aspects of ship operations (I: Oil, II: Liquid bulk, III: Packaged goods, IV: Sewage, V: Garbage and VI: Air pollution).

The classification of scrubbers as Equivalents (i.e. approved compliance methods to fulfil $SO_X$ requirements of MARPOL Annex VI) is stated in regulation 4 of MARPOL Annex VI, that entered into force 19 May 2005. According to IMO meeting notes (BLG 2011), the availability of scrubbers appears to have been of vital importance in the work towards effectively implementing MARPOL Annex VI with the successive reduction of the maximum allowable sulphur content in fuels. The development of stricter regulations by the IMO has been the forcing driver for shipowners to shift to the more expensive low sulphur fuel oils but the continued use of the cheaper HFO was enabled by allowing the scrubber technology.

The MARPOL convention also defines the areas for which the regulations apply, where the general scope is global but specific regulations may apply to certain areas. Areas that have been identified as more sensitive, both related to the oceanographic and ecological condition and the ship traffic intensity, can be classified as a *Special Areas* where stricter conditions of the regulations, already established by the specific MARPOL annexes, apply. Designated *Special Areas* within Annex VI are referred to as emission control areas (ECAs) where the emissions of $SO_X$ (SECA) and $NO_X$ (NECA) are more strictly regulated than outside ECAs. Today, there are four designated SECAs globally (Baltic Sea, North Sea, North America and United States Caribbean Sea) and, as of 1 May 2025, the entire Mediterranean Sea will be a designated SECA. The IMO can also designate Particularly Sensitive Sea Areas (PSSAs), where general measures, not bound to specific MARPOL annexes, may apply (e.g. routing measures), based on an area's established ecological, socio-economic or scientific value. Although PSSA designation can have certain protection values, the resolution itself does not have legally binding force (Choi 2022).

Issues related to MARPOL and environmental impact from shipping are handled by the IMO Marine Environmental Protection Committee (MEPC) and its subcommittees, for example the sub-committee on Pollution Prevention Response (PPR). PPR is a technical committee in charge of developing the guidelines regarding scrubber operations and scrubber water discharges that is then approved by the MEPC for implementation (e.g. MEPC 2021, MEPC 2022a). The necessity of guidelines for risk and impact assessment of discharge water from scrubbers was acknowledged in 1998 during a MEPC meeting (IMO 1998), several years before scrubbers were introduced to the shipping market (Figure 3). Although environmental concerns were raised from the start, the initial statements in the early versions of the scrubber guidelines, e.g. "ensuring that no added risk or hazard should come from the use of scrubbers", have been removed, or replaced over the course of time (compare e.g. United States (2004), MEPC (2005), BLG (2007) and MEPC (2008)).

At the European level, the European Union (EU) formulates regulations as legally binding directives, and the EU Member States ratifies and follow the regulations within their mandate. The international conventions and regulations regarding shipping is incorporated in European law through several directive. The EU Sulphur Directive (Directive 2012/33/EU of the European Parliament and of the Council of 21 November 2012 amending Council Directive 1999/32/EC) covers the sulphur content regulations of marine fuels and the ship-source pollution prevention directive (Directive 2005/35/EC of the European Parliament and of the Council of 7 September 2005) refers to ship-source pollution and introduce penalties for infringements. More recently, the

FuelEU maritime (EU 2023) initiative (Regulation 2023/1805 on the use of renewable and low-carbon fuels in maritime transport, and amending Directive 2009/16/EC) was implemented to decarbonise the maritime sector as part of the Fit for 55 package and the European Green Deal. The European Green Deal also includes several other strategies and policies of relevance for environmental pressures and impacts related to shipping, e.g. the EU Biodiversity strategy for 2030 including efforts to expand strictly protected areas and the implementation of the Nature Restoration Law which sets binding targets to restore degraded ecosystems on land and at sea. The Blue Economy Strategy is also part of the European Green Deal with objectives to facilitate the coexistence of maritime sectors and to look for synergies in the maritime space that are not damaging to the environment.

The Habitats and Birds Directives (Habitats Directive - Council Directive 92/43/EEC and The Birds Directive - Directive 2009/147/EC) are paramount in EU's priority to preserve and restore biodiversity. According to the directives, the EU Member States must designate areas of importance to protect and preserve biodiversity by establishing protected areas (e.g. marine protected areas, Natura2000). Natura 2000 is the largest coordinated network of protected areas in the world covering 9% of the EU marine territory in 2021. Natura 2000 sites are not wild and human activities are not necessarily excluded from the areas, meaning that discharges from shipping (e.g. scrubber water or tank cleaning effluents) can often continue when ships are sailing through a Natura 2000 area (Hassellöv et al. 2020b, Lunde Hermansson and Hassellöv 2020).

The Water Framework Directive (WFD, 2000/60/EC and 2013/39/EU) and the Marine Strategy Framework Directive (MSFD, 2008/56/EC) are the main governing documents regarding marine and water management and mitigation within EU (Figure 6). The aim of the MSFD is to achieve (and maintain) Good Environmental Status (GES) in European waters and to ensure that the capacity of marine ecosystems, to respond to human-induced changes, is not compromised (EC 2008). The MSFD assessment is based on eleven qualitative descriptors where at least eight are affected by shipping and for example descriptor 8 (D8, "Contaminants are at a level not giving rise to pollution effects") and descriptor 9 (D9, "Contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards") are relevant with regards to pressures related to scrubbers.

The WFD applies to waters within 1 nautical mile from the baseline (Figure 6) and is therefore geographically overlapping with the MSFD in the coastal areas. The WFD also sets goals of achieving Good Status (chemical and ecological) that is directly connected to the GES goal of MSFD. The chemical status is assessed by applying a substance-based approach where the measured environmental concentrations (MECs) of single priority substances are compared to the environmental quality standards (EQSs) of the same substances (defined by EU). In addition to the WFD priority substances, EU Member States that identify substances as pollutants of concern, can add these as River Basin Specific Pollutants (RBSP) with national EQSs and include them when assessing the ecological status in accordance with WFD. The threshold values of the priority substances of the WFD are also valid for the MSFD and failure of achieving Good chemical status thus implies that GES is not fulfilled.

The European Sea conventions take action to implement the strategies and goals of the MSFD. For example, the Convention on the Protection of the Marine Environment in the Baltic Sea Area (the Helsinki-convention, HELCOM) has implemented a strategic programme (Baltic Sea Action Plan, BSAP) to achieve GES. In BSAP, it is stated that sea-based activities should be environmentally sustainable and that the goal is to have a Baltic Sea that is unaffected by hazardous substances. The BSAP present strategic goals and actions to reach the overall vision of a "healthy Baltic Sea" (HELCOM 2021). One of the action points (S22) is dedicated to assessing the use of scrubbers and to develop a Roadmap by 2025 to reduce the input of pollutants from scrubbers. HELCOM also carries out a comprehensive thematic holistic assessment of the State of the Baltic Sea (HOLAS) to monitor ecosystems and evaluate the functioning of the commitment and measures presented in the BSAP. In the latest assessment (HOLAS III, HELCOM 2023b), input of hazardous substances was found at the top of the list of pressures with the highest potential of impact in the Baltic Sea region. HELCOM has also acknowledged shipping as a major source to environmental pressures and input of some hazardous substances (HELCOM 2023a).

Another example of a regional sea convention is the Oslo-Paris Convention to protect the marine environment of the North-East Atlantic (OSPAR 1992) where the Contracting parties have agreed on a strategy for the protection of the North-East Atlantic marine environment (NEAES 2030, (OSPAR 2021)). In the NEASES 2030 strategy, it is concluded that the ocean is at risk and that the OSPAR mission is to "develop and implement the necessary actions to respond to these issues" (i.e. ocean issues) through the implementation of 12 strategic objectives (e.g. "Prevent pollution by hazardous substances" and "Ensure that uses of the marine environment are sustainable"), sub headed by operational objectives, necessary to reach GES. OSPAR can establish legally binding agreements between the Member States and both OSPAR and HELCOM are classified as intergovernmental organizations with observer status within IMO.

## 3.2 ENVIRONMENTAL MONITORING AND RISK ASSESSMENT

Environmental monitoring of hazardous substances and the collected ecotoxicological data forms the basis of the environmental status assessments in accordance with WFD and MSFD and the regional sea conventions. Currently, most assessments apply substance-based approaches where the environmental status is determined by comparing the MEC of a single substance (sometimes substance group) with a threshold value, determined from the same substance in the appropriate matrix (EEA 2019). The substance-based threshold values should represent the concentration at which no effect on the ecosystem is expected (Gustavsson et al. 2023) and can target different matrices (water, sediment, biota), restricted to different conditions (e.g. organic carbon content). Depending on when and how the threshold value was derived, it can vary by several orders of magnitude between nations and sea conventions (Vorkamp and Sanderson 2016, Tornero Alvarez et al. 2022). In an environmental risk assessment, the substance-based risk characterisation ratio (RCR) can be calculated by comparing PEC or MEC to the predicted no effect concentration (PNEC) for the individual substances (Eq.1). To calculate RCR, experimental effect concentrations from ecotoxicological tests are required to determine substance-based PNEC values from a set of reliable and relevant ecotoxicological tests where the response of a single species, exposed to single substances, is reported. The extrapolation from ecotoxicological results to an assumed environmentally safe concentration is done using an assessment factor (Leeuwen and Hermens 1995, Moermond et al. 2016, EC 2018).

$$\mathrm{RCR} = \frac{\mathrm{P(M)EC}}{\mathrm{PNEC}} \tag{1}$$

When PEC is larger than PNEC, then the RCR > 1, implying an unacceptable risk to the environment or, in the case of monitoring campaigns in Europe, failure to achieve GES. In the field, even though substances are found at concentrations below their individual no observed effect concentrations (NOEC), they may still contribute to substantial effects as part of a complex mixture (e.g. Altenburger et al. 2003). Although each chemical is evaluated individually in current assessments (Altenburger et al. 2003), different models are available to assess the toxicity of mixtures. One of the most commonly used is the concentration addition approach (CA) (Faust et al. 2000, de Zwart and Posthuma 2005, Nys et al. 2017). The CA approach assumes that all substances have a similar mode of action, and that any mixture component can be replaced by another (de Zwart and Posthuma 2005, Nys et al. 2017). Several studies have shown that CA is a suitable and conservative approach when predicting toxicity or assessing the risk of mixtures in a worst-case scenario (Escher and Hermens 2002, Silva et al. 2002, Escher et al. 2009, Backhaus and Faust 2012, Altenburger et al. 2018, Bopp et al. 2019, Escher et al. 2020, Jakobs et al. 2020, Lai et al. 2022). CA can be conceptually used in environmental risk assessments where the individual RCRs can be summarised (Backhaus and Faust 2012, Bopp et al. 2019, Lai et al. 2022). A substance-based approach, such as CA, requires that the presence and concentrations of all the substances in the mixture are known and that ecotoxicological test results are available. However, there are often several unidentified chemicals, including transformation products, that contribute to the total effect (Escher et al. 2009, Tang et al. 2014, Brack et al. 2018, Meador and Nahrgang 2019, Escher et al. 2020, Lai et al. 2022).

For some ship-associated pressures, e.g. leakage of antifouling biocides and ballast water treatment, IMO has developed procedures to be followed before new activities, i.e. products, are approved on the market (IMO 2008b, IMO 2008a, IMO 2019). Prior to approving ballast water treatment technologies or new antifouling products, the manufacturer must for example present ecotoxicological test data and emission estimates. The emission estimates can be used to model PEC in predefined areas using the Marine Antifoulant Model to Predict Environmental Concentrations (MAMPEC, Van Hattum et al. 2002). The modelled PEC is then compared to the derived PNEC, based ecotoxicological tests where relevant organisms are exposed to the product(s) under investigation, to assess the environmental risk with introducing the products to the market.

The assessment of scrubbers, however, is rather reactive where the guidelines propose to monitor certain parameters, e.g. pH, temperature, and PAH equivalent, but where sampling and analysis of the scrubber water is voluntary. The 2015 Guidelines for exhaust gas cleaning systems (MEPC 2015) was updated in 2021 (MEPC 2021) and, as these guidelines primarily focus on certification, operational quality assurance and emission factors, it was proposed that a new framework to assess the risk and impact from scrubber water discharge should be developed. The MEPC guidelines on risk and impact assessment of scrubbers (MEPC 2022a) were approved in 2022 providing recommendations that Member States can use as support when making decisions regarding scrubber discharge regulations. The impact assessment, in section 7.4 of the guidelines, stipulates that the adoption of restrictions or a ban on discharge water from scrubbers should be considered in areas where any of four indicative criteria are fulfilled:

*1. environmental objectives in the areas are not met, e.g. good chemical status, good ecological status or good environmental status are not achieved under applicable legislation;*

*2. the discharge of EGCS effluents represents an additional risk of deteriorating the environment and the resiliency of the climate system;*

*3. the EGCS discharge water conflicts with the conventions and regulations formulated to protect the marine environment (see UNCLOS Article 195, etc.); and*

*4. the EGCS discharge effluent represents an increase in the costs of management of dredged materials in ports.*

The guidelines also propose that the cumulative effects of mixtures should be taken into account and that a PEC/PNEC summation approach ($RCR_{sum}$), analogous to CA, is recommended. The scrubber water emission factors can also be applied to the MAMPEC model to calculate PEC while PNEC values are retrieved from already established threshold values (e.g. EQS from the WFD). Similar to the assessments of antifouling products and ballast water treatment systems, the marine environmental risk from the use of scrubbers is only assessed in isolation, i.e. there is currently no requirement to evaluate the cumulative risk or impact connected to all activities and pressures from ships.

# 4 METHODS

A range of different methods was used to assess and, if possible, quantify pressure, state and impact related to ship activities with a focus on the discharge of scrubber water. The work was primarily based on a bottom-up approach where the unique ships could be tracked and, based on the ship-specific activities yielding emissions and discharges, the pressures could be calculated (section 4.1). The change in state was assessed with respect to the changes in environmental concentrations of metals and PAHs in ports (section 4.2) and the environmental impact was assessed as the added risk associated to the increased pressures from different ship activities (section 4.3). The pressure, state and impact assessment from the previous papers was further developed with an extended dataset, with more samples (section 4.1.2), and statistical methods to handle values reported as being below limit of detection from chemical analyses of scrubber water (section 4.1.3). The implementation of computational methods to derive effect concentrations allowed for further development of the environmental impact assessment (section 4.3.2), both with respect to environmental risk and the prediction of scrubber water toxicity (section 4.3.1). The impact on human welfare was also assessed with respect to socio-ecological damage cost (section 4.3.3), where the economic drivers of the shipping industry (and the society) for a (dis) continued use of scrubbers were assessed. If not stated otherwise, MATLAB R2023b (MathWorks Inc., Natick, Massachusetts, USA) was used for data compilation, curation and analysis.

## 4.1 SHIP ACTIVITY AND ENVIRONMENTAL PRESSURES

To estimate the environmental pressures following ship operations, the ship activity data was combined with substance-specific emission factors and concentrations (Figure 7). The activity data was determined from the Ship Traffic Emission Assessment Model (STEAM, Jalkanen et al. (2021)), that allowed for time resolved and georeferenced mapping of ship activities (e.g. volumes discharged of different liquid waste streams) (section 4.1.1).

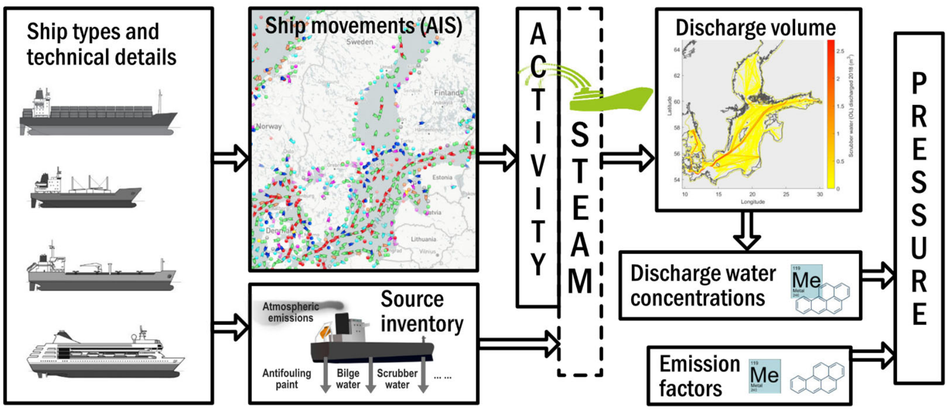


Figure 7: Schematics of how STEAM utilise ship-specific descriptions and automatic identification system (AIS) data to connect ship activity to pressure via source inventories (e.g. discharge flow rate, leakage rate of biocides, fuel consumption) and concentration of substances in the waste streams. Examples of ship types illustration from HELCOM Maritime activities (HELCOM 2018), AIS snap shot from Marine Traffic and discharge volumes are illustrated for open loop scrubbers in 2018 in the Baltic Sea area.

The pressures originating from scrubbers, i.e. load of (hazardous) substances, were calculated from the concentrations of substances in scrubber water (section 4.1.2 and Lunde Hermansson et al. (2021)) and the volumes of discharged scrubber water derived from STEAM. For vessels where scrubber water concentrations were reported together with operational information from the ship (e.g. engine load, discharge flow rate and fuel consumption), fuel-based emission factors (mg substance/kg consumed fuel) and power-based emission factors (mg substance/MWh) could be calculated and used as input in the pressure estimates (section 4.1.2). The MEPC guidelines for scrubbers (MEPC 2021) lists the metals and US-EPA 16 PAHs that should be monitored from onboard sampling. Extended chemical analyses of scrubber water samples were scanned to expand the list of potential substances of concern.

#### 4.1.1 STEAM model to derive discharge volumes

The ship activity was estimated from STEAM (Jalkanen et al. 2021 and references therein), combining the ship location data from automatic identification systems (AIS) with the fleet technical description and the ship-specific modelling of energy consumption. Ship source inventories (e.g. average discharge flow rates of scrubber water, general leakage rate of copper from antifouling paints and emissions from exhaust) were then used in STEAM to compute spatiotemporally resolved emissions to the atmosphere and direct discharges to the marine environment (Figure 7). STEAM uses AIS data positions from all vessels equipped with AIS transceiver which allows for vessel identification and based on vessel databases, consecutive determination of for example the main engine power, gross tonnage, vessel size, engine load et cetera (Jalkanen et al. 2021 and references therein). STEAM is based on real-life ship traffic and the output from the model changes with the variations in ship traffic intensity and the changes in operational variables.

For scrubber-use, STEAM provide open and closed loop discharge volumes based on discharge flow rates of 90 $m^3$/MWh (open) and 0.45 $m^3$/MWh (closed). To translate discharge water volumes to pressure and environmental load of substances of concern, e.g. metals and PAHs (Paper II), scrubber water concentrations of metals and PAHs (μg/L) and power-based emission factors (mg/MWh) were calculated (Paper I). In Paper II and III, STEAM was used together with the concentrations of metals and PAHs (from Paper I), to calculate the contaminant load within a specified geographic area and time, e.g. the entire Baltic Sea or a specific port during a year or a day. As Paper II and III were based on 2018 activity data, where there were fewer ships equipped with scrubbers, a new STEAM model run was conducted for this thesis to compare the scrubber water discharge intensity in 2018 to that of 2022 in the Baltic Sea.

In Paper V, STEAM was used, in combination with high-resolution fuel prices from Ship & Bunker (https://shipandbunker.com/), to calculate ship-specific annual economic balance. Also, an example of socio-economic costs of not restricting discharge of scrubber water was estimated. The output from STEAM then provided ship-specific annual energy and main engine load, fuel consumption of HFO and MGO, amount of discharged scrubber water, amount of energy consumed for scrubber use and kilometres travelled in different sea areas.

### 4.1.2 Concentrations and emission factors of scrubber water constituents

The concentration and emission factors of substances in scrubber water were updated from Paper I, both with respect to added number of ship samples and an with extended list of substances (Lunde Hermansson et al. 2024b). The new data was collected from available literature and online databases, and from an onboard field campaign in November 2021 where simultaneous sampling of ship exhausts and scrubber water were conducted on a container vessel equipped with an open loop scrubber, travelling from Rotterdam, Netherlands, to Gebze, Turkey (Grigoriadis et al. 2022, Grigoriadis et al. 2024).

Following the weighted approach used in Paper I, the average sulphur content of the fuel, where the summarised hourly mass release of sulphur (kg S/h) was divided by the sum of kg fuel consumed per hour (kg fuel/h). The kg sulphur per hour (kg S/h) at each sampling was calculated by multiplying the sulphur content of the fuel (kg S/kg fuel), with the ship-specific engine load (MW) and a default fuel consumption of 180 kg/MWh (assumption based on previous studies e.g. Moldanová et al. 2009, Zetterdahl et al. 2016, Winnes et al. 2018). Similarly, the ship-specific fuel consumption (kg/h) was given by multiplying the default fuel consumption (180 kg/MWh) with the engine load (MW). The reported pH values of discharge water was log-transformed ($[H^+]=10^{-pH}$) before calculating the average and dispersion measures that were then transformed back to the log scale and reported as pH (arithmetic mean ± 95% confidence interval).

The updated statistics on scrubber water concentrations also included new procedures for handling values when substances were reported as being below limit of detection [<LOD] and/or below the reporting limit (from here on collectively referred to as censored values; see section 4.1.3 for further details). Due to the non-normal distribution of most substances, mean and standard deviation might not be the appropriate measure to represent the central value and dispersion (Helsel 1990). To avoid strong influence of outliers, the central value was presented as the geometric mean, i.e. the exponentiated mean of log-transformed data, and the dispersion measure was represented by the geometric coefficient of variation (CV), calculated as

$$CV = \sqrt{e^{s^2} - 1} \tag{2}$$

where s is the standard deviation of the log-transformed data. The geometric mean and CV are appropriate measures of location and dispersion for strictly positive and right-skewed data. For the CV, expressing the dispersion in terms of the standard deviation in relation to the mean, a value <1, 1, or >1 indicates that the standard deviation is less than, equal to, or greater than the mean, respectively.

To (re)calculate emission factors of substances found in scrubber water, concentration of chemical substances together with ship-specific operational data (e.g. discharge flow rate, engine load) were required (Eq. 3).

$$EF_{A,\ water} = [A] \times Q \tag{3}$$

Where EF is the power-based emission factors (mg/MWh), [A] is the concentration of the substances A (μg/L) and Q is the discharge flow rate ($m^3$/MWh) of the scrubber water. If the concentration of substance A was reported as <LOD, the calculated emission factor

was treated as a censored value. For example, for a discharge flow rate of 100 $m^3$/MWh and concentration of A reported as <10 μg/L, the emission factor was treated as <1 g/MWh in the analysis. The results of the new data analysis were compared to the findings from Paper I, II and III and the implications of applying the new concentrations and emission factors were discussed with respect to environmental load and change in scrubber discharges on a Baltic Sea level.

### 4.1.3 Evaluation of methods to treat censored values

The data from analysis of highly complex chemical mixtures (such as scrubber water), that have been analysed with different analytical techniques after, sometimes, unknown sampling procedures, often show highly variable censoring limits (i.e. the limit of detection or reporting limit), sometimes by several orders of magnitude for the same substance (e.g. <1 to <100 μg/L in the case of chromium (Lunde Hermansson et al. 2024b)). However, censored values provide important information about the data distribution and different approaches have been applied to handle censored values in the data analysis. In Paper I and II, a simplified assumption was made where <LOD was defined as ½LOD. In Paper III <LOD was also defined as ½LOD, and a brief sensitivity analysis was conducted, where the results were compared when <LOD were defined as 0, ½LOD or equal to the reported LOD. Although commonly used (Helsel 2005), such simple substitution methods (e.g. setting all values below the limit of detection as equal to ½LOD) have weaknesses and are not deemed appropriate as there are other robust methods available to estimate censored values and to estimate descriptive statistics from data sets containing censored data (Helsel 1990, Helsel 2005).

In addition to the simple substitution approach, two different statistical methods were used to include the censored values and to calculate summary statistics of scrubber water constituents: the regression on statistics (ROS) method (described in Helsel and Cohn (1988)) and the Tobit model (Tobin 1958, Cunillera 2014). The summary statistics, the geometric mean, and the geometric coefficient of variation, were compared for scrubber water constituents by applying different approaches of handling censored values:

1. simple substitution, replacing values <LOD as ½LOD (as done previously)
2. including <LOD according to the ROS method
3. including <LOD according to Tobit model

The ROS method has been described as a robust method that can be applied to data of unknown distribution, with large fractions of censored values (3% (vanadium)-83% (indeno[*123-cd*]pyrene)) and multiple censoring levels (i.e. many different censoring limits for each substance). The ROS method can briefly be described as a probability plotting procedure where the log-transformed data are displayed on a probability plot (Figure 8A). The probability of exceedance ($prob_{exceedance}$) corresponds to the positions (i) of censored and uncensored values that can then be used to calculate the censored concentration at based on linear regression of the uncensored, i.e. observed, concentrations and their given positions (Eq 4, Figure 8B).

$$[A]_{censored,\ i} = k \times prob_{exceedance,\ i} + m \qquad 4$$

Where $[A]_{censored,i}$ is the imputed log-transformed concentration of substance A at position index i, k is the slope and m is the intercept from the linear regression of uncensored values of substance A and their respective $prob_{exceedance}$. The ROS methodology described in Helsel and Cohn (1988) was adopted to an inhouse MATLAB script to assign values to the censored values and to calculate the statistics of the concentrations and emission factors of scrubber water constituents. The applicability of the method, including comparison to other available models, are discussed in detail elsewhere (Helsel and Cohn 1988, Helsel 1990, Croghan and Egeghy 2003, Helsel 2005, Lee and Helsel 2005).

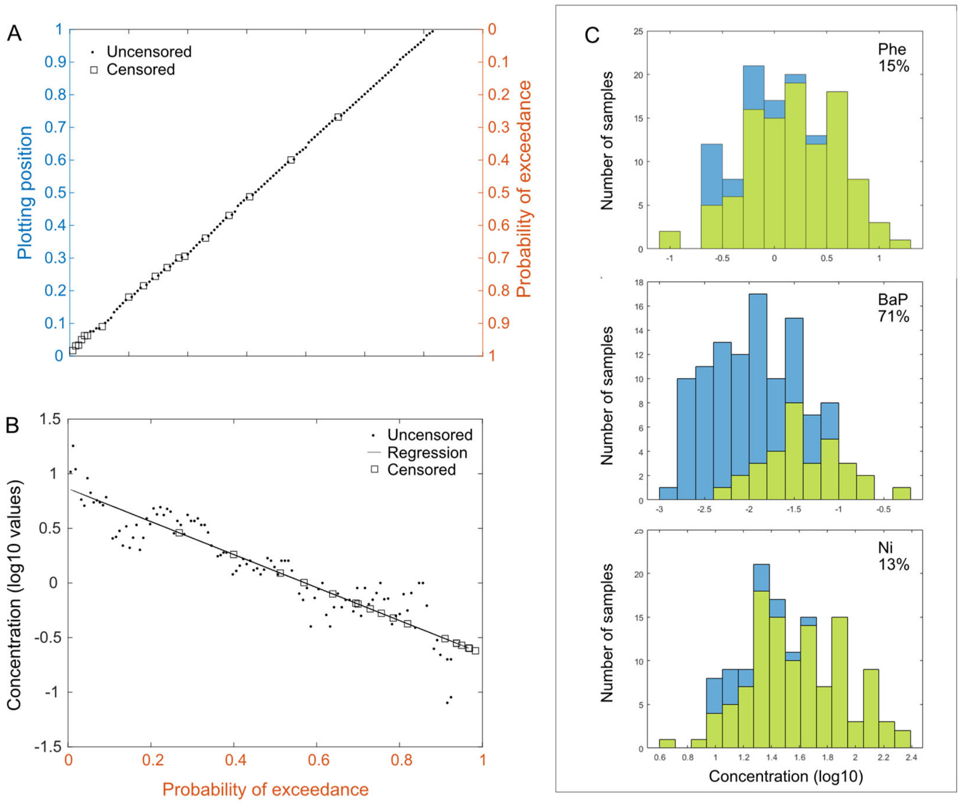


Figure 8: Example with Phenanthrene of how <LOD values are derived. A: probability plot showing the plotting position of the censored squares) and uncensored (dots) data based on probability of exceedance. B: linear regression (black full line) from the log transformed uncensored values (=concentration, displayed as squares) and the probability of exceedance (=1-plotting position in A) and the derived <LOD concentrations of the censored points (stars). Censored data had 8 different "censoring limits" (<LOD values) ranging from 0.1 to 10 ug/L. C: Example of how the distribution of the data changes when adding censored values (blue) to the uncensored (green) data, applying the ROS methodology (fraction of censored values for each substance is indicated in %.. Upper: Phenanthrene (Phe) from derivation shown in A and B. Middle: Benzo[a]pyrene (BaP) with many censored values and Lower: Nickel (Ni) with few censored values.

The other method, the Tobit model, was applied to estimate the summary statistics of scrubber water concentrations and emission factors, using a log-normal regression model accounting for censoring. Unlike ROS, the Tobit model is not an imputation method, i.e. censored values are not imputed, but handled in the estimation procedure when fitting the model to the data using maximum likelihood estimation (Helsel and Cohn 1988). The Tobit model enables the inclusion of predictive variables (e.g. engine load, scrubber water discharge flow rates or sulphur content in the fuel) that could be incorporated in the analysis. Hence, the Tobit model is suitable for inferential and predictive analyses, e.g., comparison between groups or to study the impact of multiple explanatory variables on a response variable (e.g., concentration of a substance in scrubber water). However, the Tobit model is more model dependent, assuming a particular probability distribution for the data, and less robust as compared to the ROS method when a large proportion (e.g. >50%) of the dataset consist of censored values (Helsel 2005). The script for running the Tobit model was implemented using the R language and environment for statistical computing (R Core Team, Vienna, Austria) by statistician Henrik Imberg, using the VGAM library to run Tobit regression.

## 4.2 STATE ASSESSMENT AS CHANGED ENVIRONMENTAL CONCENTRATIONS

The PECs can be estimated from more or less sophisticated hydrodynamic and chemical fate models (e.g. MAMPEC (Huijbregts et al. 2017), ChemicalDrift (Aghito et al. 2023)) where a single discharge or a continuous daily load is used as model input and the physico-chemical characterisation of the substance and the environmental conditions determine the predicted concentration. In Paper III, the simplistic modelling of the hydrodynamic properties of the ports and the chemical properties of the metals and organic substances were used in MAMPEC. MAMPEC is a two dimensional hydrodynamic and chemical fate model that assumes steady-state when calculating PECs in the water and sediment compartment of the defined environment and its surroundings (Van Hattum et al. 2002). As mentioned, MAMPEC has previously been used in risk assessment of antifouling paint products and ballast water treatment systems (Deltares, Van Hattum et al. 2002) and has also been proposed by MEPC as the preferred tool for environmental risk assessment of scrubber water discharge (MEPC 2022a). MAMPEC is divided into three modules; the Environment module, defining the area of interest; the Compound module, defining the chemical characteristics of the substance(s) of interest; and the Emission module that compute the daily loads (g/day) of the different substances to the pre-defined environment. The loads from ships were derived from STEAM model output, for the defined ports in 2018 (Lunde Hermansson et al. 2023).

The influence of the alkylated PAHs on the change in environmental state (and impact) was assessed by extending the PEC modelling with MAMPEC by adding ten alkylated PAHs. The characteristics (e.g. solubility, biodegradation, partitioning coefficients) of the selection of alkylated PAHs were added to the Compound module of MAMPEC based on estimations from the US-EPA software EPISuiteTM (EPIWEB 4.1), favouring experimental values where available. The daily loads of nine metals, US-EPA 16 PAHs and ten alkylated PAHs were then calculated from the geometric means of the concentrations in open loop scrubber water (ROS method to handle censored values) multiplied with the daily discharge volumes determined in Paper III (Lunde Hermansson et al. 2023). The PECs due to open loop scrubber discharge in Port of Copenhagen and Port of Gdynia were modelled and calculated in accordance with the methodology of Paper III but with the updated dataset applying ROS to handle censored values and including alkylated PAHs.

## 4.3 IMPACT ASSESSMENT

The impact of scrubber water discharge was quantified as the added environmental risk, i.e. not including background concentrations or pressures from other activities, (Paper III and section 4.3.1) and as the toxicity potential of scrubber water as a complex mixture (Paper IV). Computational methods were applied to enable inclusion of more substances in the toxicity potential prediction where the predicted response was compared to actual whole effluent toxicity (WET) test results (Paper IV and section 4.3.2). Finally, the cost of not restricting scrubber water discharge was explored by estimating the damage cost connected to marine ecotoxicology of 9 metals and 10 of the US-EPA 16 PAHs in discharged scrubber water (Paper V and section 4.3.3).

### 4.3.1 Impact on the marine environment as added risk

The potential environmental impact was assessed as an added cumulative risk to the environment (Paper III). The substance-based risk was calculated by comparing the PECs of one substance to their respective PNEC value [Eq. 1](Backhaus and Faust 2012). In complex mixtures such as scrubber water, several different substances could cause adverse effects and contribute to the environmental risk, and in Paper III the cumulative environmental risk was calculated as the sum of RCR ($RCR_{sum}$), analogous to the CA concept.

In Paper III, only substances with existing PNEC values (i.e. previously derived and published values) were included in the risk assessment. For several of the newly identified substances in scrubber water, e.g. alkylated PAHs, experimental ecotoxicological tests of marine organisms were lacking and hence, did not have any PNEC values derived. Therefore, four different Quantitative Structure-Activity Relationship (QSAR) models were applied to mathematically derive acute and chronic effect concentrations for different species and endpoints (section 4.3.2 and Paper IV) to estimate PNEC values. This allowed for more substances to be included in the cumulative risk assessment, exemplified for the Port of Copenhagen and Port of Gdynia.

### 4.3.2 Computational Modelling (QSAR) to predict ecotoxicological effects

QSAR models relate the physicochemical properties of a substance to a toxicity endpoint (Hermens 1989, Muratov et al. 2020). QSAR models have been developed from highly stratified data to consider the strong variability in toxicity between different chemical classes and durations of exposure. Regression models are then fitted to the separated data based on for example chemical class, toxicological effects and/or species (Wright et al. 2022). More recent developments have included using different machine learning algorithms to train on larger, more diverse, datasets (Sheffield and Judson 2019, Martin 2020). QSARs have also been developed using artificial intelligence (AI) methods where the chemical structure is interpreted using methods commonly applied in large language models (Gustavsson et al. 2024).

Three regression models (ECOSAR (Wright et al. 2022), T.E.S.T (Martin 2020) and VEGA (Benfenati et al. 2013)) and one machine learning model (TRIDENT (Gustavsson et al. 2024)) formed the basis for the production of effect concentrations, the calculations of toxic units of scrubber water, the derivation of new PNEC values and the subsequent prediction of the toxicity response as a result of scrubber water exposure (Paper IV). The extended dataset of effect concentrations also allowed for calculation of (sum of) toxic units, analogous to the sum of RCRs applying CA, in scrubber water as a measure to predict the toxicity response in WET tests of scrubber water discharge.

#### 4.3.3 Impact on human welfare

An alternative approach to assess the impact from scrubber water discharge on the environment and, consequently, on human welfare was to estimate the environmental damage cost related to marine ecotoxicological deterioration in the Baltic Sea area (Paper V). The damage cost calculations were based on a previous study (Noring 2014, Noring et al. 2016) that valuated ecotoxicological impacts from the organotin compound tributyltin (TBT) in Sweden, based on willingness to pay (WTP) estimates of Swedish households. The WTP estimates for TBT was normalised to the marine toxicity potential, expressed as 1,4-dichlorobenzene equivalents (1,4-$DCB_{eq}$), through the life cycle impact assessment tool ReCiPe (Huijbregts et al. 2017). ReCiPe provided a state-of-the-art life-cycle impact assessment approach where characterisation factors for organic substances and metals for different environmental compartments, including marine waters, have been produced. The harmonised indicator approach enabled a cost conversion for substances present in scrubber water with known toxicity potentials (9 metals and 10 PAHs in total).

The WTP results from Noring et al. (2016) (1.07 $€_{2019}$ /kg 1,4-$DCB_{eq}$ (0.73-1.29 $€_{2019}$ /kg 1,4-$DCB_{eq}$) was multiplied with the cumulative toxicity potential (i.e. kg 1,4-$DCB_{eq}/m^3$) of open and closed loop scrubber water, calculated from characterization factors from ReCiPe (Huijbregts et al. 2017) and concentrations given in Paper I. The resulting cost estimate on scrubber water discharge ($€_{2019}/m^3$) was used to calculate the annual damage cost for marine ecotoxicity ($€_{2019}$/yr), resulting from scrubber discharge water in the Baltic Sea Area (including Skagerrak), by multiplying the total volume of discharge scrubber water in the area (from STEAM) with the marine ecotoxicity damage cost of open and closed loop scrubber water.

### 4.4 Response through Science-Policy interactions

The work of this thesis is based on marine and maritime science of transdisciplinary nature, demanding interaction with stakeholders in industry, policy and society. Since the suitability of scrubbers as a compliance alternative has been raised as a topical issue on global, regional and local level, it was important to ensure that the research findings were made accessible to the policy makers. It was also important to be responsive to the information required by decision-makers. The research questions were developed in an iterative process, with regular interactions between science and policy. Every new insight often resulted in new input to the policy-agenda with new research questions emerging. For example, the motivation for Paper V was directly linked to the concerns raised by certain stakeholders regarding the (economic) “uncertainty for the industry, which has in good faith invested in EGCS technology” (MEPC 2022b) and the wish to factor in the cost of not restricting the use of scrubbers by Member States (MEPC 2023b). The research findings were actively shared at both national and international level where the primary receivers in Sweden were the governmental agencies such as the Swedish Transport Agency, that are leading the Swedish delegation at the IMO meetings of MEPC and PPR, and the Swedish Agency for Marine and Water Management, with responsibility concerning marine environmental management in Sweden. At the regional level, material was submitted and presented at HELCOM Maritime, a HELCOM working group dedicated to shipping.

# 5 Results and discussion

The comparison of emission factors shows that ships running on HFO will emit more metals and PAHs per power output than vessels running on MGO. Therefore, a continued use of HFO, with or without scrubbers, will result in a greater environmental load with respect to metals and PAHs as compared to a shift to MGO (Paper I). The emission factors from HFO combustion also suggest that scrubbers can act as a new source of for example chromium to the environment, where the corrosive nature of hot, acidic seawater result in corrosion of piping and the scrubber-unit (Paper I).

In the Baltic Sea area, when the environmental loads of metals and PAHs from shipping and leisure boating were compared to other sources (atmospheric deposition, riverine input and coastal point sources), shipping is identified as a main contributor (Paper II). Antifouling paint and open loop scrubbers were identified as the ship activities with the largest contribution to the metal and PAH loads. (Paper II). This is noteworthy as the study is based on ship activities in 2018, where only 178 ships operated with an open loop scrubber in the area but still accounted for a substantial fraction (e.g. phenanthrene 8.9%, vanadium 18% (Swedish EEZ)) of the total load (Paper II).

The environmental risk assessment in port environments, based on RCRsum, shows unacceptable risk in three out of four ports (Paper III). Again, antifouling paint and open loop scrubber discharge are identified as major contributors to risk, highlighting the importance of accounting for multiple activities when assessing the pressures and marine environmental impact of shipping (Paper III). The suitability of the proposed new international guidelines on how to assess risk of scrubber water discharge (MEPC 2022a) is also challenged, as the volumes of scrubber water discharge that would be considered to remain within acceptable risk to the environment in the Port of Copenhagen would surpass 200,000 $m^3$ $day^{-1}$, corresponding to half of the total annual discharge volume of scrubber water in the entire Baltic Sea area during 2018 (Paper III).

By applying computational methods, i.e. QSAR models, to derive effect concentrations, more substances can be included in the holistic assessment of scrubbers (Paper IV). Alkylated PAHs have been identified at relevant concentrations in scrubber water (Du et al. 2022, Marin-Enriquez et al. 2023, García-Gómez et al. 2024) and the modelled effect concentrations suggest higher toxicity than exhibited by their parent compounds (Paper IV). Alkylated PAHs are thus proposed to substantially contribute to the toxic response and risk associated to scrubber water exposure (Paper IV). However, when predicted responses, based on CA approach of toxic units, were compared to actual responses, the inclusion of alkylated PAHs is not enough to accurately predict the adverse effects of scrubber water exposure, suggesting there are other substances of concern yet to be identified and/or possible synergistic effects (Paper IV).

Investigation of the economic aspects of installing and operating scrubbers, using HFO instead of low sulphur options, show that a majority of the scrubber-fleet have reached economic break-even by the end of 2022, with a surplus of 4.7 billion $€_{2019}$ (Paper V). When factoring in the cost of not restricting scrubber-use (as requested by e.g. MEPC 2023b), the marine ecotoxicity damage cost, from scrubber water discharge in the Baltic

Sea Area 2014–2022, amounts to >680 million €2019, showing that private economic interests come at the expense of marine environmental damage.

## 5.1 Increased scrubber Activity and Pressure in the Baltic Sea

In absolute terms, the discharges of open loop scrubber water in the Baltic Sea have increased with the most pronounced differences along the shipping lanes with the highest traffic intensity (Figure 9A). When summarised, the discharge from open loop scrubbers in the Baltic Sea area have increased, from 200 million $m^3$ in 2018 to over 300 million $m^3$ in 2022 with a maximum in 2020 where almost 1 billion $m^3$ of scrubber water was discharged in the Baltic Sea area alone (Figure 9B). The 2020 discharge volumes correspond to 10% of the global discharges of scrubber water in 2019, reported by Osipova et al. (2021), in the small semi-enclosed basins of the Baltic Sea (corresponding to an area covering 0.1% of the ocean surface).

The year 2022 appears to be an underestimation in ship traffic intensity, where there is a small decline in number of ships equipped with scrubbers within the Baltic Sea area but a large decrease in the volume of scrubber water discharge as compared to 2020 and 2021. The results indicate that the ships discharged less volumes per time spent in the area, possibly due to less mobility and lower fuel consumption. The lower activity in 2022 could be the result of the dramatic geopolitical change in when Russia invaded Ukraine in February 2022. This can also explain the decreasing trend of open loop discharges in the far east of the Bay of Finland (Figure 9A).

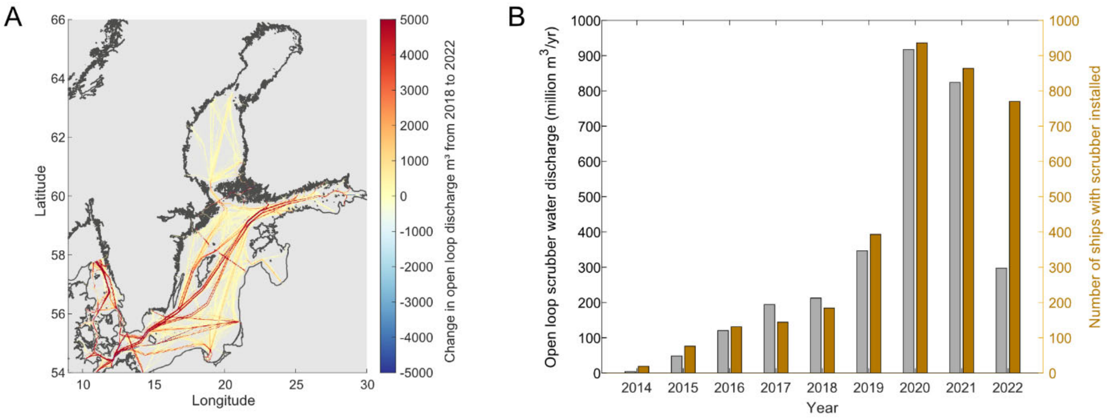


Figure 9: A: The difference of scrubber water discharge volumes, in absolute numbers, in the Baltic Sea between 2018 and 2022. Red colours indicate increase and blue colour indicate decrease. B: Annual discharge (million $m^3$/yr) of open loop scrubber water (grey, left y-axis) and number of scrubbers operating (orange, right y-axis) in the Baltic Sea area from 2014 to 2022.

The increase in discharge volumes result in higher loads of metals and PAHs to the Baltic Sea as compared to the 2018 data presented in Paper II. If all other factors remain as defined in Paper II, the loads from open loop scrubbers increased by 50% from 2018 to 2022 and by almost 400% from 2018 to 2020, the year of the highest recorded discharge volumes. The relative contribution of shipping would thus also increase. For example, contribution of open loop discharges to the vanadium load in the Swedish EEZ increase from 18% (13 tonnes in 2018, Paper II) to 52% (63 tonnes in 2020), if atmospheric deposition and riverine input remain on 2018 levels (at 10 and 47 tonnes vanadium/year respectively). For the entire Baltic Sea, the vanadium load from shipping increase from 35 tonnes in 2018 (Paper II) to 53 tonnes in 2022 with a maximum of 175 tonnes in 2020 based on the annual open loop discharge volumes from STEAM.

Similarly, the direct discharges of the other metals and PAHs increase with higher scrubber activity. The direct discharges of nickel, a priority substance according to WFD Annex I (EC 2013), increase from 10 tonnes/yr in 2018 (Paper II) to 50 tonnes/yr in 2020, being more than three times higher than the contribution from point sources (i.e. industries and waste water treatment plants). The load of nickel from open loop scrubber discharge in the Baltic Sea is of similar magnitude as approximately 13% of the total air emissions of nickel reported for all of the 27 EU Member States in 2020 (EEA 2024).

## 5.2 EMISSION FACTORS TO QUANTIFY ENVIRONMENTAL PRESSURE

The updated dataset on scrubber water content consist of 457 samples, including inlet and outlet water samples from open, closed and hybrid systems and also sludge samples (n=5) from closed loop systems (Lunde Hermansson et al. 2024b). For open loop discharge water, 135 samples from 53 vessels were included as compared to 92 samples from 41 ships included in Paper I (Table 1) (Lunde Hermansson et al. 2021). The average discharge flow rate in the updated dataset (71±6 $m^3$/MWh) was significantly lower than the discharge flow rates in Paper I (91±13 $m^3$/MWh , $p<0.05$ using a two-tailed t-test assuming unequal variances) but higher than the default of 45 $m^3$/MWh suggested by IMO (MEPC 2021). The weighted average of the sulphur content in the fuel was higher, and the average pH lower for the updated dataset (Table 1). The purpose of Paper I was to compare emission factors to water and to air, and due to the often-unstable emissions at low engine loads, only samples taken at engine loads ≥50% were included in Paper I. In the updated dataset, no selection was made based on engine load (on average 62%) and the number of samples increased from 31 to 135 (Table 1).

Table 1: Comparison of the characteristics of open loop scrubber water samples presented in Paper I (Lunde Hermansson et al. 2021) and the updated dataset (Lunde Hermansson et al. 2024b). AM=arithmetic mean and CI=confidence interval.

| **For concentration calculation** | **Paper I** | **Updated dataset** |
|---|---|---|
| Number of vessels | 41 | 53 |
| Number of samples | 92 | 135 |
| Discharge flow rate ($m^3$/MWh) AM ± 95%CI | 91 ± 13 | 71 ± 6 |
| Weighted average sulphur in fuel (% m/m) | 2.2 (0.7–3.2, n=46) | 2.6 (0.56–3.5, n=108) |
| pH scrubber water AM ± 95%CI | 3.9 ± 0.2 (n=36) | 3.5 ± 0.1 (n=92) |
| **For power-based emission factors** | **Paper I** | **Updated dataset** |
| Load selection | ≥50% | 10–92% (average=61%) |
| Number of samples | 31 | 135 |
| Discharge flow rate ($m^3$/MWh) AM ± 95%CI | 68 ± 9 | 71 ± 6 |
| Weighted average sulphur in fuel (% m/m) | 2.29 (0.46–3.48, n=28) | 2.56 (0.46–3.48, n=108) |
| pH scrubber water AM ± 95%CI | 3.67 ± 0.4 (n=7) | 3.46 ± 0.1 (n=92) |

The implementation of simple substitution methods, the ROS method or the Tobit model allow for inclusion of censored values when calculating the summary statistics, e.g. the mean and standard deviation. For the updated dataset, when the censored values are treated as ½LOD and arithmetic mean is reported, the average vanadium, nickel and naphthalene concentrations are similar to the averages in Paper I, while the average concentration of copper, zinc and arsenic is reduced by half (Table 2). The average concentration of the remaining PAHs increased with the new dataset. Since most data are right skewed, the geometric mean was deemed more appropriate than the arithmetic mean, yielding lower central values for all metals and PAHs, confirming the skewness. In addition of favouring the geometric mean instead of the arithmetic mean, the different procedures for handling censored values also affect the concentration estimates. The large coefficient of variation suggests a large dispersion in relation to the mean (Table 2).

When comparing the geometric mean of vanadium and nickel for the three different methods of handling censored values, the ROS method and Tobit model produce slightly higher concentrations than simple substitution. Zinc and copper show similar patterns while for the remaining metals and all the US-EPA 16 PAHs, the simple substitution method yields the highest concentrations. The lower geometric mean could be explained by the larger number of censored values and potential reporting with high censoring limits (Table 2). For alkylated PAHs, there are no/very few censored values and the handling of censored values have little/no influence on the reported geometric mean of the concentrations.

There are several aspects to consider when assessing the suitability of the different methods for handling censored values. As for the simple substitution, this has been a common approach to include censored values, but the method has little scientific robustness and the procedure is not deemed defensible by statisticians (Helsel 1990, Helsel 2005). However, simple substitution methods could encourage ship owners and other stakeholders to strive towards more sensitive analytical methods yielding lower censoring limits to avoid the influence of high reporting limits. The ROS method is robust and simple to perform, there is no need for any special software, and it is easy to include multiple censoring limits (Helsel 1990, Helsel 2005). However, the method is only suitable for descriptive statistics, treating the dataset as one random sample. For inferential statistics, the Tobit model can be applied to perform multivariable analysis (e.g. regression analysis), statistical testing (comparing two or more groups) and allows for framing of more complex problems (e.g. including factors such as experimental design) (Cunillera 2014). Depending on the research question and the framing of the problem, different methods for handling censored values can be applied and it is beyond the scope of this thesis to propose a final method. It can be concluded that, for the substances included in this study, both methods produce similar summary statistics (Table 2). Irrespective of the method used, it is important to report the raw data that is included in the analysis and, when possible, enable the possibility to update the results as more data become available. Robust statistical methods are available to allow for handling of censored values, even for dataset with large number of censored values with multiple censoring limits, and there is no excuse to proceed with simple substitution methods.

Table 2: Concentration of substances found in scrubber water where different approaches for handling censored values have been applied. From the left: the arithmetic mean (AM) from Paper I and AM for the updated dataset, where censored values are treated as ½LOD. The results from applying the three methods for including censored values (simple substitution [½LOD], regression on order statistics [ROS] and Tobit model) are presented with the geometric mean (GM) and coefficient of variation (CV) as these are more appropriate to express the central value and dispersion for strictly positive and right-skewed data. The final column shows the percentage of censored values in the updated dataset for each substance.

| | **Paper I** | **Updated dataset (Lunde Hermansson et al. 2024b)** | | | | |
|---|---|---|---|---|---|---|
| | ½LOD AM (µg/L) | ½LOD AM (µg/L) | ½LOD GM (µg/L) [CV] | ROS GM (µg/L) [CV] | Tobit GM (µg/L[CV] | Fraction censored values (%) |
| **Arsenic*** | 6.8 | 3.7 | 2.98 [0.70] | 2.46 [0.69] | 1.98 [1.2] | 61 |
| **Cadmium** | 0.8 | 1.3 | 0.561 [4.1] | 0.096 [1.2] | 0.031 [1.6] | 78 |
| **Chromium** | 15 | 10 | 4.84 [1.6] | 3.61 [2.7] | 3.47 [3.2] | 52 |
| **Copper** | 36 | 19 | 8.08 [1.8] | 8.51 [1.9] | 7.99 [2.2] | 37 |
| **Lead** | 8.8 | 27 | 1.83 [3.5] | 0.129 [26] | 0.305 [19] | 68 |
| **Nickel** | 48 | 46 | 29.5 [1.4] | 34.7 [1.0] | 33.1 [1.1] | 13 |
| **Selenium** | | 6.5 | 4.87 [0.80] | 4.13 [1.1] | 3.41 [1.5] | 72 |
| **Vanadium** | 170 | 170 | 112 [1.3] | 117 [1.1] | 117 [1.1] | 3 |
| **Zinc** | 110 | 61 | 17.6 [2.6] | 20.0 [2.6] | 19.9 [2.8] | 43 |
| **Naphthalene** | 2.8 | 3.0 | 1.63 [1.7] | 1.46 [1.9] | 1.42 [2.1] | 13 |
| **Acenaphthylene** | 0.12 | 0.45 | 0.218 [2.3] | 0.0617 [2.3] | 0.056 [3.1] | 48 |
| **Acenaphthene** | 0.19 | 0.57 | 0.347 [1.5] | 0.131 [2.1] | 0.110 [3.4] | 45 |
| **Fluorene** | 0.46 | 0.92 | 0.601 [1.1] | 0.402 [1.4] | 0.392 [1.6] | 28 |
| **Phenanthrene** | 1.51 | 2.3 | 1.42 [1.3] | 1.29 [1.4] | 1.21 [1.7] | 15 |
| **Anthracene** | 0.08 | 0.45 | 0.199 [2.7] | 0.025 [2.6] | 0.022 [3.9] | 63 |
| **Fluoranthene** | 0.16 | 0.48 | 0.241 [1.9] | 0.103 [1.5] | 0.094 [1.8] | 40 |
| **Pyrene** | 0.31 | 0.61 | 0.351 [1.8] | 0.139 [2.7] | 0.124 [4.0] | 41 |
| **Benz[*a*]anthrancene** | 0.12 | 0.41 | 0.186 [2.3] | 0.030 [2.7] | 0.025 [4.5] | 58 |
| **Chrysene** | 0.19 | 0.46 | 0.243 [1.8] | 0.078 [2.0] | 0.067 [3.1] | 46 |
| **Benzo[*b*]fluoranthene** | 0.04 | 0.35 | 0.139 [2.5] | 0.014 [2.8] | 0.016 [3.0] | 61 |
| **Benzo[*k*]fluoranthene** | 0.01 | 0.37 | 0.143 [2.9] | 0.011 [1.0] | 0.006 [2.4] | 77 |
| **Benzo[*a*]pyrene*** | 0.05 | 0.38 | 0.156 [2.4] | 0.013 [2.0] | 0.009 [3.6] | 71 |
| **Dibenzo[*a,h*]anthracene*** | 0.07 | 0.38 | 0.145 [3.1] | 0.009 [0.9] | 0.006 [1.8] | 77 |
| **Benzo[*ghi*]perylene*** | 0.02 | 0.35 | 0.152 [2.2] | 0.010 [3.1] | 0.008 [4.7] | 71 |
| **Indeno[*123-cd*]pyrene*** | 0.03 | 0.38 | 0.167 [2.2] | 0.004 [3.5] | 0.003 [7.1] | 83 |
| **2-methylnaphtalene** | | | | 3.75 [1.5] | 3.51 [1.6] | 4 |
| **1-methylnaphtalene** | | | | 2.76 [0.87] | 2.61 [0.93] | 4 |
| **Naphthalene-C2** | | | | 7.25 [1.3] | 7.25 [1.2] | 0 |
| **Naphthalene-C3** | | | | 3.77 [1.5] | 3.77 [1.4] | 0 |
| **Naphthalene-C4** | | | | 1.75 [0.37] | 1.75 [0.36] | 0 |
| **Phenanthrene-C1** | | | | 4.68 [0.80] | 4.68 [0.76] | 0 |
| **Phenanthrene-C2** | | | | 2.72 [0.21] | 2.72 [0.20] | 0 |
| **Phenanthrene-C3** | | | | 1.12 [0.25] | 1.12 [0.24] | 0 |
| **Phenanthrene-C4** | | | | 0.578 [0.38] | 0.58 [0.36] | 0 |
| **Fluorene-C1** | | | | 1.74 [0.77] | 1.74 [0.74] | 0 |
| **Fluorene-C2** | | | | 1.02 [0.28] | 1.02 [0.27] | 0 |
| **Fluoranthene/Pyrene-C1** | | | | 0.64 [1.5] | 0.64 [1.4] | 0 |

*Five of the substances that had large contribution to the cumulative environmental risk in port environments (Paper III) and that were represented by large fractions of censored values.

In the context of Paper II, where it was concluded that open loop scrubbers are a significant source of metals and PAHs to the Baltic Sea, the updated concentration estimates (ROS and Tobit to handle censored values in the updated dataset (Lunde Hermansson et al. 2024b)) do not change that conclusion. Depending on the method for handling censored values, the total load of the individual substances and the scrubbers' relative contribution to the pressure can change but remains substantial. For example, the new concentration estimate for vanadium, applying geometric mean and handling censored values with ROS or Tobit model, decrease by approximately 30%, implying a decreased load from open loop scrubbers by 30%. In Swedish EEZ, open loop scrubbers then contribute to 14% of the total vanadium input (as compared to 18% in Paper II) in 2018. Similarly, the estimated phenanthrene load from open loop scrubbers would decrease with approximately 15%, but the contribution from the 178 ships that operated in the Baltic Sea in 2018 is still 7.6% (as compared to 8.9% in Paper II).

When environmental risk is assessed, the magnitude of PNEC will greatly influence the outcome of the assessment. If the PNEC values are very low, indicating high toxicity, and the analytical methods not sensitive enough, yielding LOD>>PNEC, the method for handling censored values will affect the derived PEC and may thus substantially influence the outcome of the assessment. In Paper III, a sensitivity analysis of treating <LOD as equal to 0, ½LOD or equal to the reported LOD was conducted for five of the substances that had large contribution to the cumulative environmental risk and that were represented by large fractions of censored values (arsenic, benzo[*a*]pyrene, dibenzo[*a,h*]anthracene, benzo[*ghi*]perylene, indeno[*1,2,3-cd*]pyrene, denoted with stars in Table 2). The sensitivity analysis showed that the $RCR_{sum,OL}$ would decrease (LOD=0) or increase (LOD=LOD) by approximately 50% depending on how the censored values were treated. The new approaches for handling censored values and the use of the geometric mean instead of the arithmetic mean of concentrations in scrubber water, results in lower PECs in the ports. However, the low PNEC values, i.e. high toxicity, of the five substances still make their relative contribution to risk relevant.

The fuel consumption, engine load and discharge flow rates on ships can vary which may impact the detected concentrations in scrubber water. Therefore, the presentation of power-based emission factors, where the discharge flow rate (either reported or calculated from discharge flow and engine load), can be more appropriate when comparing emissions and contaminant load. With the updated dataset, the geometric mean of the emission factors of six PAHs (naphthalene, acenaphthene, fluorene, phenanthrene, anthracene and indeno[123-cd]pyrene) are slightly lower than in Paper I, while the rest of the US-EPA PAHs have higher emission factors (Table 3). In addition to the larger number of samples, the extension of allowable engine loads may impact the result as studies show that PAH emissions at lower engine loads can be less stable and deviate from emissions at higher loads (Zhao et al. 2020). For metals, only zinc emission factors are higher in the updated dataset. In contrast to most PAHs, metals are not formed in the combustion process and the change in emission factors cannot be explained by the change in engine load but is rather related to the specific fuel oil and lubrication oil consumption. The metal content in emissions is related to the fuel oil content (Moldanová et al. 2011), lubrication oil content (Eichler et al. 2017) and the engine and scrubber structure where metals can be released to the exhaust or scrubber water (Aragon et al. 2009, Kim and Jeong 2019).

Table 3: Comparison of the median, min and max emission factors from Paper I and calculated geometric mean (GM) and coefficient of variation (CV) from the updated dataset applying ROS and Tobit method for handling of censored values. Coloured columns indicate lower (red) or higher (grey) emission factors in updated dataset. Finally, the min and max emission factors, for the uncensored values, are listed.

| | Paper I (mg/MWh) | | | Update dataset geometric mean (GM) (mg/MWh) and CV | | | | Update dataset (uncensored) (mg/MWh) | |
|---|---|---|---|---|---|---|---|---|---|
| | **Median** | **Min** | **Max** | **ROS GM** | **ROS CV** | **Tobit GM** | **Tobit CV** | **Min** | **Max** |
| **Arsenic** | 270 | 2.5 | 520 | 114 | 0.8 | 103 | 1.6 | 53 | 1600 |
| **Cadmium** | 45 | 0.9 | 100 | 1.98 | 0.8 | 1.38 | 2.5 | 0.4 | 570 |
| **Chromium** | 350 | 25 | 4200 | 195 | 2.6 | 199 | 3.7 | 34 | 13000 |
| **Copper** | 750 | 180 | 14000 | 459 | 2.4 | 462 | 3.0 | 30 | 17000 |
| **Lead** | 270 | 9.2 | 1100 | 173 | 5.3 | 14.4 | 43 | 1.1 | 135000 |
| **Nickel** | 2200 | 520 | 16000 | 2220 | 0.9 | 2110 | 1.1 | 280 | 19700 |
| **Selenium** | | | | 202 | 0.9 | 190 | 1.7 | 108 | 2700 |
| **Vanadium** | 9000 | 1500 | 52000 | 7420 | 1.0 | 7400 | 1.0 | 660 | 52000 |
| **Zinc** | 860 | 220 | 90000 | 1230 | 2.2 | 1230 | 2.9 | 155 | 90000 |
| **Naphthalene** | 180 | 11 | 1300 | 93.8 | 1.9 | 91.7 | 2.0 | 1.4 | 1300 |
| **Acenaphthylene** | 2.7 | 0.2 | 34 | 3.65 | 2.1 | 3.57 | 3.2 | 0.3 | 130 |
| **Acenaphthene** | 13 | 0.4 | 60 | 8.05 | 1.8 | 7.42 | 2.8 | 0.9 | 220 |
| **Fluorene** | 41 | 3.3 | 110 | 25.9 | 1.4 | 25.1 | 1.6 | 3.3 | 590 |
| **Phenanthrene** | 120 | 7.4 | 400 | 82.3 | 1.4 | 77.8 | 1.7 | 7.4 | 1023 |
| **Anthracene** | 1.8 | 0.4 | 25 | 1.74 | 1.8 | 1.62 | 3.6 | 0.4 | 73 |
| **Fluoranthene** | 5.1 | 0.4 | 45 | 6.39 | 1.5 | 5.95 | 2.0 | 0.5 | 420 |
| **Pyrene** | 6.0 | 0.4 | 100 | 8.42 | 2.4 | 8.22 | 3.4 | 0.9 | 202 |
| **Benz[*a*]anthrancene** | 1.2 | 0.2 | 64 | 1.61 | 2.4 | 1.62 | 4.1 | 0.3 | 126 |
| **Chrysene** | 2.7 | 0.2 | 40 | 4.76 | 1.9 | 4.37 | 3.0 | 0.4 | 137 |
| **Benzo[*b*]fluoranthene** | 0.4 | 0.2 | 15 | 0.701 | 3.0 | 0.903 | 4.1 | 0.1 | 32 |
| **Benzo[*k*]fluoranthene** | 0.3 | 0.0 | 5.8 | 0.475 | 1.0 | 0.342 | 2.9 | 0.2 | 39 |
| **Benzo[*a*]pyrene** | 0.4 | 0.2 | 14 | 0.590 | 2.1 | 0.614 | 4.4 | 0.2 | 33 |
| **Dibenzo[*a,h*]anthracene** | 0.4 | 0.2 | 3.2 | 0.476 | 0.6 | 0.402 | 1.4 | 0.2 | 4.9 |
| **Benzo[*ghi*]perylene** | 0.4 | 0.2 | 7.9 | 0.610 | 1.6 | 0.630 | 3.0 | 0.1 | 22 |
| **Indeno[*123-cd*]pyrene** | 0.4 | 0.2 | 26 | 0.151 | 2.5 | 0.201 | 6.8 | 0.2 | 26 |
| **2-methylnaphtalene** | | | | 182 | 1.5 | 165 | 1.8 | 35 | 2400 |
| **1-methylnaphtalene** | | | | 126 | 0.8 | 115 | 1.0 | 35 | 1100 |
| **Naphthalene-C2** | | | | 269 | 1.2 | 269 | 1.1 | 18 | 1100 |
| **Naphthalene-C3** | | | | 140 | 1.2 | 140 | 1.1 | 7.0 | 370 |
| **Naphthalene-C4** | | | | 52.7 | 0.4 | 52.7 | 0.4 | 23 | 92 |
| **Phenanthrene-C1** | | | | 174 | 1.1 | 174 | 1.1 | 61 | 1800 |
| **Phenanthrene-C2** | | | | 81.9 | 0.2 | 81.9 | 0.2 | 54 | 110 |
| **Phenanthrene-C3** | | | | 33.7 | 0.3 | 33.7 | 0.3 | 21 | 46 |
| **Phenanthrene-C4** | | | | 17.4 | 0.4 | 17.4 | 0.4 | 11 | 32 |
| **Fluorene-C1** | | | | 64.8 | 0.8 | 64.8 | 0.8 | 12 | 257 |
| **Fluorene-C2** | | | | 30.7 | 0.3 | 30.7 | 0.3 | 17 | 40 |
| **Fluoranthene/Pyrene-C1** | | | | 21.7 | 1.2 | 21.7 | 1.2 | 1.0 | 110 |

Importantly, if a risk assessment is to be based on worst-case emission factors, the very large range, represented by minimum and maximum values (Table 3), and variability, indicated by high coefficients of variation (Table 3), suggest that the geometric mean may not be the most suitable measure and should be complemented by an upper-level measure. Also, to fully cover the environmental load as a result from the use of scrubbers, emissions to air and water should be accounted for simultaneously (Paper I).

As the updated dataset of scrubber water content showed substantial occurrence of additional substances (e.g. alkylated PAHs (Du et al. 2022, Marin-Enriquez et al. 2023, García-Gómez et al. 2024)), the metal and PAH concentrations in scrubber water are complemented with summary statistics of alkylated PAHs (Table 2 and 3). In the Baltic Sea area in 2020, the load of alkylated naphthalenes and phenanthrenes from open loop scrubber water discharge exceed 19 and 9 tonnes per year respectively. Based on the updated dataset, the sum of the emission factors for the alkylated PAHs exceeds the emission factors of the respective parent PAHs by a factor of 2 (fluoranthene/pyrene) up to more than a factor of 8 (naphthalenes)(Table 3). The alkylated PAHs generally have higher octanol water partition coefficients and lower water solubility than their parent homologues (Kang et al. 2016). The physico-chemical properties will determine the environmental fate of the substances that will affect the change in state and impact in the marine environment. Higher hydrophobicity suggest that the alkylated PAHs are primarily found in the particulate phase, confirmed by the chemical analysis of scrubber water (García-Gómez et al. 2024) and compositional analysis of the particles in scrubber water effluent, showing that PAHs are enriched on the particles and that the particles will act as carriers of both metals and organic substances to the marine environment (Gondikas et al. 2024).

## 5.3 SCRUBBER WATER DISCHARGES AFFECT STATE & IMPACT

### 5.3.1 State and Impact update with new Activity & Pressure data

The new calculations of concentrations and emission factors (section 5.2) also affect the assessment of state change (Paper III). Applying the new loads as pressures, calculated from the geometric mean of the updated dataset (utilizing ROS method to include censored values of concentrations) resulted in lower PECs of most substances from the MAMPEC modelling in port environments. The lower PEC implies that, following the same procedure as in Paper III, the $RCR_{sum}$ for the 9 metals and 16 PAHs discharged with open loop scrubber water is reduced by 70-75% in the four port environments if the discharge volumes remain constant, i.e. same as in 2018. When the contribution from bilge water, closed loop scrubber, near-ship atmospheric deposition and antifouling are included, the reduction of $RCR_{sum}$ varies between 1-30%. However, Port of Gdynia ($RCR_{sum}$=12.9) and Port of Copenhagen ($RCR_{sum}$=2.71) still face unacceptable risk based on 2018 operational data.

The new compilation and risk assessment of scrubber water content reveal that alkylated PAHs are relevant constituents. Based on open loop scrubber water discharge (operational data from 2018), the PECs of the alkylated PAHs ranged from 0.7 ng/L for alkylated naphthalenes in Port of Gdynia to 7 ng/L for alkylated phenanthrenes in the

Port of Copenhagen. The inclusion of alkylated PAHs in the environmental risk assessment in the Port of Copenhagen result in a substantial increase of $RCR_{sum}$, including all sources, from almost 3 to >9. The relative contribution from scrubbers in Port of Copenhagen also increases, from 19% (Paper III) to >70%. A similar trend is seen in Port of Gdynia, where the relative contribution to the cumulative risk increases from 2% ($RCR_{sum,OL}$=0.21, Paper III) to almost 11% ($RCR_{sum,OL}$=1.53) implying that the discharge of open loop scrubber water alone entails unacceptable environmental risk.

If the alkylated PAHs are omitted, only including the substances listed in Paper III, the top ten most toxic substances found in open loop scrubber water remain as top contributors to risk with the updated dataset. The major differences are that the relative contribution of vanadium increases, accounting for 20-47% of the total risk and that the calculated cumulative risk of the open loop discharge is lower due to the lower estimations of load and PECs. The top ten substances from Paper III account for >99% of the cumulative risk associated with open loop scrubber discharge. However, when the alkylated PAHs are included in the analysis, eight out of the previous top ten contributors are replaced and only vanadium and dibenzo[*a,h*]anthracene remain on the list (Figure 10). Alkylated phenanthrenes contribute to 75-80% of the $RCR_{sum,OL}$ in the Port of Copenhagen (figure 10) and Port of Gdynia.

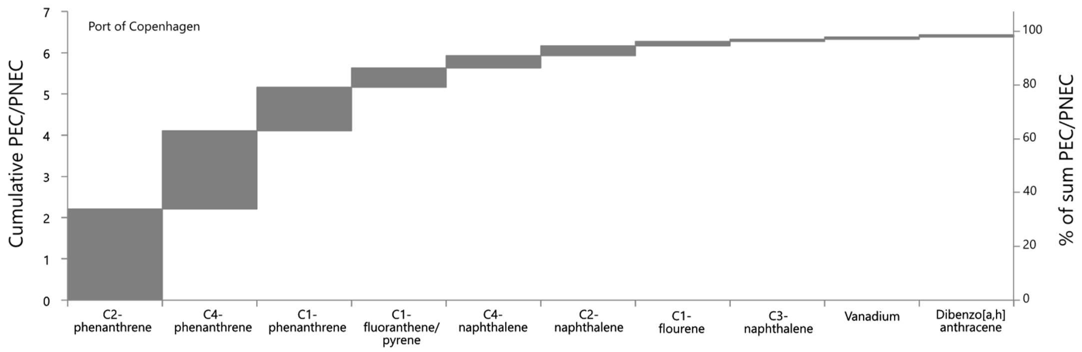


Figure 10: Waterfall plot showing the cumulative PEC/PNEC ratio of open loop scrubber water in the port of Copenhagen (left y axis) and the relative contribution to $RCR_{sum,OL}$ in % is shown on the right y-axis.

Alkylated homologues of phenanthrene have shown to be more toxic to Japanese medaka embryos (*Oryzias latipes*) than the unsubstituted phenanthrene (Turcotte et al. 2011). Similarly, ecotoxicological tests on early life stages of two fish species (*Oncorhynchus mykiss* and *Oryzias latipes*) revealed higher toxicity when exposed to petrogenic mixtures with higher proportions of alkylated PAHs as compared to pyrolytic mixtures (Le Bihanic et al. 2014a, Le Bihanic et al. 2014b). Also, exposure of parent and alkylated PAHs to the nauplii stages of crustacean *Artemia parthenogenetica* showed that alkylated phenanthrene and anthracene exhibited higher acute toxicity than the parent equivalents (Cong et al. 2021). Previous studies have also shown increased photoinduced toxicity of alkylated phenanthrenes (Wassenaar and Verbruggen 2021) and benzo[*a*]anthracene where the photoinduced toxicity could exceed the expected toxicity with one order of magnitude (Boese et al. 1998). Motivated by their presence in scrubber water and their relatively higher toxicity as compared to the parent PAHs (Achten and Andersson 2015, Wassenaar and Verbruggen 2021, Lunde Hermansson et al. (in review)), alkylated PAHs should also be included when assessing pressure and impact from the use of scrubbers and discharge of scrubber water.

### 5.3.2 Other aspects of environmental Impact

In addition to the risk concept, a semi-quantitative assessment of the current knowledge can be compiled to contemplate the potential environmental impact from an increased use of scrubbers. The very large volumes of discharged scrubber water and the high concentrations of toxic substances in the discharge imply high pressures to the marine environment. In the extended risk assessment, based on Paper IV, the PNEC values of the alkylated PAHs are not formally derived based on ecotoxicological experiments but are based on the chronic effect concentrations from the QSAR model runs (Paper IV). The results should thus be interpreted as an indication of increased toxicity with estimates of the relative contribution of alkylated PAHs to the cumulative risk. More importantly, as shown in Paper IV, the added risk will only explain a small fraction of the observed response from the WET tests (Figure 11) when marine organisms are exposed to open loop scrubber water. The added risk/toxic unit concept, only including the identified substances, is most likely an underestimation of hazard and toxicity of scrubber water. The environmental hazards of scrubbers may thus be greater than what can be described from the current substance-based risk assessment, assuming $RCR_{sum}$. The WET tests of scrubber water (e.g. Magnusson and Granberg 2022, Picone et al. 2023, Monteiro et al. 2024) suggest that scrubber water is toxic when diluted 100000 times (equivalent to a NOEC of 0.001%) and that the increased pressure would imply adverse effects on marine organisms with negative environmental impact.

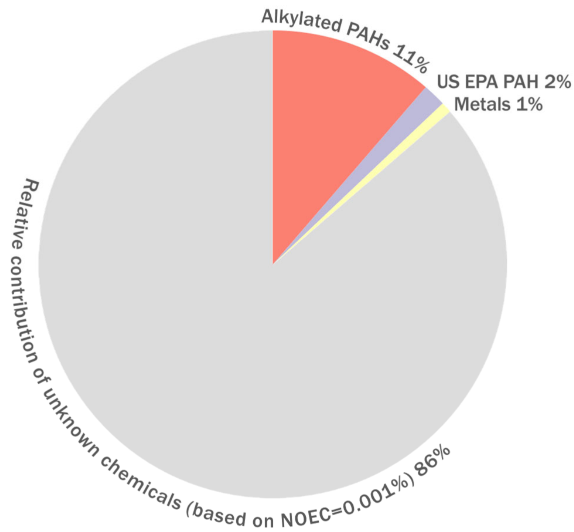


Figure 11: Relative contribution to predicted risk of alkylated PAHs, US-EPA 16 PAHs, metals (based on concentrations retrieved from García-Gómez et al. (2024) and PNECs derived in this publication) and the unexplained effect size required to explain the NOEC of scrubber water exposure corresponding to a dilution down to 0.001%. From Paper IV.

The results from Paper IV (Figure 11), suggest that there are other substances that contribute to the toxicity of scrubber water. Other substances of concern that have been detected in HFO combustion products, scrubber water samples and scrubber sludge samples are for example dioxins and dioxin-like compounds (Magnusson et al. 2018, Marin-Enriquez et al. 2023), dibenzothiophenes (e.g. Streibel et al. 2017), carbonyl compounds such as formaldehyde and acetaldehyde (e.g. Reda et al. 2014, Bendl et al. 2024) and oxidised PAHs that can be more mutagenic and cancerogenic than the parent PAHs (Moldanová et al. 2011, Achten and Andersson 2015). An extended analysis of scrubber water can reveal more substances of concern, analogous to the detection of alkylated PAHS, but the complexity of scrubber water will not be fully understood without the supporting information from WET tests.

Shifting the emissions from air to water means that many of the compounds will not be susceptible to atmospheric chemistry reactions before entering the surface waters (Paper I). The shift of primary recipient will also affect the dispersion of contaminants where atmospheric long-range transport may be reduced but where the local footprint of shipping will increase. Although the differences in degradation processes and environmental fate have not been fully elucidated, the whole effluent exposure of scrubber water to marine organisms reveals high sensitivity, especially in the developmental stages. Therefore, greater impacts on marine ecosystems can be expected if scrubber discharges occur in spawning and nursery areas, especially during periods when organisms are in sensitive developmental stages. In addition, when the highly acidic scrubber water is discharged, this will consume alkalinity which in turn reduces the ocean buffer capacity and potential for $CO_2$ uptake (Hassellöv et al. 2013, Dulière et al. 2020). The acidity may also affect the mobility and speciation of metals, potentially increasing their bioavailability and thus their toxicity (Paper I).

Given that shipping is forecasted to increase by 240-1200% within the next 30 years (Sardain et al. 2019), substantial efforts will be required to reduce the environmental impact from single ships. The results from this thesis show that the use of scrubbers adversely affects the marine environment and that the pressures related to scrubber-use are unproportionally large. At the same time, contaminants, targeting many ecosystem services, have been identified as high-risk pressures, with contribution from many sectors (Aarflot et al. 2024). Thus, the shipping sector will have to co-exist with other sectors, sharing the burden of responsibility not to exceed the planetary boundaries (Rockström et al. 2009, Richardson et al. 2023).

### 5.3.3 Impact on human welfare

The pressures on the marine environment that are related to scrubber water discharge can lead to impacts on the environment and human welfare that can be connected to losses of ecosystem services. The ecosystem services and linked impacts from the discharge of scrubber water can result in socio-economic costs. There is a lack of cost estimates of environmental damage, especially with respect to the marine environment (Ytreberg et al. 2021b) and this can result in biased decision-making where important aspects of costs are overlooked. It is challenging to assess the cause-effect relationships between environmental impact and ecosystem services and also to adequately account for benefits and costs related to the ecosystem services provided by the marine environment (Lew 2015, Moldanová et al. 2022).

In Paper V, the limited valuation of the socio-economic cost of scrubber water discharge in the Baltic Sea, shows a substantial cost related to the input of toxic substances to the marine environment. The average cumulative societal damage cost, by not restricting scrubbers in the Baltic Sea Area since the implementation of SECA in 2015 until 2022, was estimated to 680 million €$_{2019}$, only including characterisation factors from 9 metals and 10 PAHs (Lunde Hermansson et al. 2024a). As shown in previous sections, scrubber water discharge implies increased pressure of more hazardous substances, increased $CO_2$ emissions due to fuel penalty and reduced buffer capacity due to strong acid addition that could all result in costs if shown to contribute to adverse effects. Although not yet quantified as monetary costs, these pressures remain important aspects that should be included when assessing the suitability of using scrubbers as alternative compliance technology.

The WTP methodology can be subject to bias, where the geographical scope is often limited to developed countries and the protection of wildlife is focused on so called "charismatic mega fauna" (Lew 2015). The WTP for marine conservation can also differ depending on if the goods have a value that is consumptive (e.g. harvesting) or non-consumptive (e.g. viewing wildlife) (Lew 2015), demographic characteristics (e.g. age, income and education) (Yu et al. 2018) and the level of environmental awareness (Yu et al. 2018). The WTP in Paper V is based on a study conducted in 2012 (Noring 2014, Noring et al. 2016) with Swedish households being asked about their WTP to prevent the release of TBT paint flakes and decrease the levels of TBT in the environment by natural degradation (108 USD per household and year, corresponding to a summarised Swedish WTP of approximately 500 million USD/year based on 4.7 million Swedish households according to Statistics Sweden). In a more recent Swedish report (Nordzell et al. 2020), the summarised Swedish WTP, to reach GES by 2040, amounted to approximately 900 million USD year (calculated from the 2020 exchange rate 1USD=9.2SEK and WTP of 90 USD/person/yr). When the Swedish WTP from (Nordzell et al. 2020) was compared to other Baltic Sea countries (HELCOM 2023c), the Swedish WTP was higher than Germany and Latvia but lower than Finland. The WTP estimate used in Paper V can thus be representative for the Baltic Sea region but extrapolation beyond the Baltic Sea must be made with care. The temporal (in)stability can be another complicating factor, if the preferences and values change over time, so will the WTP estimates (Lew 2015). Nonetheless, valuating ecosystem services in monetary terms may also highlight the dependence between the wellbeing of the marine environment and the wellbeing of humans.

## 5.4 Science-Policy interactions as a Response to new knowledge

The shipping industry accounts for many different pressures to the environment (Figure 1) but when being regulated or assessed, the scope rarely covers more than one waste stream or activity simultaneously. As showed in Paper III, when accounting for more than one activity and including more than one substance, the cumulative risk was found unacceptable in three out of four ports. At the same time, in European waters and especially in the Baltic Sea, large areas do not fulfil GES with respect to MSFD D8 (Contaminants) (Figure 12).

The environmental status assessment of D8 and D9 is also in many cases inadequate as many Member States have failed to monitor or report the concentrations and/or status assessment. The heterogeneous data coverage creates challenges in linking cause-effect relationships from human activities to pressures, subsequent environmental state changes and their impacts on human welfare (grey areas, Figure 12). In the Priority Substance directive connected to the WFD (EC 2013), 45 substances are included and only few of these (e.g. nickel, cadmium, lead, naphthalene, benzo[*a*]pyrene, anthracene,) will, to a varying degree, represent the pressures associated to ship activities and scrubber water discharge (Paper I, Paper II, Paper III). The relatively large contribution of hazardous substance from scrubbers (Paper II and III), as compared to other ship generated sources as well as other anthropogenic and natural activities, propose that measures restricting scrubber-use and discharge would substantially reduce the pressure on the marine environment.

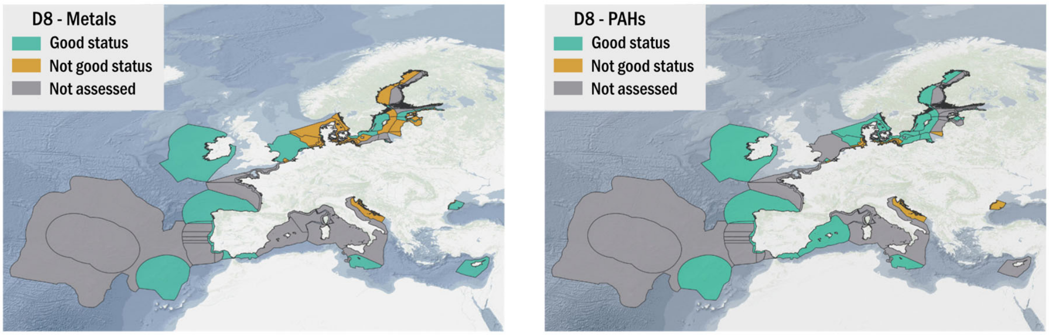


Figure 12: GES reporting Descriptor 8 based on data collected from WISE-Marine (2022).

Within EU, the Industrial Emissions Directive (EU 2010) requires industries to adopt an integrative approach, where the whole environmental performance of the industrial site must be accounted for. Emissions and emission limits should be based on best available techniques and emission data must be reported to the Member State and made publicly available through the European Pollutant Release and Transfer Register if the mass release of substances listed in Annex II are exceeded (EC 2006). For the shipping industry, no such requirements exist. If the reporting requirements, based on annual mass release limits to the water, would be implemented for ships as well, a model RoPax vessel (Ytreberg et al. 2020) operating 5000 hours per year at an engine load of 75% of the maximum continuous rating (23 MW) would be required to report discharges of at least arsenic (10 kg; limit=5 kg/yr), nickel (190 kg; limit=20 kg/yr) and zinc (110 kg; limit=100 kg) solely based on the open loop scrubber discharge (from ROS emission factors in Table 3). Scrubbers also manifest the decoupling between air and water monitoring and reporting where ship exhaust measurements usually include gases (e.g. $CO_2$, $NO_x$, $SO_x$) and particulate matter, while water measurements focus more on metals and PAHs. This can also skew the perception, that the scrubbers clean the exhaust while actually, it is only the primary recipient that changes from air to water. The higher requirements on land-based industries, to adopt an integrative approach, is not yet reflected in the shipping industry. To improve the holistic assessment of the marine environment, shipping should be included in relevant reporting systems such as the HELCOM Pollution Load Compilation and the European Pollutant Release and Transfer Register.

Regulations can be decided and adopted through the IMO, at a global level, or single countries can implement national regulations within the territorial waters (12 nautical miles from land). As the marine environment have no physical border, cooperation, coordination and harmonisation is necessary to develop sustainable and effective measures. Therefore, the input to the policy-discussion must take place at all levels, where the global decision-making within IMO can be a slow process of consensus-based decision-making, and national regulations can be faster but covering a much smaller area. While scrubber water discharge is restricted on local and national levels, with ongoing investigations regarding potential regional bans (e.g. Baltic Sea Action Plan), the global restrictions should be more appropriate as they would, to a larger degree, ensure equity and hinder potential moving of the scrubber fleet from areas, where bans have already been implemented, towards less regulated areas.

The work in this thesis reflects the attempt to balance meeting the policy needs without compromising the scientific methodology and integrity and to advice policymakers and stakeholders on potential measures based on the research findings. Examples of concrete outcomes from the science-policy interactions (Figure 13) include a Governmental report that was utilized as a background document when the Swedish Transport Agency and the Swedish Agency for Marine and Water Management developed a proposal to ban the discharge of scrubber water in Swedish internal waters. The initial proposal was later extended to encompass the Swedish territorial waters and, as of the summer of 2024, the Swedish government announced a referral, proposing to ban the discharge of open loop scrubber water by 1 July 2025 and closed loop discharges in 2029. The concepts of putting scrubber discharge in the context of environmental status assessments, as put forward in Paper I, and relate impact to an increased risk, as put forward in Paper III, was proposed through the EU submission MEPC 76/9/2 (Austria et al. 2021), as a basis for the guidelines on risk and impact assessment of scrubbers (MEPC 2022a), and was implemented in paragraph 7.4 that offers Member States the opportunity to find support in current state and impact assessments when considering potential restrictions of scrubbers. In addition, several papers and reports have been submitted to MEPC and PPR as information documents to support the ongoing discussion on potential measures related to scrubbers.

One key characteristic of the transdisciplinary process is the mutual learning between the scientific community and society that can be driven by a joint problem definition (Scholz 2011). An important aspect of science-communication refers to erroneous citation by different stakeholders (including other researchers), both intentionally and unintentionally (West and Bergstrom 2021). While the unintentional citation errors are difficult to prevent, the intentional misquotations can be limited if the wording and sentence construction are carefully thought trough. As an example, expressing “acceptable risk” in relation to environmental risk assessment becomes a very strong statement and if the statement is not followed by a disclaimer for the conditions of which “acceptable risk” holds true it can often be reinterpreted as “no risk”. Being aware of this becomes more important when the research process is transdisciplinary and when the research directly informs decision-making of polarized issues such as the restrictions of scrubbers.

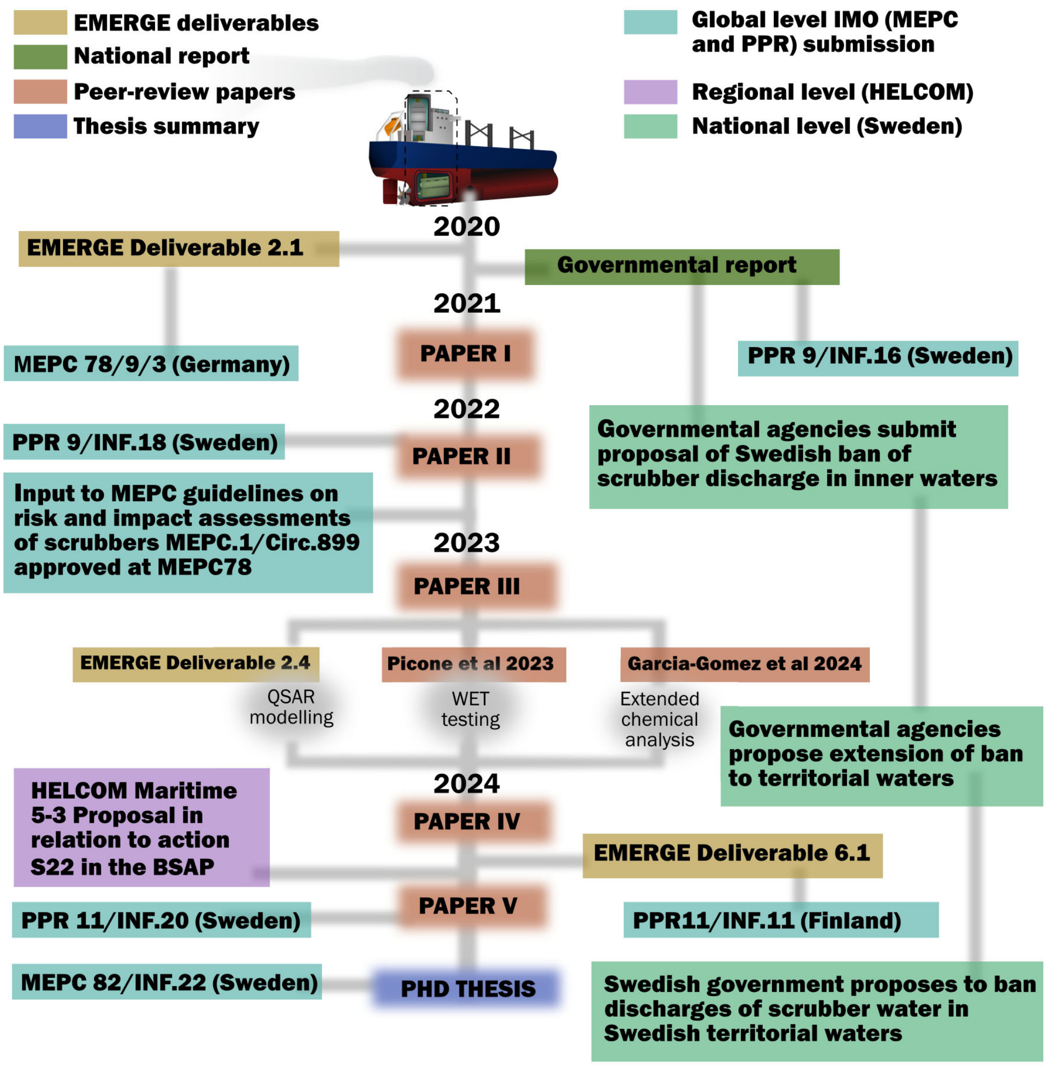


Figure 13: Timeline of the iterative process where the work connected to the papers feed into policy submissions on global (IMO), regional (HELCOM) and national (Sweden) levels. The colours differentiate between deliverables (EMERGE deliverables (EU Horizon 2020 project), national reports and peer reviewed papers) and policy-input (as submissions to IMO, HELCOM and Swedish competent authorities).

The transdisciplinary process can be challenging due to communication difficulties. To avoid misinterpretation or misuse of research findings, conveying uncertainty is essential (Fischhoff and Davis 2014). From a research perspective, uncertainty analysis often strengthens the confidence of the results and provides important information to the overall analysis. From a layman's perspective, on the other hand, presentation of uncertainties can instead overwhelm the recipient, create doubt or, if omitted, create false confidence in the results. With this work, I hope to contribute to the overall discussion on uncertainty (e.g. inclusion of censored values and the importance of presenting dispersion measures) and to highlight that if conclusions can be drawn despite high uncertainty (e.g. emissions of contaminants from HFO as compared to MGO, Paper I), this should be communicated as a robust result.

# 6 CONCLUSIONS

The DAPSIR framework offers a structured approach to holistically assess the use and impact of ship scrubbers. The interdisciplinary approach, collecting knowledge and utilizing tools and methods from many different disciplines, have been paramount in the work and the DAPSIR framework assist in structuring the work and connecting the social, environmental and policy dimensions of the assessment. The overall aim of this thesis is to assess the pressures, environmental state changes, and impact on the marine environment and human welfare from ships equipped with scrubbers.

To summarise, ships that have opted for the use of scrubbers, to continue to operate on cheaper HFO, will cause higher emissions of metals and PAHs, resulting in larger loads, i.e. higher pressure, as compared to ships operating with MGO [answer to RQ1a]. When scrubbers are compared to other ship-activities (antifouling strategies, bilge water discharge and near-ship atmospheric deposition), the scrubbers have the largest contribution to the contaminant load for all PAHs and most metals (antifouling strategies result in higher emissions of copper and zinc) [RQ1b]. Also, when put in relation to other anthropogenic and natural sources (industrial point sources, leisure boats, atmospheric deposition and riverine input) in the Baltic Sea Area in 2018, the input of metals and PAHs from scrubbers is substantial (e.g. phenanthrene (8.9%) and vanadium (18%, Swedish EEZ)) [RQ1c]. Since 2018, the scrubber fleet in the Baltic Sea have grown from less than 200 to almost 800 in 2022 and the larger volumes suggest that the contribution to environmental pressure and load from scrubber water discharge is even higher today. The input of metals and organic substances to the marine environment imply increased environmental concentrations and thus a change in state [RQ2]. When the state change was assessed in European ports, based on 2018 operational data, the change in PEC was mostly driven by antifouling (for copper and zinc) and open loop scrubber water discharge (for the remaining seven metals, the US-EPA 16 PAHs and ten alkylated PAHs) [RQ2]. Consequently, when the PECs were compared to PNEC values for the specific substances, the impact, as added cumulative risk, from antifouling and scrubbers were significant, suggesting that restrictions of these activities could substantially reduce the pressure on the marine environment [RQ2]. The addition of more substances of concern, by extended chemical analysis and the derivation of effect concentrations from QSAR models, show that alkylated PAHs are relevant, both with respect to high concentrations and relatively high toxicity. When alkylated PAHs are included in the cumulative environmental risk assessment, open loop scrubber water discharge alone causes unacceptable environmental risk in Port of Copenhagen and Port of Gdynia, estimated from 2018 ship-operations, i.e. fewer scrubbers than today [RQ2]. From the economic perspective, there is a strong economic incentive to invest in scrubbers, where a majority of the global fleet having reached break-even by the end of 2022 with a surplus of 4.7 billion $€_{2019}$ [RQ3]. Given the volatile fuel price market of the recent years, the surplus from investing in a scrubber instead of shifting to low sulphur fuels has likely increased [RQ3]. Between 2014-2022, the average cumulative societal damage cost, for marine ecotoxicity from the discharge of nine metals and 10 PAHs with open loop scrubber water, exceeds 680 million $€_{2019}$ in the Baltic Sea area [RQ3].

The use of scrubbers is economically motivated by the continued use of cheaper HFO, i.e. a continued use of fossil fuels, enforcing the carbon dependence and unsustainable patterns of development. Ocean industrialization and the growth of ocean claims must not come at the expense of (marine) environmental damage. Mapping the specific and cumulative impact from different activities and sectors will become even more important to direct action and propose equitable measures to fulfil the commitments of "conserving and sustainably use the oceans, seas and marine resources for sustainable development" as stated at the United Nations Conference to Support the Implementation of Sustainable Development Goal 14 (UNOC 2022). The ocean is essential, scrubbers are not.

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