THESIS FOR THE DEGREE OF LICENTIATE OF PHILOSOPHY

Evaluating environmental policy instruments for ships

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Gothenburg, Sweden 2025

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Printed by Chalmers Digitaltryck

Gothenburg, Sweden 2025

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Abstract

Shipping is crucial for international trade, but has significant environmental impacts at local, regional, and global levels. Ship operations contribute to climate change, air pollution, and marine ecotoxicity through emissions of greenhouse gases, sulfur oxides, nitrogen oxides, particulate matter, and contaminants from sources like antifouling paints. Although the International Maritime Organization (IMO) aim to mitigate environmental impacts from shipping, the sector's environmental policies often lag behind land-based measures, highlighting the need for stronger policies and comprehensive policy evaluations.

This thesis investigates the effectiveness of environmental policy instruments in reducing emissions from shipping, with a focus on three policy cases addressing different policy approaches: port and fairway fee discounts (**Paper I**), inclusion of shipping in the EU Emission Trading Scheme (EU ETS) (**Paper II**), and the IMO global sulfur cap (**Paper III**). By examining these policy cases, the thesis analyzes how these policies influence shipping companies´ decisions to invest in different abatement strategies. Additionally, this thesis evaluates how different national, regional, and global policy instruments, both individually and in combination, influence shipowners' decisions to invest in abatement strategies. It also examines the subsequent impact on emissions beyond those directly targeted by the policies.

The first case (**Paper I**), which addresses Swedish environmentally differentiated port and fairway fees, shows that the discounts provided to shipowners are insufficient by themselves to motivate significant investment in abatement strategies. The second case (**Paper II**) evaluates the inclusion of shipping in the EU ETS, indicating that while abatement strategies such as batteries, e-methanol, and liquid hydrogen show potential, their high costs, in combination with the current EU ETS price, hinder broad adoption. Bio-methanol can for some ships and scenarios be a cost-effective fuel choice, however biofuels are limited by the amount of sustainably sourced feedstock. Overall, the Swedish environmentally differentiated port and fairway fees (**Paper I**) and the EU ETS policy cases (**Paper II**) highlight that while environmental policies are relevant, their current designs often fail to provide sufficient financial incentives for significant technological investment and emission reductions. The third case (**Paper III**) focuses on the IMO global sulfur cap, which successfully has reduced sulfur oxides emissions to the atmosphere but instead lead to a widespread use of scrubbers, generating new environmental concerns from scrubber water discharges. The results also highlight that combining the effects of the Energy Efficiency Design Index, a potential global scrubber ban, and an expanded EU ETS (including 100% of incoming and outgoing traffic) have synergistic effects that will reduce abatement costs for shipowners and thereby stimulate emission reductions or directly reduce emissions. However, abatement costs per amount avoided CO_2-emissions for retrofitting to use e-methanol are still more than four times higher than the current EU ETS-price. Furthermore, the results show that including smaller ships in EU ETS is important for reducing direct GHG emissions, further highlighting the need for a broader regulatory scope.

Keywords: abatement strategies, maritime economics, policy instruments, EU ETS, alternative renewable fuels, Scrubbers, global sulfur cap, port and fairway fees

List of publications

Paper I

Parsmo, R., E. Ytreberg, M. Verdaasdonk, and E. Fridell. 2024. Environmental discounts for Swedish ports and fairways: A shipowner perspective. Marine Policy **159**:105950. https://doi.org/10.1016/j.marpol.2023.105950

Rasmus was responsible for: Data curation, Formal analysis, Investigation, Writing – original draft, Visualization, Methodology.

Paper II

Parsmo, R., J. Hansson, S. Brynolf, E. Fridell, and E. Ytreberg. Shipping, emissions and EU emissions trading system: Evaluation of abatement strategies in different scenarios. in manuscript.

Rasmus was responsible for: Data curation, Formal analysis, Investigation, Writing – original draft, Visualization, Methodology.

Paper III

Lunde Hermansson, A., I.-M. Hassellöv, T. Grönholm, J.-P. Jalkanen, E. Fridell, R. Parsmo, J. Hassellöv, and E. Ytreberg. 2024. Strong economic incentives of ship scrubbers promoting pollution. Nature Sustainability. https://doi.org/10.1038/s41893-024-01347-1

Rasmus was contributing by writing–review and editing, resources, data curation.

List of additional publications and reports

A. Flodén, J., Zetterberg, L., Christodoulou, A., Parsmo, R., Fridell, E., Hansson, J., Rootzén, J., & Woxenius, J. (2024). Shipping in the EU emissions trading system: implications for mitigation, costs and modal split. Climate Policy, 1-19. https://doi.org/https://doi.org/10.1080/14693062.2024.2309167

B. Fridell, E., Parsmo, R., Jalkanen, J.-P., Majamäki, E., & Borken, J. (2022). Deliverable 1.4, "Description of scenarios to be used in EMERGE". Evaluation, control and Mitigation of the EnviRonmental impacts of shippinG Emissions. https://emerge-h2020.eu/results/

C. Hansson, J., Zetterberg, L., Rootzén, J., Parsmo, R., Fridell, E., Flodén, J., Woxenius, J., Raza, Z., Christodoulou, A., & Dalaklis, D. (2022). Impact of including maritime transport in the EU ETS. https://lighthouse.nu/images/pdf/Nyheter/C719_ETS_final_report.pdf

D. Jivén, K., Parsmo, R., Fridell, E., Hansson, J., Lundström, H., Wimby, P., Burgren, J., Koosup Yum, K., & Stenersen, D. (2023). Concept design and environmental analysis of a fuel cell RoPax vessel-Report in the HOPE (Hydrogen fuel cells solutions in shipping in relation to other low carbon options) project. https://www.diva-portal.org/smash/get/diva2:1796056/FULLTEXT01.pdf

E. Styhre, L., Jivén, K., Svedberg, S., Hansson, J., Priestley, M., Storm, B., Särnbratt, M., Grahn, D., Malmgren, E., Hjort, A., & Parsmo, R. (2024). Role of Port Authorities in green energy supply for transports chains: The way towards a green bunkering and charging strategy for ports and emission free inland waterways connecting a seaport with the hinterland. https://www.diva-portal.org/smash/get/diva2:1916436/FULLTEXT01.pdf

F. Ytreberg, E., Hansson, K., Hermansson, A. L., Parsmo, R., Lagerström, M., Jalkanen, J.-P., & Hassellöv, I.-M. (2022). Metal and PAH loads from ships and boats, relative other sources, in the Baltic Sea. Marine Pollution Bulletin, 182, 113904. https://doi.org/https://doi.org/10.1016/j.marpolbul.2022.113904

G. Åström, S., Parsmo, R., Merelli, L., Hult, C., & Mawdsley, I. (2022). Styrmedel för minskade NOX-utsläpp från vägtrafik, inrikes sjöfart och fiskefartyg: Utsläppseffekter, kostnader och nyttor år 2030. https://www.diva-portal.org/smash/get/diva2:1639377/FULLTEXT03.pdf

Acknowledgment

I have recently started listening to a song called “Gratitude”, and this is no coincidence. Being a PhD student is not always an easy journey, especially when the person pursuing the doctorate easily gets distracted, have an active ego, and tends to learn a little slowly (not a great combination). Pursuing a PhD is a humbling process that has taught me a lot not only about the field but also about myself. Looking back so far, I am grateful for the journey. I am also incredibly thankful to everyone who has helped me along the way and would like to say a few words about this below.

Firstly, I would like to thank all the stakeholders, both private and public, who have taught me so much about shipping and various technologies along the way. I would especially like to thank the Port of Gothenburg and the Port of Stockholm. Understanding how shipping works and receiving input from the “real world” has been crucial to my understanding of the field.

I would also like to thank all my current and former colleagues at IVL who have supported me both mentally and practically. For the very same reason, I also want to thank all my Chalmers coworkers, especially all coworkers at MES. There are so many of them with vast knowledge in numerous areas. A special thanks to the transport group, Julia, and my former IVL colleague Stefan, who have provided valuable input throughout the journey.

It's so easy to fall into a downward spiral of self-criticism, and in those moments, it’s important to have people close by. There are many who have offered supportive words and listened so far, especially at my two workplaces. If I have to mention someone, I want to mention Maja. I don't understand how you have the patience to listen to me so much! Many others have also contributed mentally along the journey, old friends, people from dance, brass band people, and my family, thank you all for that!

Finally, I would like to extend my deepest gratitude to my fantastic supervisors Selma Brynolf, Erik Fridell, and Erik Ytreberg. You have been amazing in so many ways and have supported me countless times, with everything from conceptual questions and writing to administration. Selma has often understood my thinking when I couldn't, Erik F has stayed calm during moments of panic, and Erik Y has believed in me even when I doubted myself. Tusen tack!

Rasmus Parsmo

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Abbreviations, acronyms, and description of concepts used in this thesis

Table 1. Abbreviations and acronyms

Abbreviation	
bio-MeOH	Bio Methanol
CH_4	Methane
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CSI	Clean Shipping Index
CO2	Carbon Dioxide
ECA	Emission Control Area
EEA	European Environment Agency
e-LH_2	Electrolytic Liquid Hydrogen
e-MeOH	Electrolytic Methanol
EEDI	Energy Efficiency Design Index
ESI	Environmental Ship Index
EP	European Parliament
ETS	Emission Trading System
EU	European Union
GES	Good Environmental Status
GHG	Greenhouse Gas
GT	Gross Tonnage
HFO	Heavy Fuel Oil
HVO	Hydrotreated Vegetable Oil (Biodiesel)
IMO	International Maritime Organization
LBG	Liquefied Biogas
LCA	Life Cycle Analysis
LNG	Liquefied Natural Gas
LRTAP	Long-Range Transboundary Air Pollution
MDO	Marine Distillate Oil
MGO	Marine Gas Oil
MRV	Monitoring, Reporting, and Verification
MSFD	Marine Strategy Framework Directive
N_2O	Nitrous Oxide
NO_X	Nitrogen Oxides
OPS	Onshore Power Supply
PEMFC	Proton Exchange Membrane Fuel Cell
PM	Particulate Matter
PAHs	Polycyclic Aromatic Hydrocarbons
RME	Rapeseed Methyl Ester
SO_X	Sulfur Oxides
SMA	Swedish Maritime Administration
TBT	Tributyltin
VLSFO	Very Low Sulfur Fuel Oil

1. Introduction

Shipping is important for transporting large amounts of goods over long distances and accounts for over 70% of global freight transport demand (tonne-km) according to International Transport Forum (2023). While shipping is essential for global trade, its extensive operations have environmental consequences globally (Faber et al., 2020, Johansson et al., 2017), regionally (Ytreberg et al., 2021, Jalkanen et al., 2016), and locally (Ducruet et al., 2024, Lunde Hermansson et al., 2023). Ships impact the environment in several ways, including contributing to global warming, causing adverse health effects, and leading to acidification, eutrophication, and ecotoxicity due to emissions to air and water (Ytreberg et al., 2021, Åström et al., 2018, Faber et al., 2020).

Most ships currently use fossil fuels as the primary source for operating their engines and boilers, relying on marine fuels such as marine gas oil (MGO), marine distillate oil (MDO), very low sulfur fuel oil (VLSFO), high sulfur fuel oil (HSFO), and liquefied natural gas (LNG) (EC, 2016, EC, 2017, IMO, 2022). In total, ships are estimated to emit 1,076 million tonne (Mtonnes) of carbon dioxide equivalents (CO_{2e}) in 2018, corresponding to approximately 2.89% of all anthropogenic emissions globally (Faber et al., 2020). Most of the CO_{2e} for the global fleet arises from carbon dioxide (CO_2), but there are also emissions of methane (CH_4) and nitrous oxide (N_2O) during combustion. Methane slip, in particular, can be a significant fraction of the CO_{2e} when LNG is used as fuel (Malmgren, 2023, Brynolf et al., 2014). In response to the ships' impact on global warming the International Maritime Organization (IMO) has set an objective to reach net-zero GHG emissions by the year 2050 (IMO, 2023).

During combustion, several air pollutants are formed, including sulfur oxides (SO_X), nitrogen oxides (NO_X), and particulate matter (PM). These emissions contribute to increased health risks, such as effects on the airways and lungs (Åström et al., 2019, Åström et al., 2018), to acidification (SO_X and NO_X) and to eutrophication (NO_X) on both land and sea (Jalkanen et al., 2014, Ytreberg et al., 2021). The absolute emissions from the shipping sector are significant. For instance, maritime transport accounted for 24% of NO_X emissions, 24% of SO_X emissions, and 9% of $PM_{2.5}$ emissions within the EU, relative to the total emissions from all other sectors in 2018 (EMSA, 2021). In response to emissions to air, the Convention on Long-Range Transboundary Air Pollution (CLRTAP) addresses and aims to mitigate the effects of air pollution that crosses national borders. However, this convention only includes domestic emissions (emissions from ships going between two national ports) from the maritime sector through the Gothenburg Protocol, a component of the CLRTAP, which sets emission limits for each participating country (EC, 2019). This focus is not optimal, as it excludes emissions from international shipping, which constitutes the largest share of maritime emissions. For example, “Swedish” reported international shipping NO_X emissions was about 93.9 ktonnes of NO_X, compared to about 5.4 ktonne from domestic shipping in 2021 (SCB, 2023).

Ships also release a wide variety of hazardous substances into the marine environment, including metals and organic compounds such as polycyclic aromatic hydrocarbons (PAHs) (Johansson et al., 2020, Jalkanen et al., 2021). These substances may significantly impact the marine ecosystem by affecting marine species at various life stages, leading, for instance, to malformations, diminished reproductive capacity, or increased mortality. This impact category is referred to as marine ecotoxicity (Huijbregts et al., 2016, Goedkoop et al., 2009). The risk of adverse environmental impact is influenced by several factors, including the fate of the chemical once released into the marine environment, its persistence, its toxicity to various organisms, ambient environmental concentrations, and the degree of dilution it undergoes (Huijbregts et al., 2017, Huijbregts et al., 2016, Fantke et al., 2017, Owsianiak et al., 2023, Dong et al., 2016). In some regions, the load of hazardous substances from shipping can be significant compared to other anthropogenic sources. This issue is particularly relevant in shallow water basins with low water exchange, such as the Baltic Sea, where these properties lead to higher concentrations of hazardous substances (Ytreberg et al., 2021, Ytreberg et al., 2022). Furthermore, recent studies have shown emissions of hazardous substances from shipping to pose an unacceptable environmental risk in areas with heavy maritime traffic, such as ports (Lunde Hermansson et al., 2023). To address these challenges, the EU Marine Strategy Framework Directive (MSFD) aims to implement an ecosystem-based approach to managing human activities in marine waters, ensuring the achievement and maintenance of Good Environmental Status (GES) by reducing harmful pressures and promoting sustainable use of marine resources (EC, 2008).

To achieve environmental goals such as reducing greenhouse gas (GHG) emissions, minimizing air pollution, and attaining GES in all European marine waters, policies are implemented at different levels. These policies cover a wide range of technologies and abatement strategies. However, for many types of pollution, policy development in the shipping sector often lags behind that of the land-based sector, as seen in the EU. International maritime emissions to air are primarily regulated by the IMO through MARPOL Annex VI, which establishes global standards for sulfur content in fuels and NOx emissions from ships (MEPC, 2021). The global sulfur cap for marine fuels is currently set at 0.5%, with a stricter limit of 0.1% in Sulfur Emission Control Areas, (SECAs). These limits can be met by either by using “low sulfur” fuels or by using a scrubber (Topali and Psaraftis, 2019), which is an exhaust abatement equipment that reduces the emissions of SO_X to the atmosphere to levels corresponding to the low-sulfur fuel. By comparison, the maximum allowed sulfur content in diesel for on-road vehicles in the EU has been 0.001% since 2009 (EP and EUCO, 2003). There are currently no global policy instruments directly regulating PM emissions from ships. In contrast, road vehicles in Europe are subject to stringent emissions standards for PM and other pollutants (EP and EUCO, 2003). The emissions of CO_2 from road vehicles in the EU are addressed in several ways for example directly through carbon taxes (Tax Foundation, 2024a), or indirectly through fuel taxes (Tax Foundation, 2024b). For shipping, there were no equivalent taxes, prices or fees on marine fuels, until the inclusion of shipping in the European Union Emissions Trading System (EU ETS) in 2024 (EP and EUCO, 2023a).

The shipping sector has traditionally not been viewed as a leader in environmental policy, as seen when compared to European land-based sectors. High CO_2 abatement costs, driven by low fossil fuel prices, long distances, and technical challenges at sea, also contribute to its reputation as a “hard-to-abate” sector (Malmgren et al., 2023). This perception is further influenced by the industry's international nature, which complicates the achievement of agreements, unlike the case for national or regional sectors (Gritsenko, 2017). On the contrary, reducing other types of emissions in the shipping sector could be more cost-effective than achieving additional reductions in land-based sectors. This is exemplified by initiatives such as the Norwegian NO_X Fund, which demonstrated cost-effective emission reductions in maritime operations (Parsmo et al., 2017). There are also other market based national or local policy instruments that can be combined with each other such as environmentally differentiated port and fairway fees (Lindé et al., 2019, Vierth and Johansson, 2020, Christodoulou et al., 2019, Sköld, 2019). Furthermore, the shipping sector's international governance through the IMO presents an opportunity to implement global environmental regulations in ways that may not be feasible for other sectors. However, the interaction between various maritime policies and their combined effects is not fully understood. There is a need to investigate further the interrelationships between various policy instruments, addressing different emissions and abatement strategies across diverse geographical scopes in order evaluate if the policies lead to real actions.

1.1. Research aim

The overall aim of this thesis is to evaluate how various national and international environmental policy instruments influence shipowners' decisions to invest in alternative abatement strategies. It also aims to determine whether these policies effectively reduce emissions and environmental impact. This thesis addresses the following research questions:

RQ1: *How does the current design of environmental policy instruments encourage shipping companies to invest in technical and operational measures to reduce environmental emissions?*

This is evaluated through three different policies:

- **Policy 1:** Evaluating the design and impact of Swedish environmentally differentiated port and fairway fees on shipping companies' investments in emission-reducing measures (**Paper I**).
- **Policy 2**: An early evaluation of the inclusion of shipping in the EU ETS. This case study assesses how the EU ETS, both alone and in combination with other policy instruments, encourages shipping companies to invest in cleaner fuels and energy carriers that reduce GHG emissions (**Paper II**).
- **Policy 3:** The rise of scrubbers and IMO global sulfur cap in 2020. This case evaluates the costs and benefits of scrubbers as a response to IMO's global sulfur cap, examining the economic implications for shipowners and the environmental impact on marine ecosystems (**Paper III).**

RQ2: *How do various national, regional, and global policy instruments, both individually and in combination, influence shipowners' decisions to invest in abatement strategies, and what is the subsequent impact on emissions other than those targeted by the policy?*

This is assessed through two extended analyses to evaluate the interactions and unintended consequences of policy combinations on shipowners' investment decisions. The first analysis focus on the European fleet under global and regional policies (based on work in **Paper II** and **Paper III**). The second analysis examines ships entering Swedish ports, incorporating national policies (based on work in **Paper I**) and updated Swedish emissions calculations based on port call data and route-specific emissions factors.

1.2. Scope and delimitations

The three papers included in this thesis relate to three different types of polices: a bonus system by the environmental discounts on port and fairway fees (**Paper I**), emissions trading permits though inclusion of shipping in EU ETS (**Paper II**), and emissions standards though IMO's global sulfur cap (indirectly **Paper III**).

This thesis focuses primarily on direct emissions from ships, excluding indirect emissions and those from the entire life cycle. However, a life cycle perspective is integrated in cases where it significantly influences the analysis, such as GHG emissions associated with fuel production (**Paper II**).

Key aspects of supply and demand are also omitted, such as the possible link between the actions of shipowners and the costs of abatement strategies. One example of such a non-inclusion is that the price of fossil fuels may depend on the demand in the shipping sector. Furthermore, the economic assessments are limited to the shipping sector. However, when examining different energy pathways, the impact on the entire energy system is also important.

2. Background

Evaluating environmental policy instruments in shipping requires interdisciplinary knowledge of several aspects, such as policies, environmental impacts, and shipping in general. The first section of this chapter provides a brief overview of some theories and concepts related to environmental policies, as well as describing different aspects of a policy's scope. It also outlines some policy instruments currently applied in shipping and how they are implemented. The final section elaborates on abatement strategies that could be used to reduce environmental impacts.

2.1. Theory of environmental policy instruments for ships

There are several ways to categorize environmental policies within different taxonomies, such as the classical division into command-and-control (direct) and market-based (MB) policies (Sterner and Coria, 2012, Christodoulou et al., 2019). Command-and-control regulations involves a regulatory body that enforces restrictions, while market-based approaches use economic incentives to achieve environmental goals. However, Sterner and Coria (2012) argue that it is often beneficial to break down the taxonomy of policy instruments into smaller pieces, to better understand the mechanisms of each policy. One such mechanism involves whether the policy is regulating quantity/output or price. Quantity-based policy instruments regulate the amount of emissions allowed (such as an emissions trading scheme), while price-based instruments affect the cost of emissions (e.g., a tax or subsidy). Table 2 categorize various implemented policy instruments for shipping relevant for this thesis.

Table 2. Example of categorization of relevant policy instruments for shipping

Example of taxonomy of policy instruments	Example of categorization	Examples	Source
Global emission standards	Direct	NO_X (Tier) and IMO global sulfur cap	(MEPC, 2024, MEPC, 2014, MEPC, 2021)
Bans	Direct	Tributyltin (TBT) in antifouling paint	(IMO, 2001)
Tradable emission payments	Market-based quantity	Inclusion of ships in EU ETS	(EP and EUCO, 2023a)
Tradable quotas	Market-based quantity	EU Fuel Maritime	(EP and EUCO, 2023d)
Refundable emissions payments	Market-based price	Norwegian NO_X fund	(Parsmo et al., 2017)
Detailed regulation	Direct	Obligations for some ship types to use onshore power supply in EU	(EP and EUCO, 2023d, EP and EUCO, 2023c)
Subsidies	Market-based price	Climate Leap	(Swedish EPA, 2024b)
Bonus systems	Market-based price	Environmental differentiated discount in Swedish port and fairways	(Parsmo et al., 2024)

Two other central concepts in this domain are “efficiency” and “effectiveness” of policy instruments (Vaz—EEA et al., 2001, EEA, 2001, Sterner and Coria, 2012). If a policy instrument is effective, it reduces the targeted environmental impact/s or emissions (quantity/output). If it is efficient, it achieves the targeted reduction at a low cost (or price). Efficiency is sometimes analyzed by investigating the abatement cost (Einride, 2024), which is another important concept. Generally, abatement cost is defined as the cost of an abatement strategy per unit of environmental improvement achieved, where environmental improvement can be determined by for example emission reductions or reduced environmental impacts.

However, evaluating policy instruments solely by observing improved environmental outcomes (effectiveness) and achieving the lowest possible abatement cost (efficiency) in the short term is not always ideal, especially when assessing innovation. Costs can change with increased production (Jacobsson and Bergek, 2011), and emissions may decrease as technological systems change over time. In such cases, it may be relevant to also consider these changes as part of a technological innovation system (TIS).

Originally, the TIS framework was introduced by researchers such as Geels (2005), who emphasized its utility in understanding the socio-technical transitions required for innovation. In the shipping sector, Bach et al. (2020) and Bach et al. (2021) expanded on this framework, highlighting that policies or niche environments are critical for fostering the growth of alternative solutions. For example, these studies identify regulatory support as a key enabler for low-carbon technologies. Similarly, Jivén et al. (2023) and Parsmo et al. (2024) explore how cost reductions, physical constraints, and institutional barriers influence the adoption of abatement strategies in shipping. The TIS perspective aligns well with the polycentric governance approach by Ostrom (2017) as an effective way to achieve climate targets. Ostrom (2017) defines polycentric governance as a structure in which "multiple governing bodies operate at different levels and scales", rather than relying only on a single central authority. Key aspects that make polycentric approaches effective include creating opportunities for experimentation and policy learning (Gritsenko, 2017).

Gritsenko (2017) focuses on discussion on GHG emissions, however it is worth noting that similar policy learning experiences have occurred in the shipping sector for other pollutants, such as organotin compounds such as tributyl tin (TBT) in antifouling paints. Research on the effects of TBT on oysters revealed impacts on growth and reproduction, leading to local restrictions on organotin compounds for small vessels (less than 25 m) in France as early as 1982 (Alzieu, 1991), with regional restrictions in the EU on pleasure boats since 1989 (Ytreberg et al., 2016). However, it was not until 2008 that organotin compounds were banned entirely in antifouling paints on ships through the International Convention on the Control of Harmful Anti-fouling Systems on Ships (AFS Convention) of the IMO. Following the ban on organotin compounds, most ships transitioned to copper-based antifouling paints (Amara et al., 2018).

The scope of a policy is essential in determining how to make it effective and efficient over time. Global regulations through the IMO have historically been of the command-and-control type (MEPC, 2014, IMO, 2001, Gritsenko, 2017), which may be very effective when enforced. However, the issue with this type of regulation is not only that it is challenging to reach an agreement and time-consuming to implement. Command-and-control regulations are also difficult to design in ways that first allows for learning on a smaller scale (Sterner and Coria, 2012, Gritsenko, 2017). As Gritsenko (2017) points out in the context of reducing GHG emissions:

"Instead, there is a need to introduce changes in ship design and operations, which could be more difficult to introduce in a form of command-and-control regulation, further highlighting the value of polycentric governance."

It is important to explore other methods to advance environmental performance for ships. This can include implementing regulations at different geographical scales, over various time frames, or for specific ship segments.

2.1.1. Geographical aspect of a policy

Environmental policy instruments for ships can vary significantly in their geographical scope, ranging from local, regional to global levels. However, these boundaries can sometimes be vague. For example, "national" can have different meanings depending on the regulation. National emissions reported to conventions like the LRTAP are based on the amount of fuel sold domestically (Windmark et al., 2017, Windmark, 2019, Swedish EPA, 2024a), whereas policies such as Sweden's environmental fairway fees apply to ships visiting national ports (SMA, 2017). Similarly, regional policies might target specific water zones (e.g., Emission Control Areas)(MEPC, 2021, MEPC, 2024), or ports visited by ships (e.g., the EU ETS). These distinctions demonstrate how the scope of policies varies depending on their context, as further explained in the following subsections.

Global regulations

SO_X and NO_X emissions from ships are regulated globally under the International Convention for the Prevention of Pollution from Ships (MARPOL) Annex VI (MEPC, 2024, MEPC, 2014, MEPC, 2021), through NO_X tier regulations and sulfur caps, with specific limits in designated Sulfur Emission Control Areas (SECAs) and Nitrogen Oxide Emission Control Areas (NECAs). The MARPOL convention is implemented by the IMO and is adopted by member countries.

More precisely, the 2020 sulfur cap (IMO, 2019), mandates a maximum sulfur content in marine fuels of 0.5% globally, with compliance monitored by port state control, through the use of compliant fuels or scrubbers (Topali and Psaraftis, 2019), as shown in Figure 1. The sulfur limits are even stricter in SECAs, with a limit of 0.1% in the North Sea, the Baltic Sea, the English Channel, and off the coasts of North America and the U.S. Caribbean, and from May 1, 2025, in the Mediterranean Sea.

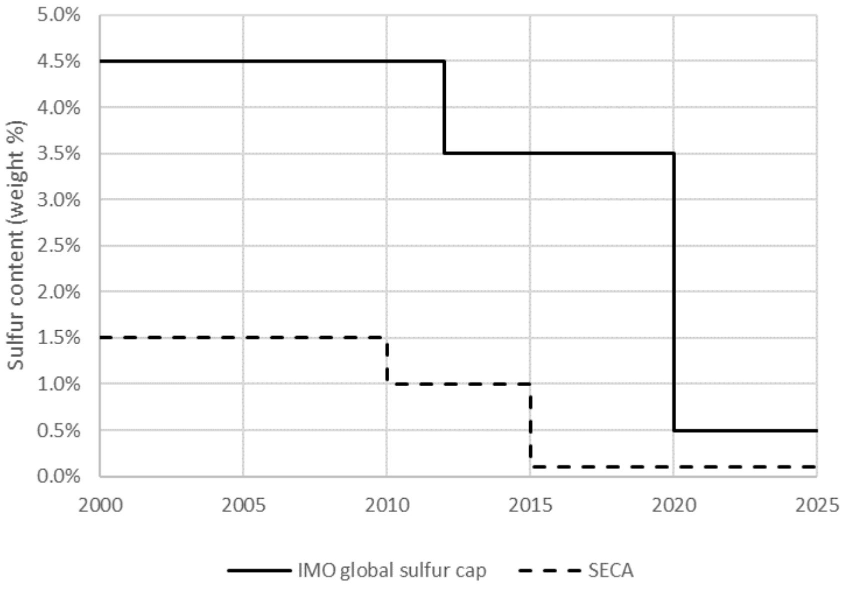

Figure 1. Sulfur limits of marine fuels for the period 2000-2025.

The NO_X tier regulations (Tiers I, II, III), on the other hand, set limits on NO_X emissions from ships' engines, divided into different tiers based on the year the ship was constructed[1] and the geographical area of operation, see Figure 2. Tier I applies globally to ships built after 2000, with lower limits under Tier II for ships built after 2011. The strictest limit, Tier III, only applies in NECAs, requiring further reductions in NO_X emissions from ships constructed after 2016 (North American and US Caribbean) or 2021 (North Sea, Baltic Sea and English Channel).

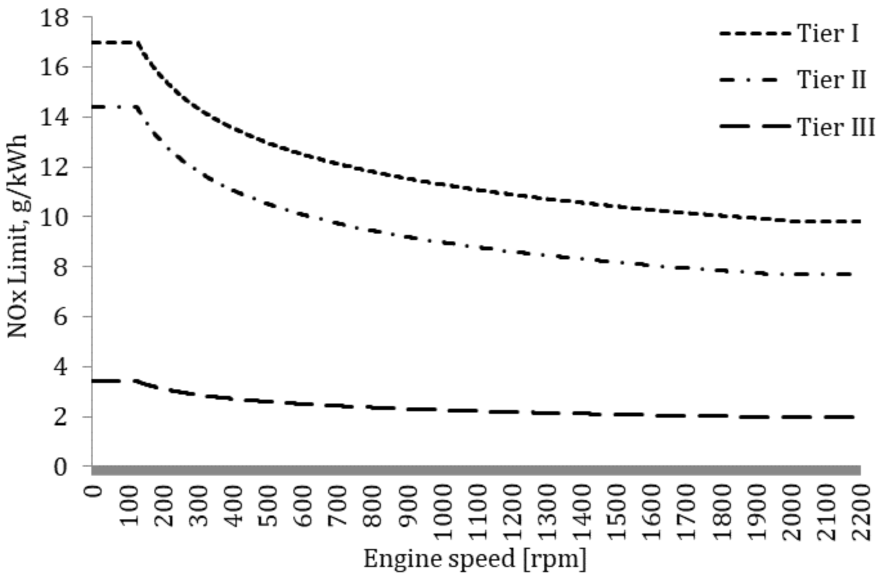

Figure 2. NO_X emission standards for marine engines in international shipping, varying by engine speed (rpm) and the ship's construction year.

European regulation: Inclusion in EUs Emission trading scheme (EU ETS)

Lately, several EU regulations have been adopted to reduce maritime GHG emissions (EP and EUCO, 2023a, EP and EUCO, 2023d, EP and EUCO, 2023c). Shipping was added to the EU ETS in January 2024, as part of the European Green Deal (Christodoulou and Cullinane, 2024, Christodoulou et al., 2021). The EU ETS is an emissions trading scheme in which actors from various sectors must either purchase or are allocated allowances to be allowed to emit CO_2; the historic prices of allowances are illustrated in Figure 3. Established in 2005, the EU ETS currently includes direct CO_2 emissions and will also cover N_2O and CH_4 from 2026 (EP and EUCO, 2023a). For shipping, the scope includes all intra-EU shipping traffic, port emissions, and 50% of emissions from inbound and outbound traffic. Additionally, the EU ETS covers only larger ships, specifically those with a gross tonnage of 5,000 and above.

[1] year of keel laying. However, if ship engines are replaced during the ships lifetime the ships Tier requirement will be based on the date of installing the new engine.

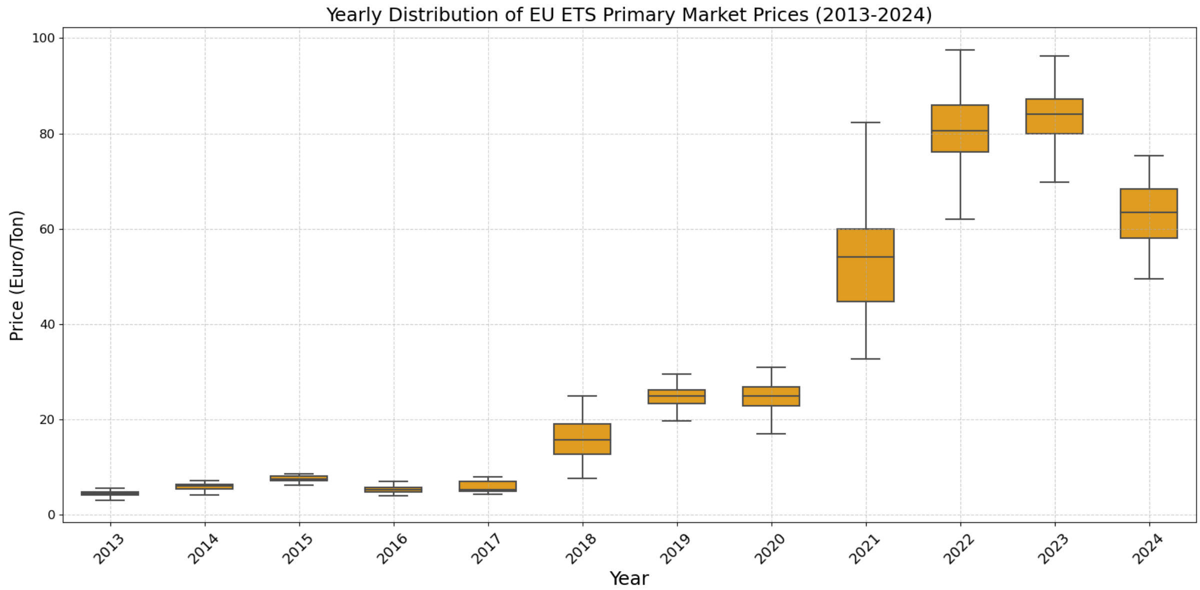

Figure 3. Boxplot illustrating daily historic EU ETS allowance price (ICAP, 2024). For 2024 only data for January-September are presented.

Since 2018, data on fuel consumption and CO_2 emissions have been collected for all ships over 5,000 gross tonnes (GT) that enter an European Economic Area (EEA) port (EP and EUCO, 2024, EP and EUCO, 2015, EC, 2017, EC, 2016, EC, 2023a, EC, 2023b). This data collection was essential for incorporating shipping into the EU ETS. The total annual CO_2 emissions for the period 2018–2023 are shown in Figure 4 (THETIS-MRV, 2023), which illustrates that emissions vary significantly across different ship segments, with container ships being the largest contributors.

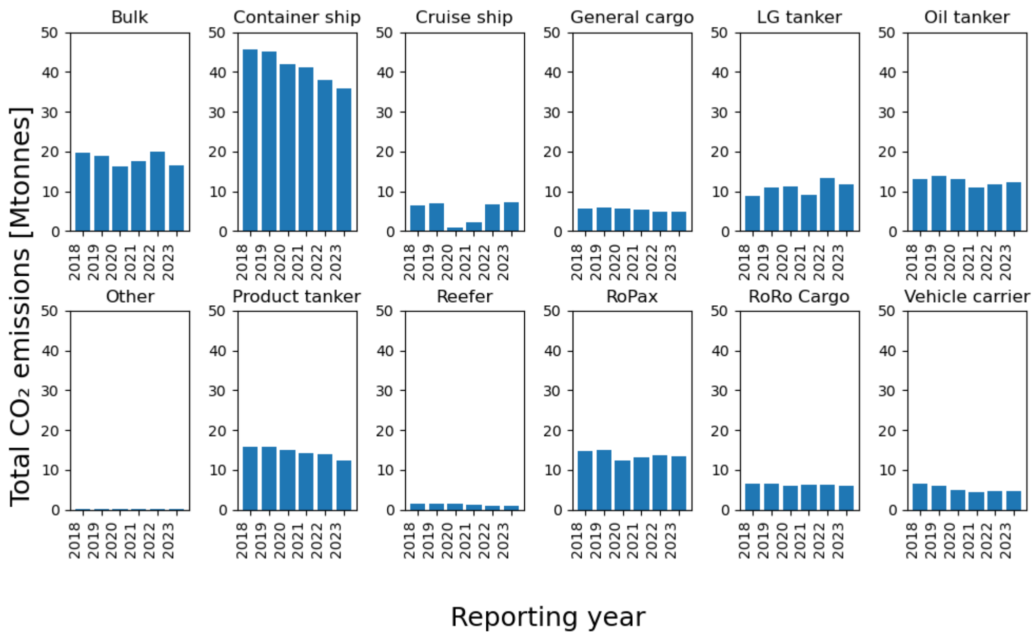

Figure 4. Total CO_2 emissions for the EU fleet (2018-2023) across 12 different ship segments, based on processed data from (THETIS-MRV, 2023). The figure illustrates the emission scope in Europe. The ship category 'Other' is currently not included in the EU ETS scope. However, some ships in this category have reported to MRV and are therefore included in this figure.

National and local maritime regulations in Sweden

In Sweden, environmentally differentiated port and fairway fees are in place (Sköld, 2019). Ships transporting goods or passengers to or from Sweden must pay a fee to the Swedish Maritime Administration (SMA). This fee was divided into three categories in 2020: port call, readiness, and pilot. In 2020 environmental discounts were available only for the port call fee, based on the ship's environmental performance as assessed by the Clean Shipping Index (CSI) labelling system (SMA, 2017, SMA, 2016). These types of environmentally differentiated fairway fees have been in place since 1998, providing rebates based on ships environmental performance (Lindé et al., 2019, Vierth and Johansson, 2020). Furthermore, 19 of 32 ports in Sweden offer environmentally differentiated port fees, with discounts related to the ship's size and environmental performance indices (Environmental Ship Index (ESI) and/or CSI). CSI and ESI provide environmental performance-based valuation based on different scoring. In 2020 CSI scored ships in five categories (CO_2, NO_X, water and waste, SO_X/PM, and chemicals) (CSI, 2020). The ESI only includes emissions to air (NO_X, SOx, and CO_2) and offers additional points for onshore power supply installations (ESI, 2020).

2.1.2. Ship segments

In different policies, ships are often divided into segments based on their functionality or characteristics. Typically, in the industry or literature, these segments are categorized by ship type and size, and this segmentation often varies across studies and databases (S&P Global, 2023, THETIS-MRV, 2023, Faber et al., 2020, DNV GL, 2020, Jalkanen et al., 2009, Windmark et al., 2017). In this thesis, ships are divided into 12 segments: bulk, container, general cargo, oil tanker, product tanker, LNG tanker, Roll-on/Roll-of (RoRo), vehicle carrier, Roll-on/Roll-of passenger (RoPax), reefer, other and cruise.

This segmentation can vary depending on the purpose of a regulation, the segments competitive context, or practical/technical considerations. Some regulations include only certain ship types or apply different pricing models based on ship function. For example, port and fairway discounts apply only to goods and passenger ships, and for port fee discounts, different ship types pay different rates (Parsmo et al., 2024). Size is also a determining factor, such as the exclusion of ships smaller than 5 000 GT in the EU ETS. Some policy instruments are instead implemented through government like for example public procurement (Bach et al., 2020), or apply directly to ships owned and operated by the public sector, like SMAs shipping fleet (SMA, 2023a). This allows certain segments, such as road ferries, icebreakers, or pilot vessels in Sweden, to be regulated directly.

2.2. Abatement strategies

Abatement strategies are approaches or sets of measures designed to reduce or eliminate harmful emissions, or environmental impacts. In the context of shipping, an abatement strategy can target the reduction of air pollution, GHGs and water pollution, as illustrated in the following three sections.

2.2.1. Abatement strategies to reduce emissions of air pollutants

Shipping contributes significantly to air pollution (Jalkanen et al., 2016), including NO_X emissions, which have harmful environmental and health impacts (Åström et al., 2018). NO_X emissions are generated during combustion mainly when nitrogen in the air reacts with oxygen at high temperatures. To address this, selective catalytic reduction (SCR) systems have been installed on many ships and are one of the most common methods for reducing NO_X emissions from marine diesel engines (S&P Global, 2023). SCR allows the ship to achieve compliance with Tier III NO_X limits by chemically converting nitrogen oxides into nitrogen and water using a catalyst and urea injection. Its effectiveness in reducing NO_X emissions and its retrofit potential have made it a common abatement strategy (S&P Global, 2023), particularly for new-built ships operating in NECAs.

Fossil fuels, particularly HFO, contain high concentrations of sulfur, which reacts with oxygen during combustion, producing SO_X emissions (Lunde Hermansson et al., 2021). These emissions can be reduced by using low- or zero-sulfur fuels or by installing exhaust gas cleaning systems (EGCS), commonly known as scrubber. Low-sulfur fuel refers to any fuel with a sulfur content of 0.5% or less. Scrubbers are commonly wet scrubbers, which create a new waste stream of contaminated and acidic wash water (Lunde Hermansson et al., 2021, Lunde Hermansson et al., 2024, Jalkanen et al., 2021). With a wet scrubber, the sulfur oxides are removed by spraying water through exhaust gases to capture SO_X and previous studies indicate particle mass reduction of on average 40% over exhaust gas scrubbers (Winnes et al., 2020, Fridell et al., 2020, Karjalainen et al., 2022). Scrubbers can reduce SO_2 emissions by over 99% (Winnes et al., 2020) (most of the sulfur oxides emitted are in the form of SO_2). However, approximately 1-8% of the sulfur emissions in the exhaust gas are in the form of sulfur trioxide (SO_3). Scrubbers appear to be less efficient at removing these emissions, achieving a removal rate of 61-78%, the reasons for this lower reduction rate are not fully understood (Winnes et al., 2020).

Another way to prevent air pollution (such as NO_X, SO_X, and PM) is by using onshore power supply (OPS), also known as cold ironing (Doves, 2006, Vaishnav et al., 2016, Parsmo et al., 2024). With OPS, ships connect to the electricity grid at berth, and the electric power either covers all or parts of the ship's hoteling power demand, eliminating the need to run auxiliary engines. By replacing fossil fuel combustion with grid electricity, OPS reduces the emissions of local air pollutants. Life cycle emissions of GHGs are also reduced if electricity from renewable energy sources is used. OPS is adopted in ports to reduce emissions during berthing but also, in some cases, to reduce fuel costs (Doves, 2006, Vaishnav et al., 2016, Parsmo et al., 2024).

2.2.2. Abatement strategies to reduce greenhouse gases

Reduction of GHG emissions can potentially be achieved by switching to alternative fuels such as biofuels, hydrogen, electrofuels or using electricity directly which is further described below.

Biofuels are renewable fuels derived from organic materials, such as plant biomass, agricultural residues, or waste, that can be used as alternatives to fossil fuels for reducing GHG emissions. The ability of biofuels to reduce GHG emissions depends primarily on how the feedstock is produced. In this context, the environmental impact of different biofuels is debated (Broch et al., 2013, Cowie et al., 2017, Berndes et al., 2016, Sparovek et al., 2016, Searchinger et al., 2018, Reid et al., 2020). In the EU, sustainability criteria are addressed through the Renewable Energy Directive (EP and EUCO, 2018, EP and EUCO, 2023b). Several options exist for using biofuels in shipping, including hydrotreated vegetable oil (HVO), rapeseed methyl ester (RME), liquefied biogas (LBG), and bio-methanol (bio-MeOH) (Kanchiralla et al., 2023, Mukherjee et al., 2023, Bach et al., 2021). Biodiesel, such as HVO or RME, can be used in diesel engines either exclusively or blended with fossil diesel. Similarly, LBG is compatible with ships using LNG dual-fuel engines, and Bio-MeOH is suitable for ships equipped with methanol dual-fuel engines.

Hydrogen can be used in combination with fuel cells paired with an electric engine or in an internal combustion engine to power a ship (Kanchiralla, 2023, Kanchiralla et al., 2023, Jivén et al., 2023). Hydrogen is produced using electrolysis, where electricity splits water into hydrogen, oxygen, and access heat. The environmental performance of hydrogen therefore profoundly depends on how the electricity is generated. The carbon intensity of the electricity production is particularly relevant since the overall efficiency of converting electricity to hydrogen (electrolyzers) and back to electricity (onboard with fuel cells) is low. For example, the grid-to-propeller “efficiency” for a RoPax vessel operating between Gothenburg and Kiel was estimated to be only about 33% (Kanchiralla et al., 2022).

To use hydrogen onboard ships, the gas must be converted into either liquid (LH_2) or compressed (CH_2) form and stored in specialized tanks (Kanchiralla et al., 2022). Although hydrogen has a higher lower heating value than other fuels such as LNG, MeOH, or MGO, it has a lower system energy density both in liquid and compressed form, as it needs special tanks and therefore requires more space onboard (Grahn et al., 2022, Kanchiralla et al., 2022). Fuel cells are a relatively new technology in the shipping sector; for instance, the world first hydrogen-powered commercial passenger ship began operating in Norway in March 2023 (Kullenberg Rothvall, 2024).

Electrofuels, also known as e-fuels, are synthetic fuels produced by combining hydrogen (generated via renewable electricity-powered electrolysis) with carbon dioxide (resulting in e.g., e-methanol or e-MeOH) or nitrogen (resulting in e.g., e-ammonia or e-NH_3) (Styhre et al., 2024, Grahn et al., 2022, Kanchiralla, 2023). As with hydrogen, the process of converting electricity to e-fuel is inefficient, not only due to the hydrogen production but also because obtaining carbon or nitrogen also requires energy inputs as well as the synthesis process. For example, producing renewable e-MeOH requires obtaining CO_2 from a non-fossil source, such as through direct air capture or capturing it from a bioproduction facility (point source capture). Furthermore, if the e-fuel or the biofuel, such as e-MeOH or bio-MeOH, is used in an internal combustion engine, it will still result in emission of air pollutants, such as NO_X and particles. Methanol (including e-MeOH and Bio-MeOH) can be used in ships equipped with special dual-fuel engines, which some vessels currently have installed (S&P Global, 2023). These engines require the injection of pilot fuel, such as MGO.

Battery-electric ships use large-capacity batteries to store and supply electricity for propulsion. The grid-to-propeller efficiency of battery-electric ships is about 84-90 % (Brynolf et al., 2023, Stolz et al., 2022), which is considerably higher than, for example, using electrofuels, also accounting for engine losses, onboard power electronics, and transfer losses. The challenges with deploying battery systems onboard ships relate more to considerations of space, weight, and energy storage capacity, which influence the ship's design and operational profile (technical feasibility), heavily affecting the costs (economic feasibility) (Kanchiralla, 2023, Kanchiralla et al., 2023, WSDOT, 2020, Stena Rederi Technical Division, 2018, Jivén et al., 2020, MAN, 2019, Kistner et al., 2024, Link et al., 2024, Kersey et al., 2022). Furthermore, charging batteries in ports poses challenges, such as the need for substantial power capacity, which can also impact electricity prices if power-based tariffs are applied (Parsmo et al., 2024).

2.2.3. Abatement strategies to reduce emissions to water

Hazardous substances released into the sea originate from various ship activities, such as release of biocides from antifouling paints (Lagerström et al., 2022), and the discharge of scrubber water as previously mentioned (Lunde Hermansson et al., 2021).

Traditional biocidal antifouling paints (such as copper-based coatings) are used on a ship's hull to prevent the settlement and growth of aquatic organisms. An effective antifouling paint reduces friction, drag and fuel consumption while also minimizing the spread of invasive species (Amara et al., 2018). However, the active substances in the paint (like copper oxide) leach (as copper) into the water over time, affecting marine organisms (Lagerström et al., 2020). One way of abating this is by switching to biocide-free coatings, like certain types of silicone paints. Silicone-based foul-release coatings have recently gained more attention due to their ability to reduce water resistance without relying on harmful biocides (Johansson et al., 2020, Oliveira et al., 2022). Silicone coatings may also reduce fuel consumption and GHG emissions by improving vessel hydrodynamics, though they usually come with higher initial application costs and specific maintenance requirements (Lagerström et al., 2022).

As mentioned in section 2.2.1, scrubbers produce wastewater that contains various hazardous substances, including metals and PAHs (Lunde Hermansson et al., 2023, Ytreberg et al., 2022, Lunde Hermansson et al., 2021). This scrubber water can either be discharged directly into the sea (open-loop system) or partially treated by storing it onboard for later disposal at onshore facilities (closed-loop system) (Lunde Hermansson et al., 2024). However, despite being referred to as "closed-loop," some discharge water is still released into the ocean, although to a lesser extent, 0.45 m^3/MWh for closed-loop systems compared to 90 m^3/MWh for open-loop systems (Ytreberg et al., 2021). This closed-loop water contains higher concentrations of some pollutants than open-loop water, even though the overall load of pollutants from ships operating with closed loop systems is typically lower than for open-loop systems (Lunde Hermansson et al., 2021). Hybrid scrubber systems also exist, combining open- and closed-loop functionalities and allowing temporary storage of scrubber water (Ytreberg et al., 2022). This storage is used in areas where scrubber discharges are prohibited, with the waste released when the ship reaches zones where discharges are allowed. Storing scrubber water on ships traveling long distances is technically challenging, for example due to the significant space required. An alternative approach to abating scrubber water by storing scrubber water is to switch to the more expensive low- or zero-sulfur fuels, eliminating the need for a scrubber entirely.

3. Methods

In this chapter, the methods developed and applied in this thesis are presented. The first section addresses **RQ1** and provides clarifications regarding the three case studies and their connection to **Paper I**, **Paper II**, and **Paper III**. The second section addresses **RQ2** and outlines the methods used to analyze interactions between different environmental policy instruments (national, regional, and global) and explains how the national data was derived.

3.1. Three cases of policy instrument

The three cases were used to illustrate how the current design of environmental policy instruments can encourage shipping companies to invest in technical and operational measures to reduce emissions to the environment (**RQ1**). The cases involved three different types of policies: a bonus system through environmental discounts on port and fairway fees (**Paper I**), emissions trading through the inclusion of shipping in the EU ETS (**Paper II**), and emissions standards through the IMO global sulfur cap (discussed indirectly in **Paper III** and partly in **Paper II**). A brief description of the policies, abatement strategies, emissions, methods and data applied in the three papers is outlined in Table 3.

Paper I is a retrospective policy evaluation (ex-post evaluation) investigating the Swedish environmental port and fairway discounts in 2020. These discounts are market-based regulations addressing emissions to both air and water, impacting climate, human health and the marine environment. The main method used was cost-benefit analysis for seven model ships. Since it was difficult to argue that the discounts themselves led to any investments or use of the examined abatement strategies, potential emissions reductions were not quantified in this paper; instead, emissions reduction is further explored in both **Paper II** and **Paper III**.

Paper II examines the potential impacts of the inclusion of shipping in the EU ETS, in the near term (2035) through an ex-ante evaluation. The main method in this paper is to compare the abatement cost with the price of emission allowances of CO_{2e} in the EU ETS and includes the European fleet to evaluate shipowners' potential willingness to invest in different abatement strategies that reduce GHG emissions. The study is limited to assessing four different abatement strategies, but it also considers the potential and barriers of these strategies. This paper does not address the impact of environmental damage, which is instead further elaborated upon in **Paper III**.

Paper III is also an ex-post assessment, examining one of the consequences of the IMO global sulfur standard: the introduction of scrubbers. The main method used in this paper is a break-even analysis and considers the entire global fleet that has scrubbers installed. The paper also quantifies the external costs associated with the emissions of polycyclic aromatic hydrocarbons (PAHs) and metals resulting from scrubber discharge water in the Baltic Sea. Unlike the other two papers, **Paper III** focuses specifically on a single abatement strategy for SO_X emissions to the atmosphere, analyzing both the economic and environmental consequences of this strategy. It is important to note that the aim of **Paper III** was not to assess a specific policy but rather to examine a specific consequence of a policy, the adoption of scrubbers, which arose because of the IMO global sulfur cap. In policy case 3 the IMO global sulfur cap was used as a basis for analyzing technical and operational measures for reducing environmental emissions.

3.1.1. Brief description of the methods

The primary approach to evaluate environmental policy instruments in these cases focused on environmental and economic perspectives. The assessments quantified emissions to the atmosphere and/or water (related to policy instrument effectiveness) and the costs associated with different abatement strategies. The evaluation of investment decisions was performed, for example, by investigating the total cost of an abatement strategy (**Paper I**) or by investigating costs of abatement (**Paper II**) (related to efficiency). In **Paper III** a break-even analysis was instead used to investigate the impact of the investment decision in retrospective.

The cost-benefit analysis method in **Paper I** was conducted using model ships that represent various ship segments entering Swedish ports. The selection of model ships was based on Swedish port call statistics and data reported to the Clean Shipping Index (CSI, 2020, SMA, 2023b). For the investment cost component, the study utilized a private cost perspective, focusing on the economic incentives for individual shipowners to adopt specific technologies. Costs were annualized using a 10% interest rate over a 10-year depreciation period (Bosch et al., 2009, Höglund-Isaksson, 2012). The costs of four selected abatement technologies were then compared with the benefits of the port and fairway discounts, both separately and combined. The benefits were quantified by multiplying the number of port calls for each model ship during one year by the "marginal" discount. The marginal discount was defined as the maximum discount (as a percent of the port or fairway fee) a ship could receive by investing in one of the four technologies. Monte Carlo simulations were employed to model uncertainties in the cost components. The uncertainty of the benefit for the shipowners was indirectly analyzed by varying the depreciation period of the costs, representing a change in the scoring system.

The carbon abatement cost in **Paper II** was estimated by dividing the annual cost of abatement by the yearly potential CO_{2e} emissions reduction for each abatement technology. The cost of abatement was defined as the expense of reducing one additional 'unit' of emissions, more precisely as the estimated CO_{2e} emissions and associated costs that occur when a shipowner switches from one technology to another, either by investing in a new ship or retrofitting an existing one. One crucial aspect is that different actors will face varying prospects based on their unique conditions. The cost of new buildings and retrofits was estimated for four and two different abatement strategies respectively (Table 3). The emissions were based on historical emissions and fuel consumption data from 2018 to 2023 for ships recorded in the MRV system (THETIS-MRV, 2023). These emissions and consumption data were then projected onto two future demand scenarios in 2035 (Fridell et al., 2022). Cost estimates were based on the production costs of renewable fuels, projected fossil fuel prices, and estimated investment costs for onboard technologies. Costs for onboard also considered key ship characteristics such as size, engine type, and age (S&P Global, 2023). Policy instruments were assessed from the perspective of private economic stakeholders, including shipping companies and cargo owners.

The break-even method used in **Paper III** involved calculating the point at which the cost savings from using cheaper HFO, due to the installation of scrubbers, offset the initial and operational costs of the scrubbers. This was done by comparing the difference in fuel prices with the investment and maintenance costs of the scrubbers, determining how long it would take for the savings to match or exceed the expenses. The analysis was carried out for all ships identified as having invested in a scrubber between 2014 and 2022. Data from the Ship Traffic Emisssion Assessment Model (STEAM) was utilized in the break-even analysis, covering all ships using scrubbers globally (Johansson et al., 2017, Jalkanen et al., 2021).

The scope of three policy instruments and the reasons for including the abatement strategies in **Paper I** and **Paper II** are described in section 3.1.2. and 3.1.3. A more detailed descriptions of the methods applied can be found in each respective paper.

Table 3 Overview of the three-policy instrument studied. These policy cases used for evaluating the current design of environmental policy instruments (RQ1)

Article	Policy measures	Emission sources/pressures	Relevant areas of protection/impact category	Abatement strategies	Data for estimated fuel consumption and emissions	Method for estimated fuel consumption and emissions	Methods analyzing policy
Paper I	Environmentally differentiated fairways fees	Energy carriers and conversion technologies, anti-fouling paints	Not quantified as the policy measures.	1. OPS 2. SCR, 3. Hybridization (1MWh battery) 4. FRC	Port call statistics, EU-MRV, Sea-web (S&P Global), clean shipping index database, data from shipyard, literature, and stakeholders	Fuel consumption for seven model ships, separated at sea (fuel consumption from EU MRV) and at berth (load factor (kW) and SFC).	Cost-benefit analysis, annuitization of cost
Paper II	EU-ETS, extended EU ETS and scrubber ban	Scrubbers, energy carriers and conversion technologies	Climate change	1. BioMeOH with dual-fuel engine. 2. e-MeOH with dual-fuel engine. 3. LH_2 with PEM fuel cell. 4. Battery electric.	EU MRV (THETIS-MRV, 2023), Sea-web (S&P Global) (S&P Global, 2023), literature and stakeholders (Jivén et al., 2023), IMO GHG study IV (Faber et al., 2020)	Type of fuel from EU MRV based on ratio of CO_2 emissions/ fuel consumption Propulsion and auxiliary output (engine work) (SFC/efficiency, IMO GHG study IV). Voyage specific energy requirement. Emission Fuel based (SO_X, CH_4, N_2O) and output based (engine work) (NO_X, PM, scrubber water)	Carbon abatement cost, scenario analysis
Paper III	IMO 2020 sulfur cap	Scrubbers	Marine ecotoxicity	Scrubbers	AIS, Sea-web (S&P Global, 2023), Ship & bunker (Ship & Bunker, 2024), literature, and stakeholders	STEAM model (Johansson et al., 2017, Jalkanen et al., 2021)	Break-even analysis total cost

3.1.2. Scope of the three policy instruments

The scope includes the timeframe, geographical coverage, ship segment (ship type and size), and operation (at sea, in port). These are summarized for the three policy cases in Table 4. It is worth noting that in **Paper I**, seven model ships are used, raising the question of whether these adequately represent the diversity of the Swedish fleet. This aspect is therefore assessed in section 4.2.2.

Table 4. Summary of the scope of the three policies

Scope	Fairway discount	EU ETS	IMO global sulfur cap
Timeframe	Until SMA change structure	Reach net zero 2050	Permanent
Geographical coverage	Ships arriving at Swedish ports	100% internal traffic and for 50% of the emissions in the in- and outgoing traffic (EEA)	Global
Operational mode	All modes	All modes	All modes
Emissions	CO_2, NO_X, Water and Waste, SO_X/PM and Chemicals	CO_{2e}	Sulfur
Ship segment	All ships above 300 GT that are paying the fairway fees,	All cargo and passenger ships above 5000 GT.	All ships globally

3.1.3. Selection of abatement strategies

The Swedish fairway fee discount includes several additional types of abatement strategies beyond those investigated in **Paper I**, such as efficiency measures, educational programs, and waste management. The abatement strategies chosen in **Paper I** were selected because they appear effective in reducing emissions identified as particularly relevant in the Baltic Sea by Ytreberg et al. (2021).

In **Paper II**, only four abatement strategies were used to illustrate the GHG abatement cost. These strategies were chosen because they represent four distinct paths for carbon reduction:

1. **Using bio-MeOH with a dual-fuel engine:** Bio-MeOH was selected since it can be produced from a wide variety of feedstocks. Other relevant biofuels include LBG, HVO and RME (Brynolf et al., 2023, Bach et al., 2021). The concerns about the sustainable production potential for biofuels (Broch et al., 2013, Cowie et al., 2017, Berndes et al., 2016, Sparovek et al., 2016, Searchinger et al., 2018, Reid et al., 2020) have similarities across different biofuel types. The raw material costs associated with producing these different types of biofuels are also comparable. A dual-fuel engine with methanol tanks was selected because this technology is proven to work onboard ships and can be used for retrofitting.
2. **e-MeOH with a dual-fuel engine:** This option was chosen to represent liquid electrofuels. The potential for electricity-based fuels is very high, e.g., solar radiation to Earth is about 9 000 times higher than current energy usage (1000 times higher over land) (Brynolf et al., 2022). Although this path is relatively inefficient in converting electricity, leading to high costs (and emissions) for current abatement efforts, it represents a possible future direction.
3. **LH_2 with a Proton Exchange Membrane (PEM) fuel cell:** This strategy has the potential for greater efficiency than e-MeOH in internal combustion engines (ICE) and represents an onboard technology that reduces air pollutants such as PM and NO_X. However, it has significantly higher CAPEX, challenges with LH_2 storage, limited development in PEMFC technology, and very limited demonstrated onboard use.

4. **Direct electricity use with batteries:** This abatement strategy is the most efficient and eliminates all direct emissions to air. However, the CAPEX and storage requirements for batteries limit their potential relative to other technologies.

In **Paper II**, two other abatement strategies, OPS and energy efficiency, were included in the baseline, for two reasons: (1) they are required by regulation (EP and EUCO, 2023c, EP and EUCO, 2023d), and (2) under favorable conditions, electricity can be cheaper to use, as shown in **Paper I**.

3.2. Evaluating interactions between environmental policies

Two extended analyses based on **Paper I** and **II** were conducted in this thesis to address **RQ2**: *"Identify potential interactions with other policies and/or unintended consequences of the policy combination, influence shipowners' decisions to invest in abatement strategies, and what is the subsequent impact on emissions other than those targeted by the policy"*. This evaluation primarily used the concept and methods outlined in **Paper II**.

The evaluation was accomplished by stepwise adding individual policies. The interaction effects were evaluated by comparing abatement costs and emissions reductions at each step.

In the first complementary analysis, the European fleet (referring to inbound and outbound ships to an EEA port in this thesis) in 2023 is analyzed under global and regional policies. The following policies are included step-by-step:

- **Step 1: Energy Efficiency Design Index (EEDI):** existing policy aiming at improving the energy efficiency of newly built ships. This policy was included as a baseline in **Paper II** but was never evaluated separately.
- **Step 2: Global ban on discharging scrubber water**: potential policy implying that all ships are using MGO or VLSFO instead of HFO (as in **Paper II**).
- **Step 3: Extension of the EU ETS:** possible policy implying expanding the scope of EU ETS to include 100% of emissions in the in- and outgoing traffic (as in **Paper II**).

In the second complementary analysis ships entering Swedish ports were studied and global, regional and national policies are considered. In addition to Steps 1-3 above a fourth step is added:

- **Step 4: Environmentally differentiated fairway fees:** existing policy is assessed by applying a discount based on the number of calls in Sweden (as in **Paper I**). Each ship has a unique ID, allowing for the matching of data from MRV statistics with SMA port call statistics.

The Swedish case also discusses an expanded application of the EU ETS, including smaller (<5000 GT) ships. The calculation of Swedish emissions was updated in this thesis and is described in more detail below.

3.2.1. The national perspective: Sweden

The estimation of the Swedish emissions of CO_2 was based on the Swedish port call statistics (SMA, 2023b). The number of calls in 2022 can be seen in Table 5. In the port call statistics, each port call is accompanied by information about the arrival port and the destination port. This information was used to identify the routes of all arrivals. The distances of the routes ($Distance_{port\ call}$) were gathered from Searoutes (2022) which represents generic routes, while the emissions factors (CO_2 *per NM*) were from THETIS-MRV (2023).

$$Swedish\ emissions = \sum_{port\ call,\ \ ship} CO_2\ per\ NM_{ship} \cdot Distance_{port\ call} \ (1)$$

The method applied is described in more depth in Jivén et al. (2023), Styhre et al. (2024). However, some essential assumptions are summarized in Table 6 and Table 7.

Table 5. Swedish port calls statistics in 2022 [number of port calls]

Alternative ship type	> 5 000 GT	400-5 000 GT	Total
Container ship	1 725	167	1 892
Cruise ship	462	18	480
General cargo	1 719	9 175	10 894
LG tanker	514	249	763
Oil tanker	226	120	346
Product tanker	2 615	2 801	5 416
Reefer	31		31
RoPax	44 546	2 203	46 749
RoRo Cargo	1 832	2	1 834
Vehicle carrier	558		558
Total	**54 228**	**14 735**	**68 963**

Table 6. Key assumption regarding the calculation of Swedish emissions of CO_2 and Swedish port calls statistics in 2022of

Type of note	Comments
Emissions calculation	Emissions of CO_2 are computed by multiplying the distance traveled by the emission factor specific to each individual route.
Data source	Port call statistics are sourced from SMA 2022.
Ropax lines information source	The destination and arrival ports for the majority of RoPax lines were derived from online sources, as these routes were not initially available in the original statistics from SMA.
Emission factors	Emission factors are derived from the MRV data for each individual ship.
Use of default emission factors	Default emission factors for CO_2/NM are employed when MRV data is unavailable. These have been updated in this study and can be found Appendix.
Distance calculation	Distance Calculation: Emissions are calculated for 50% of the distance between two ports.
Distance	Distances were based on SeaRoute (2023-2024), VesselTracker (2017), or default distances in some cases.
Missing distance information	For routes with missing distance data, default distances are determined based on the country associated with the port. If this information is unavailable, defaults are based on the average distance for that ship segment.
Energy calculation	Energy calculations are rough estimates derived from MRV data, see **Paper II**, assuming ships use only two types of fuels.
Port vs. sea emissions	Estimates of emissions at port versus at sea for each ship are very rough. The accuracy of the assumption depends on whether the route is representative for that ship. For example. estimates may be more accurate for RoPax ships compared to container feeders.

Table 7. Default emission factors for small ships, used when calculating the Swedish emissions

Ship category	Annual average CO_2 emissions per distance modified [kg CO_2 / n mile]	Comment
Bulk carrier (0-10 000 dwt)	137	
Product tanker (0 - 5 000 dwt)	157	
Container ship small feeder	221	
General cargo (0-5 000 dwt)	0.0061*dwt +107	Updated (Segerer, 2024)
LG tanker (0-25 000 dwt)	223	
Oil tanker (0 - 5 000 dwt)	113	
RoRo (0 - 5 000 dwt)	174	
RoPax/Ferry (0-2000 GT)	125	
Vehicle carrier (0-10 000 dwt)	216	

3.2.2. Definition of fleets

The term "fleet" could be used in various ways. In this thesis, it refers to multiple ships grouped together due to their relevance to the policy instruments that were analyzed. The grouping was based on factors such as the ship segments included, the ports they visited, or the geographical areas where they had operated. Below, the two terms, "the European fleet" and "the Swedish fleet", which were used in this thesis, are described below.

The European fleet was defined as ships reporting under the MRV regulation (EP and EUCO, 2024, THETIS-MRV, 2023), representing commercial ships traveling to and from EEA ports over one year. The definition only included ships larger than 5,000 GT and covered 12 segments: bulk, container, general cargo, oil tanker, product tanker, LNG tanker, RoRo, vehicle carrier, RoPax, reefer, and cruise. Only 50% of the fleet emissions and fuel consumption were included for outgoing and ingoing travels to an EEA port, while 100% of emissions and fuel consumption were included for intraregional travels.

The Swedish fleet included all ships that paid the fairway fee, representing commercial ships (>300 GT) traveling to and from Swedish ports over one year (SMA, 2023b). Similar to the European fleet, Swedish emissions were defined as 50% of the fleet emissions for outgoing and ingoing travels, while 100% of emissions and fuel consumption were included for domestic travels.

4. Results and discussion

The first section examines three specific policies for context, while the final section assesses the interaction between different policies.

4.1. Current design of environmental policy instrument: three case studies

The following section elaborates on how the current design of an environmental policy instrument potentially can encourage shipping companies to invest in technical and operational measures to reduce environmental emissions. This is done by using three case studies that are directly or indirectly derived from **Paper I**, **Paper II,** and **Paper III**.

4.1.1. Swedish environmentally differentiated port and fairway fees (Case 1)

The aim of **Paper I** was to "investigate whether environmental discounts provided at Swedish ports and fairways could significantly impact shipowners, and consequently determine if the discounts could potentially lead to emission reductions in the Baltic Sea". The results indicate that the discounts alone are unlikely to influence shipowners' willingness to invest in any of the investigated abatement strategies (selective catalytic reduction (SCR), switching to biocide-free antifouling coating, small battery (1 MWh), and onshore power supply (OPS), as the discounts are too low relative to the cost of these abatement strategies. Port discounts were too low for all types of ships studied; in four of the five ports, the discount amounted to only 10% of the port fee. The conclusion of the study suggests that the policy is not effective in reducing emissions since shipping companies are not likely to invest in technical and operational measures (**RQ1**). This is also in line with what for example Trafikanalys (2017) concluded before the modification of the environmentally differentiated fairway due. Trafikanalys (2017) argued that the fairway dues system would weaken previous environmental incentive, with a broader index and a smaller portion of the fees environmentally differentiated compared to the current system, likely leading to weaker overall environmental incentives.

If all the funds in the fairway discount had been allocated to a specific purpose, either a technology or a specific emission category, the results in **Paper I** suggest that the discount amount could approach the cost of some abatement strategies. For instance, switching from a commercial copper-based antifouling paint to a biocide-free silicone-based fouling release paint appears to be feasible within the rebate "ceiling." This strategy is likely advantageous for the shipowner as well since fouling release coatings have demonstrated equal or superior effectiveness compared to copper-based paints in preventing biological fouling on the ship hull (Lagerström et al., 2022). However, the structure of the Swedish fairway discount is not designed to encourage investment (**RQ1**). Instead, the discount is applied continuously based on the number of calls to port. Over time, the system has undergone adjustments, which further creates uncertainty for shipowners considering investment decisions based on this policy instrument. Ultimately, as argued in **Paper I** the total discounts provided (€3.6 million in 2018 and €5.8 million in 2019) are by itself insufficient to motivate investment in cleaner technologies when spread out on many different abatement strategies and shipowners.

4.1.2. Inclusion of shipping in EU ETS (Case 2)

The EU ETS has existed since 2005, but the inclusion of ships in the trading scheme began in 2024. **Paper II** evaluates whether the EU ETS alone, or in combination with other policy instruments, can encourage shipping companies to invest in cleaner fuels or energy carriers and thus reduce GHG emissions.

Paper II shows that bio-MeOH could be a feasible alternative from a cost perspective, and that even current EU ETS prices (70–100 euros/ tonneCO_{2e}, see Figure 3) can have an influence, if biomass prices are low. However, domestic biomass potential is limited unless feedstock from forestry or agriculture is included. As mentioned in **Paper II** a rough estimate suggests that the European waste biomass, meeting sustainability criteria, could potentially supply 28–63% of the European fleet's fuel consumption at sea. This estimate assumes that only 50% of the incoming and outgoing traffic from the European Economic Area (EEA) is considered, and that all available waste biomass feedstock is dedicated to shipping. Batteries could be feasible for ships on shorter routes with lower voyage energy requirements. RoPax ships show potential for this application, but battery size and charging challenges need to be addressed. PEMFC and e-MeOH are cost efficient abatement strategies at higher EU ETS prices (300 euros/tonneCO_{2e}) but would require a significant increase in electricity production in combination with low electricity prices.

As further elaborated upon in **Paper II,** expanding the EU ETS to include 100% of emissions from in- and outgoing traffic would spread investment costs across a larger emissions scope, potentially increasing shipowners' willingness to invest in PEMFCs, batteries, and retrofits for conventional ships to dual-fuel systems. For ships using scrubbers, a ban on scrubbers would increase the baseline cost (as low sulfur fuel would be needed instead) and thereby reduce the resulting abatement cost for the investigated alternatives more than the current EU ETS price. However, the uncertainty in renewable energy production has a larger impact on the abatement cost than both expanding the EU ETS and introducing a scrubber ban.

Furthermore, **Paper II** is based on a future scenario of estimated production costs for renewable fuels, rather than on actual prices. To address **RQ1**, i.e. if *current design of environmental policy instruments encourages shipping companies to invest in technical and operational measures to reduce environmental emissions*, it is also relevant to assess the present conditions. If the high renewable fuel production costs in **Paper II** are assumed to better represent current price levels, in combination with higher prices for batteries (500 euros/kWh) (**Paper I**) and PEMFCs (1400 euros/kW) (Jivén et al., 2023), while considering the fleet composition of 2023 and fossil fuel prices from that year, the results appear as shown in Figure 5. The figure illustrates the estimated CO_{2e} abatement cost (y-axis) for 2023 across four strategies for each ship in the entire European fleet as a function of the CO_2 emissions from each ship, where 50% of CO_{2e} emissions from ingoing and outgoing traveland 100% of domestic is included. One can, for example, see that for bio-MeOH there is many ships, with annual emissions reduction potential up to about 73 Mtonnes, where the abatement cost is below 300 €/tonne CO_2, while for batteries the abatement cost is below 300 €/tonne CO_2 only for ships emitting less than approximately 3 Mtonne CO_2 per year. The result suggests that current EU ETS prices (50-75 € per tonne CO_2, see Figure 3) are likely too low to incentivize abatement (**RQ1**), except for ships operating on shorter, frequent routes where batteries could be a feasible option.

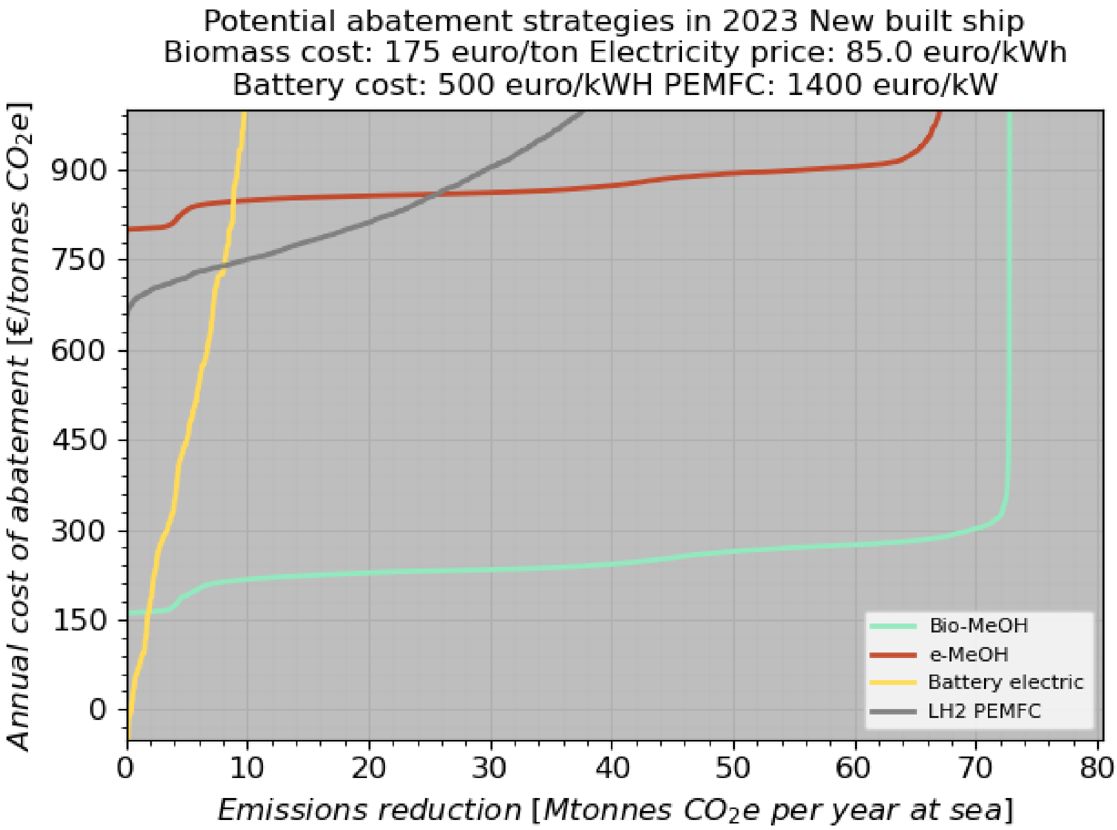

Figure 5. Estimated abatement cost in the European fleet based on fossil fuel prices in 2023 and high cost of producing renewable alternatives and high cost of battery and PEMFC.

The issue with low use of alternative fuels in shipping sector today

It is important to note that the aim of the analysis in Figure 5 is not to decide which abatement strategies are most efficient to prioritize from a societal perspective in the long run. Evaluating abatement cost curves, like the one in Figure 5 above, rather provides an indication of the level at which an environmental policy instrument, such as a subsidy, refundable emissions payment, or feed-in tariff, would need to be to incentivize action and the magnitude of the reduction. The abatement cost curve also describes how different ships will face different types of costs, which, for example, can be valuable inputs when designing the policy or considering different timeframes.

Focusing solely on the efficiency of the environmental policy instrument, like EU ETS, could in this case create lock-in effects and sub-optimal solutions, as new technologies face market entry barriers and find it difficult to compete (Geels, 2005, Azar and Sandén, 2011). In **Paper II**, bio-MeOH was shown to have the lowest abatement cost (except for RoPax ships covering shorter distances). If policy were to "let the market decide," as with the EU ETS, results in this thesis and partly in **Paper II** suggest that ships (and other sectors needing liquid fuels) would likely begin with electrification (for ships frequently traveling shorter distances) while some ships would begin to use biofuel. Early indications suggest this trend is occurring, as shown by battery-powered ferries in Norway (Bach et al., 2020) and Sweden (**Paper I**), and initial steps within the shipping sector toward biofuels (Fagerbaek Guldstrand and Wall, 2024, Ahlström and Johnsson, 2023, Emanuelsson and Fransson, 2023). However, as the potential for biofuels is limited and/or poses challenges, relying solely on the lowest-cost options could delay the adoption of other alternatives.

The time delay is important because gaining experience with hydrogen (involving fuel cells and complex fueling), e-fuels, or other potentially costly technologies will take time, so building knowledge, doing mistakes, and creating stable supply chains is essential if the goal is to achieve net-zero emissions of GHG within the desired timeframe. A more technology-specific policy, with a high guaranteed price and a long-term horizon, would likely be necessary to incentivize ship companies, ports, and energy producers to invest in and operate this type of costly carbon abatement strategies. An example of such a policy is the feed-in tariff used for solar and wind power (Jacobsson and Lauber, 2004). However, considering the current carbon intensity of electricity production (on average 200 g CO_{2e}/kWh)(EEA, 2024), it is crucial not to rush this process too quickly.

4.1.3. The rise of scrubbers: IMO global sulfur cap (Case 3)

Globally, a new amendment to the IMO global sulfur cap came into force in 2020 including a new emissions standard for ships to reduce sulfur emissions. The new global sulfur standard for ships was very effective in reducing emissions of sulfur to air, and reductions in concentrations of SO_X in the air were even measurable after implementation, see for example Tauchi et al. (2022). A side effect of the updated IMO global sulfur cap was that many ships chose to install and use a scrubber in combination with HFO instead of using low sulfur fuels such as MGO. The use of the latter give rise to lower emissions of e.g. PAHs and metals compared to HFO (Lunde Hermansson et al., 2021).

The underlying motive to assess the economic impact in **Paper III** is that several ports and nations already have decided on local bans on the discharges of scrubber water (BIMCO, 2024, EGCSA, 2024). Some actors in the shipping sector have been questioning this type of decisions since they claim that they have been investing in scrubbers in "good faith" in accordance with the new regulation and could potentially lose the investment if further regulations are taken (**Paper III**).

Since scrubber water is a large source of PAHs and metal to the marine environment in areas with high shipping activity, such as the Baltic Sea (Ytreberg et al., 2022), it is crucial to identify the magnitude of the damage that the scrubber discharge water may have on the Baltic Sea. This is particularly of importance since **Paper III** showed that the Baltic Sea, which is designated as a Particularly Sensitive Sea Area (PSSA) by IMO, has received 3.2 billion m^3 of open loop scrubber water between 2015-2022. The environmental impact assessment in **Paper III** was not carried out in a way that enables a clear comparison between the damage cost and the cost/benefit of using a scrubber, since different methodological approaches were applied. Additionally, the damage cost calculation was not entirely comprehensive, as it only considered nine metals and ten PAHs for which ReCiPe characterization factors (Huijbregts et al., 2017, Huijbregts et al., 2016), were available (**Paper III).** Consequently, the damage costs are likely underestimated given that 69 substances have been identified in scrubber water (Lunde Hermansson, 2024). Furthermore, other external costs, such as influence on GHG emissions, and marine acidification, were not included.

The result from **Paper III** shows that 51% of all shipowners had already reached break-even by the end of 2022 in the central cost case (i.e. had made savings on fuel corresponding to the entire scrubber installation cost). This is in line with the fact that over 5 000 of all the world's ships chose to invest in scrubbers instead of using low-sulfur alternatives. However, there is indication that the percentage (51%) is an underestimate, as shipowners have been able to store large volumes of oil when the price is low and use it when the price is high, an effect that **Paper III** was not able to take into consideration. The damage cost for the marine environment due to scrubber water discharge in the Baltic Sea for the period 2014-2022 was estimated to be over 680 million euro.

Hence, **Paper III** is an illustrative example where the environmental policy instrument was effective in reducing emissions of SO_X to air, not only encouraging but forcing shipowners to take measures (**RQ2**). However, due to the design of the policy, which permits the use of scrubbers as an allowed means of compliance, the discharge of scrubber water containing a cocktail of hazardous substances has increased. As clearly demonstrated in **Paper III,** the main driver for this new waste stream and subsequent marine pollution seems to be the ship companies' intent to reduce their costs.

4.2. Interactions between different environmental policies

For the three policy cases investigated, costs and reduction potentials were examined separately. However, to better understand the connections between different types of environmental policy instruments it is important to assess: *"How various national, regional, and global policy instruments, both individually and in combination, influence shipowners' decisions to invest in abatement strategies" (**RQ2**).*

In this chapter, the abatement costs and reduction potentials of the three cases are combined by evaluating the interactions among four policies: 1) EEDI (**Paper II**), 2) a global ban on scrubbers (**Paper II** related to **Paper III**), 3) the extension of the EU ETS (Paper II), and 4) the Swedish environmentally differentiated fairway fee (**Paper I**). Policies 1–3 are initially evaluated for Europe, and all these policies are then applied to a fourth case that includes only ships arriving or departing to Sweden (Policy 4). In the Swedish case, an additional expansion of the EU ETS is discussed: including small ships, i.e., ships <5000 GT.

4.2.1. European fleet scenario analysis of different policy instrument

Figure 6 demonstrates one way to conceptualize the interaction among three policies (EEDI, scrubber ban and extension of EU ETS) in the shipping sector for the European fleet. The figure illustrates the renewable central-cost scenario for the abatement cost of e-MeOH in a hypothetical scenario, where all ships were rebuilt to use e-MeOH. The methods used in the analysis were the same as presented in **Paper II**. The dark brown curve shows the abatement cost for e-MeOH without the three studied policy instruments applied (Without EEDI). The first step represents a change to the Baseline where the impact of the EEDI (a policy aimed at improving the energy efficiency of newly built ships) is considered. The new resulting abatement cost is represented by the light-blue curve (Baseline). The main effect of the EEDI policy is to reduce total emissions of GHG. The second step illustrates the effect of a global ban on scrubbers, which would increase the base cost (and thereby indirectly lower the cost of abatement) as shipowners would be forced to use low-sulfur fuel with a higher cost, such as MGO instead of HFO. The resulting abatement cost for changing to e-MeOH after adding a scrubber ban is shown by the green line (Scrubber ban). In the third step, an extension of the EU ETS is added on top of the other two policies, covering 100% of inbound and outbound traffic. This extension decreases the resulting abatement cost, as the capital cost of retrofitting a ship is distributed over a larger operational scope and increases the CO_{2e} reduction potential by including more emissions within the policy's scope, resulting in the yellow curve. The abatement cost in the final case is still above the current EU ETS allowance price

(50-75 € per tonne CO_2, see Figure 3), since the extended EU ETS only reduces the annualized investment cost and not the variable costs.

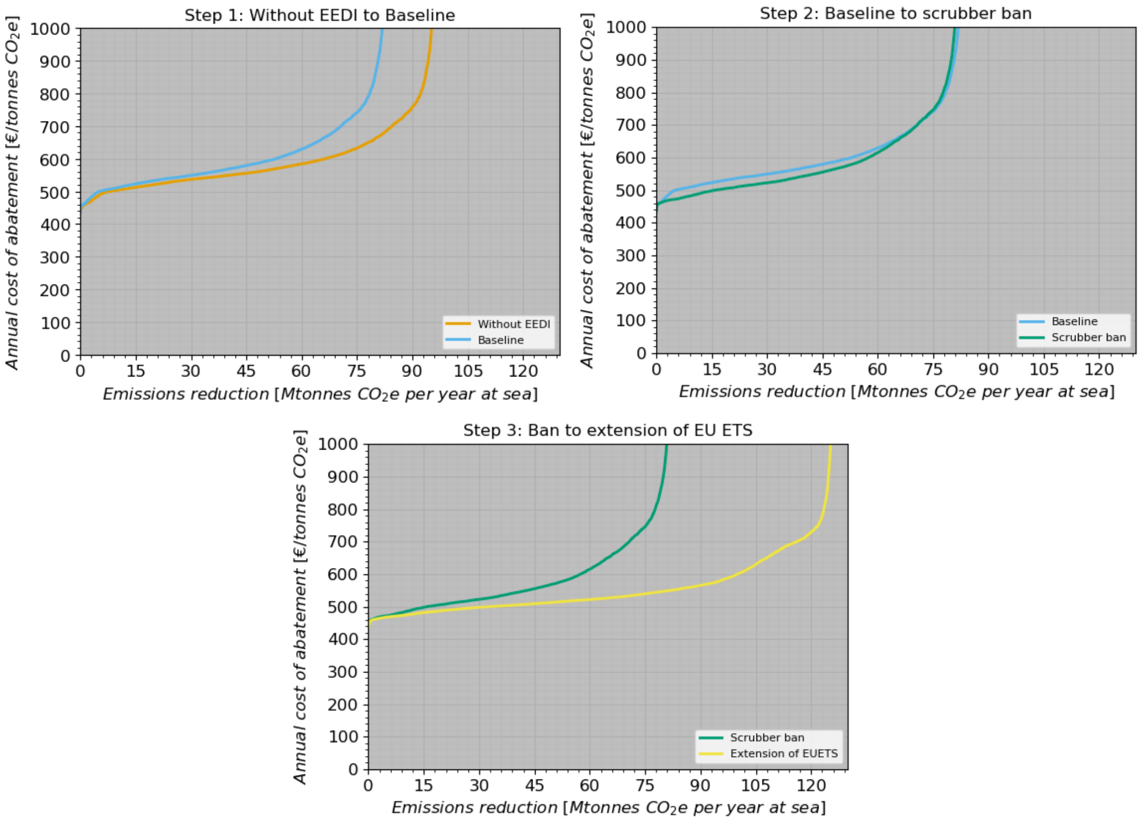

Figure 6. Estimated resulting abatement cost and emissions reduction potential for retrofitting e-MeOH ships and the CO_{2e} reduction potential for one existing policy (EEDI) and two proposed policies (a global scrubber ban and an extension of the EU ETS) for the European fleet in 2035. The three figures represent three sequential steps: 1) the effect of the EEDI on the fleet compared to a scenario without efficiency improvements, 2) the additional effect of implementing a global scrubber ban, and 3) the effect of expanding the scope of EU ETS.

In terms of economic and environmental impacts, there are symbiotic relationships among the three policies. The sulfur cap increases fossil fuel prices for ships that was previously using HFO and did not have a scrubber (**Paper III**), thereby lowering abatement costs for these shipowners (**Paper II**), which, in the long run, is fundamental for reducing the dependence on fossil fuels. The additional cost of fossil fuels adds to the cost of purchasing allowances in the EU ETS and indirectly lowering the resulting abatement cost (**Paper II**). Similarly, the use of scrubbers raises abatement costs (**Paper II**) due to decreased fuel prices (for HFO vs. MGO). Since the fuel choice and use of scrubbers influence emissions to both air and water, they are all interconnected. The discount on fairway and port fees (**Paper I**) similarly affects the effective abatement costs and interacts with other regulations, such as the NO_X fund (Parsmo et al., 2017). Since these regulations operate at different geographical levels and are governed by different regulatory bodies, some shipowners will experience very different relationships between costs and benefits compared to others.

As also further discussed in **Paper III** the use of HFO also directly affects emissions of air pollutants, such as PM and CO_2. PM emissions from low-sulfur fuels are lower than those from HFO with a scrubber (Lunde Hermansson et al., 2021). Furthermore, as discussed in **Paper III**, scrubbers increase fuel consumption onboard due to their energy demands, leading to greater fuel consumption per distance traveled and thus higher direct CO_2 emissions under operation.

4.2.2. The national perspective: Sweden

In **Paper I**, model ships were used to assess the impact of the fairway discount on shipowners' investment decisions. This type of analysis can also be conducted at the fleet level by evaluating the costs and emissions for the entire fleet. Swedish emissions of GHG were estimated in Styhre et al. (2024) and are slightly modified in this thesis. Figure 7 shows the resulting GHG emissions (WTW) for the entire fleet, broken down by ship segment and grouped according to the number of fairway fees paid in 2022. As illustrated, the RoPax and RoRo segments include many ships that frequently call at Swedish ports, which aligns with the assumptions made for the seven model ships in **Paper I**. The estimated emissions represents 50% of calculated emissions from outgoing and incoming ships and 100% of domestic emissions, totaling 4.58 million tonnes of CO_{2e}.

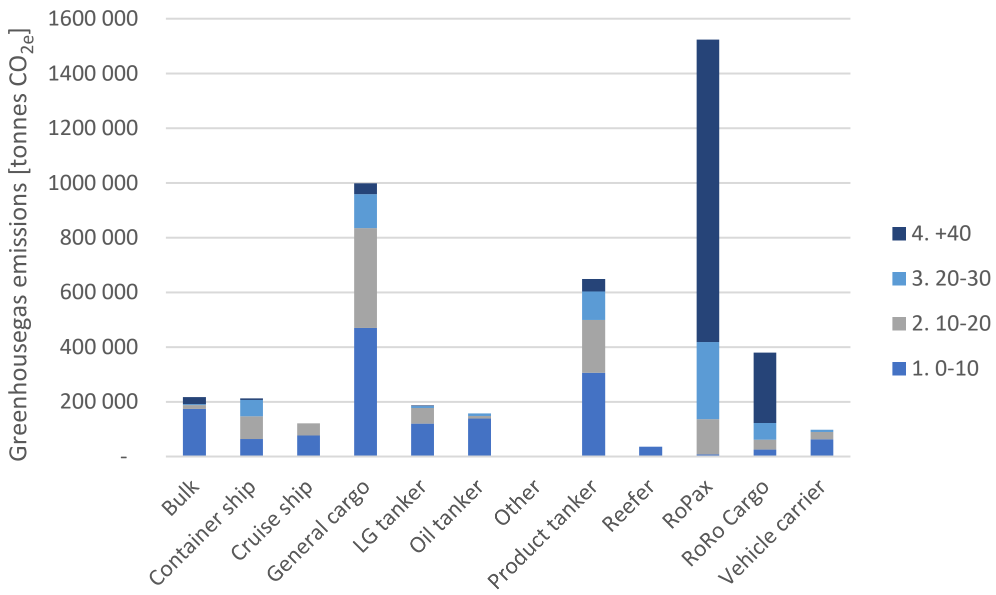

Figure 7. Swedish fleet GHG emissions (WTW) estimate including 50 % of incoming and outgoing ships and 100% of domestic emissions. The colors indicate how many times each ship paid for the fairway fee in 2022. The figure includes all ships that paid the fairway fee, i.e., passenger and cargo ships that are larger than 300 GT.

Figure 8 shows the reduction in abatement costs when applying the discount from Swedish environmentally differentiated fairway fees (**Paper I**). Ships entering Swedish ports Figure 7 are extracted from the European scenarios (Figure 6) and included in Figure 7, meaning that smaller ships (300–5000 GT) from the Swedish data and ships not entering Swedish ports are excluded. As shown in Figure 8, the impact of adding the discount in the analysis is minimal. This result aligns with the conclusions from **Paper I**, that the Swedish environmentally differentiated fairway fees is not designed to support CO_2e abatement.

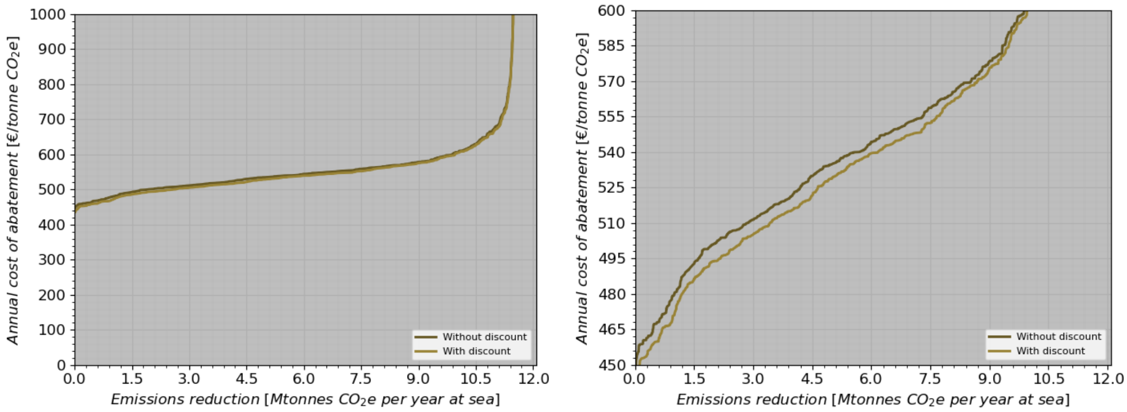

Figure 8. Abatement cost for e-MeOH compared to CO_{2e} reduction potential for one existing policy (the environmentally differentiated discount for Swedish fairways), for ships entering a Swedish port in 2035. The figure includes all emissions reduction potential for the extended scope pf EU ETS. The left figure is cut at 0-1000 euro/tonnesCO_{2e} while the left figure is cut at 450-600 euro/tonnesCO_{2e}

An important aspect of the EU ETS regulation is that it excludes small ships, a point illustrated for Sweden in Figure 9. Small ships calling at Swedish ports in 2022 represent about 20% of direct GHG emissions from the total Swedish fleet. From a Swedish perspective, small general cargo ships and product tankers appear to be particularly relevant, result that is in line with conclusions in Vierth et al. (2024) and Trosvik and Brynolf (2024). This case study illustrates the importance of also including smaller ships in regulations, like for example EU ETS.

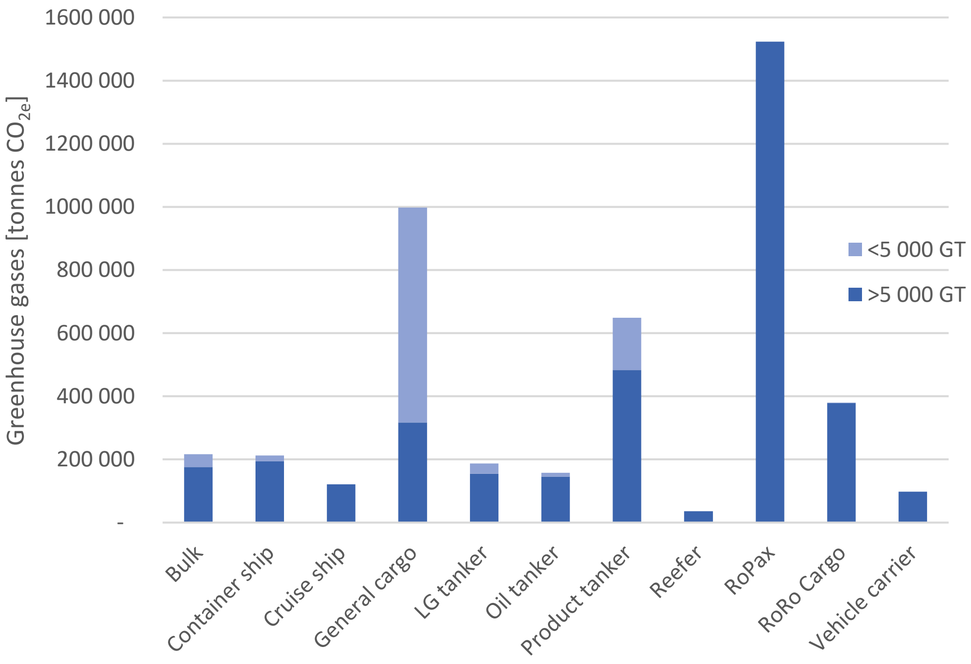

Figure 9. Swedish fleet GHG emissions (WTW) estimate including 50 % of incoming and outgoing ships and 100% of domestic emissions 2022, representing 50% of distance for incoming and outgoing ships, categorized in size bins.

5. Conclusions

In this thesis, three different policy instruments have been evaluated: environmental discounts on port and fairway fees (**Paper I**), the inclusion of shipping in the EU ETS (**Paper II**), and the IMO global sulfur cap (indirectly in **Paper III**). The results indicate that neither the discounts nor the EU ETS are currently leading to investments in the selected "*technical and operational measures*" (**RQ1**). For the EU ETS, the policy may have an effect in the long term if the EU ETS price increases and/or the cost of renewable options decreases. Conversely, the IMO global sulfur cap effectively reduced SOx emissions to the atmosphere by requiring shipowners to take *"measures"* (**RQ1**), though allowing scrubbers has resulted in increased marine pollution. This outcome also raises the question of whether policy evaluations should consider a broader perspective **(RQ2**).

In this thesis, national, regional, and global policy instruments were evaluated in combination to assess how they interact and influence each other (**RQ2**). It is shown that a scrubber ban (global) and efficiency improvements in combination with the expansion of the EU ETS (regional) have a complementary impact. However, this effect is not substantial enough to motivate a shipowner to retrofit and operate a ship powered by e-MeOH. The impact of discounts on port and fairway fees (national) is small in comparison.

Another goal conflict involves the time aspect. On one hand, the "environmental problem" may require immediate action (such as reduction of GHGs) because the turnover of the shipping fleet is slow, knowledge, competence and infrastructure are missing, implying that change needs to start now if future goals are to be met. However, as exemplified with scrubbers it takes time for the IMO to develop and adopt final regulations. As shown in **Paper III**, the wide-scale use of scrubbers also implies costs related to the degradation of the marine environment, and the cost of not restricting scrubbers should be factored into the decision-making process. Similarly, a technology may have high emissions now (such as production of electrofuels) but holds potential for emission reductions in the future (**Paper II**). In these cases, it is essential to consider both short-term and long-term effectiveness to determine whether the policy instrument is effective over different timeframes.

Given these factors, it can be challenging to prioritize aspects when evaluating a policy design. One specific area, only partially considered in **Paper III**, is quantifying the societal damage or benefit of a particular abatement strategy. Therefore, this thesis suggests exploring this aspect further in future studies.

6. Future outlook: Broadening the perspective

For market-based policy instruments, such as subsidies, taxes, or fees, it is required that private actors invest in measures to achieve environmental improvements, as elaborated in **Paper I** and **Paper II**. In these papers, the abatement strategies are analyzed from a private economic perspective. In addition to assessing the economic and “real-world” effects of decisions made by the private shipping sector, it is also relevant to assess the environmental impact of ships. Evaluating the societal effects of ships´ environmental impacts provides an important cornerstone for prioritizing regulations, which can, for example, be done from a socio-economic perspective.

The socio-economic perspective is “explored” in **Paper III**, which includes an estimate of external costs of marine ecotoxicity due to discharge of scrubber water from shipping in the Baltic Sea. This was the first step in the work toward conducting a cost-benefit analysis for different abatement strategies in the shipping sector, considering only one damage cost (marine ecotoxicity) in one water basin (the Baltic Sea). In a study by Ytreberg et al. (2021), an assessment of these damage costs for a broad spectrum of ship pollutants in the Baltic Sea in 2018 was conducted. However, considering that many decisions relevant for the shipping sector now are made at the EU level, there is an increased relevance in exploring whether the data and methodology can also be used to evaluate externalities on a European scale. Moreover, there might be more recent data available compared to what was utilized by Ytreberg et al. (2021), highlighting a potential improvement in accuracy. Future work is therefore suggested to include the socio-economic perspective and include external costs, which was not considered in **Paper I** and **Paper II**.

In this context, the socio-economic perspective is theoretical. It can be exemplified by a hypothetical social planner who aims to optimize welfare for society. It is theoretical in the sense that actors in society generally makes individual decisions based on actual prices, abstract principles, or even randomness, while in this context, decisions are made by a social planner based on a “bounded” definition of welfare. Furthermore, in this type of socio-economic assessment, the social planner attempts to weigh costs and benefits from a broader perspective, considering the entire or different subsets of society. This impacts, for example, discount rate and investment horizon, but also what is viewed as ”cost” and what is not.

A social planner should strive to include all societal costs, such as environmental externalities, even though these are not always reflected in the market prices. On the other hand, certain costs, such as the EU ETS price analyzed in **Paper I**, are not considered as direct cost from the social planner's perspective. The prices of a market-based permit, tax, or subsidy are in this perspective not considered as a cost or benefit but is rather a transfer between actors. Nevertheless, the price of a permit in EU ETS could still attempt to include the external costs of GHG, i.e. it may indirectly cover parts or all the external costs.

The external cost could be seen as the market's failure to include all costs of a product in its price (Sterner and Coria, 2012). For instance, in the context of fishing, the product, fish, may incur external costs as environmental consequences, related to the fuel combustion (resulting in global warming, human toxicity, or eutrophication). Additionally, substances directly released into the water from fishing vessels could contribute to other environmental impacts such as marine eutrophication and ecotoxicity. Furthermore, these external costs might extend to social consequences, such as the livelihoods of fishermen, food security, or the cultural identity of local communities.

For society to be able to compensate for the maritime industry's investment costs or create new regulations, a value is needed including what environmental benefits a certain measure has for society. One way to describe this is with the external costs that a certain emission gives rise to. Hence, there is a need to analyze and calculate the maritime industry's external costs and compare these with direct costs (such as the technology costs) in order to evaluate the benefit of various abatement strategies for society and their expected outcome.

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