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Evaluating environmental policy instruments for ships

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Abstract

Maritime transport is central to the transport system but creates pressures affecting climate, human health, air quality, and marine ecosystems. Policy instruments have been introduced to limit emissions and other environmental pressures from ships. Their effectiveness depends on targeted emission reductions, as well as on their influence on investment decisions, policy interactions, and environmental outcomes. This thesis evaluates shipping policy instruments from three perspectives: (1) their influence on shipowners' choice of abatement strategies, (2) interactions among policies, technologies, environmental pressures, and actors, and (3) the role of external costs, uncertainty, and evaluative assumptions. The analysis combines techno-economic assessment, fleet and scenario modelling, abatement-cost analysis, external-cost assessment, and stakeholder interviews.

The results show that current policy instruments do not consistently provide the early and predictable incentives needed to initiate a larger-scale transition in shipping. Swedish environmental discounts are too small relative to investment costs to motivate the measures examined, and the EU ETS price signal remains below estimated abatement costs. Although FuelEU Maritime becomes substantially stricter over time, its early targets and flexibility mechanisms allow compliance with limited near-term changes. The IMO sulfur limit demonstrates that binding requirements can induce action, but it also shows the importance of compliance options: the use of scrubbers reduced sulfur emissions to air while shifting part of the environmental pressure to the marine environment.

Furthermore, policy instruments operate as part of a system rather than in isolation. For example, energy-efficiency requirements, restrictions on scrubber-water discharge, and the geographical scope of the EU ETS change the reference costs, emissions, and compliance options against which the renewable options are evaluated. These interactions differ between ships because fleet composition, routes, existing technologies, and access to alternative compliance options determine which ships are affected. The stakeholder analysis further shows that investment decisions depend on costs, policy stability, fuel availability, infrastructure, permitting, public acceptance, and coordination among actors.

The private investment case and the socio-economic case for emission reduction may lead to different conclusions. Using bio-methanol as a proxy for renewable fuel with substantial emission-reduction potential, private abatement costs exceed the 2023 EU ETS allowance price for almost the entire theoretical emission-reduction potential, whereas socio-economic abatement costs are lower than the estimated avoided external costs for a substantial share of the fleet. This indicates that environmental damage is not fully reflected in the costs faced by shipowners. However, the estimated societal benefits depend on how environmental damage is defined and valued. For example, lower social discount rates and longer assessment periods give greater weight to future climate and marine ecotoxicity impacts. The resulting differences do not reflect only uncertainty in the underlying data, but also judgements about which consequences to include and how impacts on present and future generations should be valued. Policy effectiveness must therefore be interpreted in relation to the stated environmental objectives, time perspective, and valuation approach.

Keywords: shipping, environmental policy instruments, policy evaluation, emission abatement, decarbonization, renewable marine fuels, external costs, marine ecotoxicity

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

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Paper II

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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.

Paper IV

Laurin, M. de O., Parsmo, R., Johansson, O., Grahn, M., Hansson, J., Kanchiralla, F. M., Lehtveer, M., & Brynolf, S. (2026). Is there a role for liquid fuels in future low-carbon energy systems? – A continuous interaction between global energy systems modeling and local energy port cluster participation. Applied Energy, 412, 127640. https://doi.org/https://doi.org/10.1016/j.apenergy.2026.127640

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Paper V

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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 ship-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

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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

H. Parsmo, R., Ytreberg, E., Brynolf, S., & Fridell, E. (2025). *External costs, policy instruments and efficient measures for sustainable shipping Final Report* [Scientific Report]. IVL Swedish Environmental Research Institute Ltd. https://ivl.diva-portal.org/smash/get/diva2:1960037/FULLTEXT01.pdf

I. Parsmo, R., Ytreberg, E., Fridell, E., Åström, S., Hassellöv, I.-M., & Brynolf, S. (2026). O-SLC-03 A socio-economic cost-benefit analysis assessing present and future European shipping scenarios. JRC. Proceedings of the 26th Transport and Pollution (TaP) Conference JRC Publications Repository. https://doi.org/10.2760/7863475

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Tack, tack!

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Abbreviations, acronyms, and concepts used in this thesis

Table 1. Abbreviations and acronyms

AIS	Automatic Identification System	MARPOL	International Convention for the Prevention of Pollution from Ships
AFIR	Alternative Fuels Infrastructure Regulation	MB	Market-based
ARP	Alpha RiskPoll model	MDO	Marine Distillate Oil
BC	Black Carbon	MEPC	Marine Environment Protection Committee
bio-MeOH	Bio Methanol	MGO	Marine Gas Oil
CBA	Cost-Benefit Analysis	MRV	Monitoring, Reporting, and Verification
CH_4	Methane	MSFD	Marine Strategy Framework Directive
CLRTAP	Convention on Long-Range Transboundary Air Pollution	N_2O	Nitrous Oxide
CO2	Carbon Dioxide	NECA	Nitrogen Oxide Emission Control Area
CO_2e	Carbon dioxide equivalent	NO_X	Nitrogen Oxides
CSI	Clean Shipping Index	OPS	Onshore Power Supply
EC	European Commission	PAHs	Polycyclic Aromatic Hydrocarbons
ECA	Emission Control Area	PEMFC	Proton Exchange Membrane Fuel Cell
EEA	European Environment Agency	PM	Particulate Matter
EEDI	Energy Efficiency Design Index	PSSA	Particularly Sensitive Sea Area
e-LH_2	Electrolytic Liquid Hydrogen	RED	Renewable Energy Directive
e-MeOH	Electrolytic Methanol	RME	Rapeseed Methyl Ester
EMEP	European Monitoring and Evaluation Programme	RoPax	Roll-on/Roll-off passenger
EMSA	European Maritime Safety Agency	RoRo	Roll-on/Roll-off
EP	European Parliament	RQ	Research Question
ESI	Environmental Ship Index	SCC	Social Cost of Carbon
ETS	Emission Trading System	SCR	Selective Catalytic Reduction
EU	European Union	SECA	Sulfur Emission Control Area
EUCO	Council of the European Union	SFOC	Specific Fuel Oil Consumption
FRC	Foul-release coating	SLCF	Short-Lived Climate Forcer
GES	Good Environmental Status	SMA	Swedish Maritime Administration
GET	Global Energy Transition model	SoP	Socio-economic Perspective
IPCC	Intergovernmental Panel on Climate Change	PoP	Private-economic Perspective
GHG	Greenhouse Gas	SO_X	Sulfur Oxides
GT	Gross Tonnage	STEAM	Ship Traffic Emission Assessment Model
GWP	Global Warming Potential	TBT	Tributyltin
HFO	Heavy Fuel Oil	TIS	Technological Innovation System
HSFO	High Sulfur Fuel Oil	UN	United Nations
HVO	Hydrotreated Vegetable Oil (Biodiesel)	VLSFO	Very Low Sulfur Fuel Oil
IMO	International Maritime Organization	VSL	Value of a Statistical Life
LCA	Life Cycle Analysis	VOLY	Value of a Life Year
LCIA	Life Cycle Impact Assessment	WTP	Willingness to Pay
LMG	Liquefied methane gas. The term LMG is used instead of the more common LNG (liquefied natural gas) to also include bio-LMG and e-LMG.	WTW	Well-to-Wake
LRTAP	Long-Range Transboundary Air Pollution		

1. Introduction

Globally, shipping is the mode of transport that moves the largest quantities of goods over long distances and accounts for more than 70% of worlds total freight transport work, measured in tonne-kilometers (International Transport Forum, 2023). While shipping is essential for global trade, its extensive operations have environmental impacts globally (Faber et al., 2020; L. Johansson et al., 2017), regionally (Jalkanen et al., 2016; Ytreberg et al., 2021), and locally (Ducruet et al., 2024; Lunde Hermansson et al., 2023). These impacts include contributing to global warming, causing adverse health effects, and leading to acidification, eutrophication, and ecotoxicity due to emissions to air and water (Åström et al., 2018; Faber et al., 2020; Ytreberg et al., 2021). When these impacts are not reflected in the market price of shipping services, they constitute environmental externalities and may give rise to external costs.

Environmental policy instruments are introduced to address such externalities and to support the achievement of environmental objectives. In shipping, they include mandatory emission standards (IMO, 2023; MEPC, 2021; Topali & Psaraftis, 2019), energy-efficiency standards (MEPC, 2021), tradable permits (Christodoulou & Cullinane, 2024; EP & EUCO, 2023a; Flodén et al., 2024), environmental differentiation of fees (Christodoulou et al., 2019; Lindé et al., 2019; Sköld, 2019; Vierth & Johansson, 2020), fuel-intensity requirements (Christodoulou & Cullinane, 2022; EP & EUCO, 2023d; Transport & Environment, 2023), regulations targeting technology (BIMCO, 2024), and practice-specific requirements or restrictions (Alzieu, 1991; EP & EUCO, 2023c; IMO, 2001; MEPC, 2014). These policy instruments operate at different geographical levels and address different emissions, ships, and activities. They also differ in how they influence shipowners. Some directly prescribe or even enforce the abatement strategies that can or must be used, while others change the relative costs of fuels, technologies, or operational measures. Their environmental effects therefore depend not only on whether a regulation is in place, but also on how its design influences the choices available to shipowners. These mechanisms and interactions make the evaluation of environmental policy instruments for shipping a **multi-layered task**, in which conclusions at one analytical level depend on assumptions and outcomes at other levels.

Studies of individual policy instruments can partly identify how a specific regulation changes compliance requirements, relative costs, and incentives for particular abatement strategies (Åström et al., 2018; Christodoulou & Cullinane, 2022, 2024; Vierth & Johansson, 2020). However, the response to an individual instrument does not always occur in isolation. The feasibility and attractiveness of different compliance options can depend on other policy instruments (Christodoulou et al., 2019), fuel and infrastructure availability (Malmgren et al., 2023), technological characteristics, and the decisions of actors across the maritime and energy systems at different governance levels (Acciaro et al., 2014; de Oliveira et al., 2026; Gritsenko, 2017; Ostrom, 2017). They may also affect other types of emissions than those in compliance (Ytreberg et al., 2021). Consequently, an instrument that appears effective (or ineffective) when assessed separately may produce different incentives or environmental outcomes when considered as part of a larger policy system.

Furthermore, although abatement costs and private investment conditions indicate whether a policy is likely to influence shipowners' decisions, they do not establish whether the resulting choice is beneficial from a socio-economic perspective (Åström, 2023; P. O. Johansson & Kriström, 2018). Such an assessment also requires consideration of environmental impacts and external

costs across different emission pathways (de Vries et al., 2024; M. A. Huijbregts et al., 2017; Ytreberg et al., 2021). However, conclusions from these assessments depend on uncertain data and causal relationships (Frey et al., 2006; Klinke & Renn, 2002), as well as on choices concerning system boundaries (Baumann & Tillman, 2004), time horizons, monetary valuation (Aldred, 2006; Craswell, 1998; Reiff, 2014), and the environmental objectives used to define effectiveness (EU, 2026; Meadows, 2015; Niiniluoto, 1993; UN, 2015). Variation between assessment results may therefore arise from incomplete knowledge about the system, but also from differences in how evidence, objectives, and evaluative criteria are interpreted. Making these sources of variation explicit is necessary for understanding why assessments of the same policy or abatement strategy may lead to different conclusions.

An evaluation of environmental policy instruments for shipping through three related perspectives is therefore conducted in this thesis. 1). It examines how the design of individual policy instruments influences shipowners' incentives and choices among abatement strategies. 2). It investigates how the effects of policy instruments are shaped by interactions among policies, abatement strategies, and actors. 3). It examines how socio-economic costs, environmental impacts, uncertainty, and evaluative assumptions influence the interpretation of policy effectiveness.

1.1. Research aim

The overall aim of this thesis is to evaluate how environmental policy instruments influence technological choices, abatement investments, and emissions in the shipping sector. The analysis considers both the private costs faced by shipowners and the external costs imposed on society. It also aims to determine whether these policies effectively reduce emissions and environmental impact and if the design could be improved. The thesis addresses the following research questions (RQ):

RQ1 Individual policy-instrument perspective: *How does the current design of individual environmental policy instruments impact shipping companies incentives to invest in technical and operational measures to reduce emissions to air and water?*

This RQ is addressed through analysis of four national and international environmental policy instruments and how they affect shipowners' investment decisions in alternative abatement strategies.

- **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 evaluation of the inclusion of shipping in the **EU ETS**. This case study assesses how the EU ETS, encourages shipping companies to invest in fuels and energy carriers that reduce GHG emissions (**Paper II**). The **evaluation** extends **Paper II** by incorporating updated cost assumptions.
- **Policy 3**: **FuelEU Maritime** is analyzed in **Paper V**.
- **Policy 4:** 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 Policy-system perspective: *How do interactions among environmental policy instruments, abatement strategies and different stakeholders influence investment decisions and environmental outcomes in shipping?*

For this question interactions among environmental policy instruments influence investment decisions, emissions to air and water, and associated environmental impacts in the shipping sector (**Paper II**, **Paper III**, **Paper IV** and **Paper V)** are investigated. Understanding future policy pathways requires integrating stakeholder perspectives and system-level constraints, which cannot be captured by an evaluation based solely on abatement costs or private investment costs.

In addition, this is assessed through an extended analyses to evaluate the interactions and unintended consequences of policy combinations on shipowners' investment decisions. The analysis focuses on the European fleet under global and regional policies (based on work in **Paper II** and **Paper III**).

RQ3 Perspective on policy evaluation: *How does the inclusion of societal perspectives, including external cost and considerations of uncertainty and ambiguity shape the interpretation of policy effectiveness?*

This research question extends the analyses conducted in **RQ1** and **RQ2** by moving past investment decisions and emissions to consider the associated environmental impacts and external costs from a societal perspective. **RQ3** integrates a societal perspective into the evaluation of policy effectiveness by comparing shipowner-focused cost assessments with external cost estimates, particularly in **Paper III** and **Paper V**. The analysis applies distinctions between private and societal perspectives, descriptive and normative elements of policy evaluation, and uncertainty and ambiguity to examine how calculation assumptions, valuation choices, system boundaries, and alternative interpretations of effectiveness influence the conclusions. **RQ3** thereby examines why different evaluation perspectives may lead to different conclusions about policy effectiveness and preferred abatement strategies.

1.2. Scope and delimitations

The studies differ in geographical coverage, regulated emissions, ship types, operational scope, and time horizon. Their empirical basis ranges from model ships and ships covered by EU MRV to the global scrubber-fitted fleet and a Swedish energy-port cluster. The results are therefore compared mainly in terms of policy mechanisms and are not assumed to represent all ships, companies, or geographical contexts. The thesis mainly examines how policy design affects costs, incentives, and compliance choices. Most analyses do not establish that a policy caused a specific investment or emission reduction. Modelled outcomes and scenarios are therefore conditional on the technical, economic, and operational assumptions applied.

The environmental assessments focus mainly on emissions from ship operation. Upstream emissions are included when important for comparisons between energy carriers, particularly for climate impacts, but the thesis does not provide a complete life-cycle assessment. Fuel prices, technology costs, and transport demand are generally treated as external assumptions, and feedback between shipping and the energy system are not fully modelled. The socio-economic assessments are limited to environmental impacts for which suitable data and monetary values are available.

1.3. Structure

The thesis is structured to move from the context of environmental policy instruments in shipping, to the concepts used to evaluate them, and then to the empirical analyses. Chapter 2 introduces the shipping sector, its environmental impacts, the policy objectives, the policy instruments examined, and the abatement strategies available to shipowners. This chapter defines the empirical and regulatory context of the thesis. Chapter 3 presents the theoretical concepts used to interpret this context, including policy evaluation, policy interactions, stakeholder participation, external costs, uncertainty, ambiguity, and normative assumptions. Chapter 4 describes the material and methods used in the five appended papers and explains how these studies relate to the three research questions. Chapter 5 presents and discusses the results through the same three perspectives: first, how individual policy instruments influence shipowners' incentives and abatement choices; second, how these effects are shaped by interactions within a broader policy system; and third, how societal costs, uncertainty, and evaluative assumptions affect the interpretation of policy effectiveness. Chapter 6 concludes the thesis by discussing their implications for the evaluation and design of environmental policy instruments for shipping.

2. Shipping, emissions and abatement strategies

This chapter summarizes the sectoral, environmental, regulatory, and technical background needed for the thesis analyses. It covers shipping activity and fleet characteristics, emissions to air and water, current policy instruments, and abatement strategies for greenhouse gases, air pollutants, and waterborne emissions.

2.1. Shipping, trade, and transport work

Shipping enables large quantities of goods to be transported over long distances and is therefore closely connected to international trade. However, the magnitude of trade can be expressed in different ways. Trade statistics may refer to the monetary value of traded goods, their physical mass or volume, or the amount of transport activity required to move them. These measures describe related but different aspects of trade and should therefore not be used interchangeably. The physical scale of extra-EU maritime trade is illustrated in Figure 1, which shows gross weights of goods handled in EU main ports in trade with major non-EU partner countries. In total, approximately 1.43 billion tonnes of goods transported to or from destinations outside the EU were handled in EU main ports in 2024 (Eurostat, 2026).

Differences between reported shares of seaborn trade demand may reflect not only different years and data sources, but also different definitions and measurement units, such as monetary value, mass, volume, or another measure of trade. A frequently cited statement is that approximately 80 to 90% of international trade is carried by sea, e.g. Balcombe et al (2019). However, it is not clear how these estimates were derived or what is included in the denominator. The value of approximately 80% may originate from UNCTAD statistics referring to the share of international trade measured by physical volume, although the precise definition of “volume” and the underlying calculation require further verification (UNCTAD, 2025). The upper estimate of 90% appears to be associated with IMO Maritime knowledge center, which states that “it is generally accepted that more than 90 percent of global trade is carried by sea” (Maritime Knowledge Centre, 2009).

For environmental transport analysis, transport work provides a more explicit measure of physical transport activity because it accounts for both the quantity transported and the distance over which it is moved. Freight transport work is calculated as cargo mass multiplied by transport distance and is commonly expressed in tonne-kilometres (tonne-km) or tonne-nautical miles. One nautical mile corresponds to 1.852 km. Passenger transport work is defined correspondingly as the number of passengers multiplied by distance and is commonly expressed in passenger-kilometres. Transport work is related to transport demand, but the terms are not always synonymous because transport demand can also refer more broadly to the underlying need or demand for transport services. According to the International Transport Forum (2023), maritime shipping accounts for approximately 70% of global freight transport work measured in tonne-kilometres. This estimate compares maritime shipping with freight transport by road, rail, inland waterways, and air.

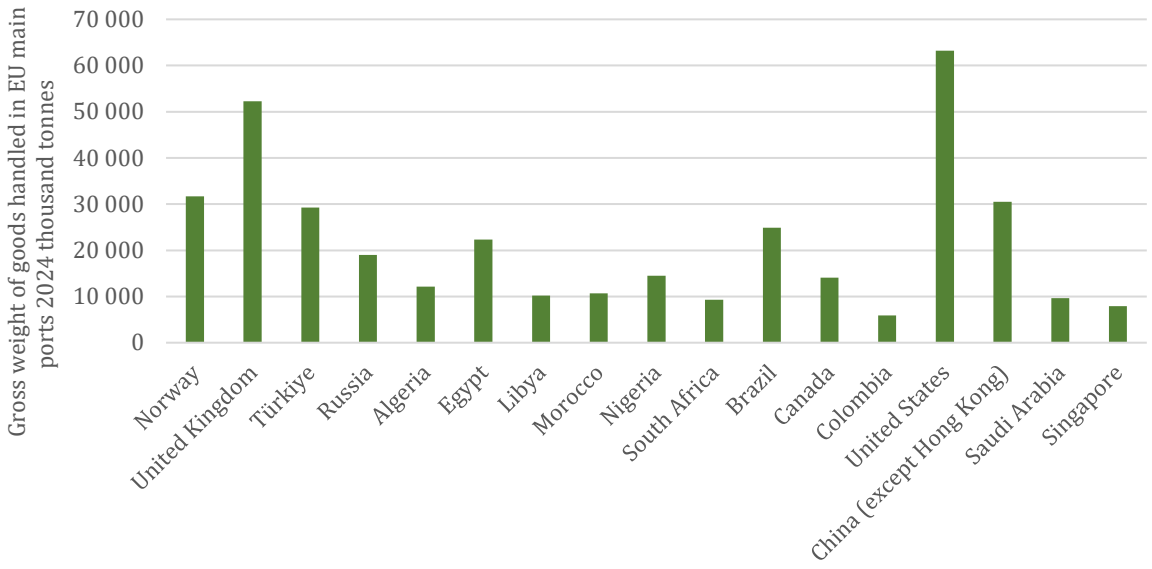


Figure 1. Gross weight of goods handled in EU main ports in trade with selected extra-EU partner countries in 2024. The figure illustrates the relative importance of major non-EU partner countries in maritime freight flows to and from the EU, measured as the total gross weight handled in EU main ports. Source: Eurostat (2026).

2.1.1. 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, LG tanker, Roll-on/Roll-off (RoRo), vehicle carrier, Roll-on/Roll-off passenger (RoPax), reefer, other and cruise (Faber et al., 2020).

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 in Sweden 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 governments like for example public procurement (Bach et al., 2020), or apply directly to ships owned and operated by the public sector, like to the Swedish Maritime Administrations (SMA) shipping fleet (SMA, 2023a). This allows certain segments, such as road ferries, icebreakers, or pilot ships in Sweden, to be regulated directly.

2.1.2. Ship age and replacement rate

Ships are long-lived assets, and assumptions about fleet age and replacement rates influence assessments of how rapidly new technologies can enter the fleet. The Fourth IMO GHG Study applies an average replacement age of 25 years across ship types, but fleet and decommissioning data indicate considerable variation between ship segments and size classes (Parsmo et al., 2026). Data for ships currently operating in the EU (MRV scope) show differences in age structure across segments (THETIS-MRV, 2023), while Seaweb data of ships decommissioned globally between 2014 and 2023 indicate that many ships remained in service for more than 25 years (S&P Global,

2023). Smaller ships were generally decommissioned at a higher age than larger ships within several cargo-ship segments, although the observed age at decommissioning should not be interpreted as a fixed technical lifetime (Parsmo et al., 2026).

2.2. Environmental impacts of shipping

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, 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 2.9% of all anthropogenic emissions globally (Faber et al., 2020). Greenhouse gas emissions from the global fleet are dominated by carbon dioxide (CO_2), although methane (CH_4) and nitrous oxide (N_2O) are also emitted during fuel combustion. Methane slip, in particular, can be a relevant fraction of the CO_{2e} when LNG is used as fuel (Brynolf et al., 2014; Malmgren, 2023). In addition to these climate gases, shipping emits short-lived climate forcers (SLCFs), such as sulfur oxides (SO_X), nitrogen oxides (NO_X), and black carbon (BC). These emissions affect the climate through several atmospheric processes, and their net climate impact is associated with considerable uncertainty (Åström & Källmark, 2023; Szopa et al., 2023).

During combustion, several air pollutants are formed, SO_X, NO_X and particulate matter (PM). These emissions contribute to increased health risks, such as effects on the airways and lungs (Åström et al., 2018, 2019), 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 contribution of shipping to overall air pollutant emissions is substantial. 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).

Shipping also releases a wide variety of hazardous substances into the marine environment, including metals and organic compounds such as polycyclic aromatic hydrocarbons (PAHs) (Jalkanen et al., 2021; L. Johansson et al., 2020). These substances may have an impact on the marine ecosystem by affecting marine species at different life stages, leading, for instance, to malformations, diminished reproductive capacity, or increased mortality (Lunde Hermansson et al., 2026). In Life Cycle Impact Assessment (LCIA), the potential impacts of hazardous substances released into the marine environment are typically assessed through the impact category marine ecotoxicity (Goedkoop et al., 2009; M. A. Huijbregts et al., 2016). 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 marine organisms, ambient environmental concentrations, and the degree of dilution it undergoes (Dong et al., 2016; Funke et al., 2017; Huijbregts et al., 2016, 2017; Owsianiak et al., 2023). In some regions, the load of hazardous substances from shipping can be considerable 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, 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).

2.3. Policy objectives

The environmental impacts examined in this thesis connect to objectives formulated at global, regional (European), and national (Sweden) levels. At the global level, the United Nations Sustainable Development Goals provide broad objectives concerning climate change, human health, and the marine environment (UN, 2015). Corresponding objectives are expressed through EU environmental and climate policies (EU, 2026), and at national level through the Swedish environmental quality objectives (Swedish Environmental Protection Agency, 1999). More specific frameworks, including the IMO GHG Strategy (IMO, 2023), the Convention on Long-Range Transboundary Air Pollution (EC, 2019), and the EU Marine Strategy Framework Directive (EC, 2008), translate parts of these broader objectives into targets and commitments that are more directly relevant to shipping emissions. Table 2 summarizes the objectives most closely related to the environmental impacts examined in this thesis.

Table 2. Environmental objectives relevant to the impact categories examined in this thesis.

Environmental area	UN Sustainable Development Goals	EU level objectives	Swedish environmental quality objectives	Shipping-relevant framework
Climate change	SDG 13: Climate Action	Climate neutrality	Reduced Climate Impact	IMO objective of net-zero GHG emissions from international shipping by or around 2050
Air pollution and human health	SDG 3: Good Health and Well-being; SDG 11: Sustainable Cities and Communities	A high level of environmental protection and improved air quality	Clean Air; Natural Acidification Only; Zero Eutrophication	CLRTAP and the Gothenburg Protocol; MARPOL Annex VI
Marine pollution and ecosystem quality	SDG 14: Life Below Water	Protection of the marine environment and achievement of Good Environmental Status (GES)	A Balanced Marine Environment; Flourishing Coastal Areas and Archipelagos; A Non-Toxic Environment; Zero Eutrophication	EU Marine Strategy Framework Directive; EU Water Framework Directive; relevant IMO conventions

2.4. Environmental policy instruments in shipping

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. Environmental policy instruments for ships can vary much 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 (Swedish EPA, 2024; Windmark, 2019; Windmark et al., 2017), 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, 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.

2.4.1. 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, 2014, 2021, 2024), 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 developed and amended by the IMO and is enforced by flag states and port states through national legislation and inspection regimes.

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 & Psaraftis, 2019), as shown in Figure 2. 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 Since May 1, 2025, also in the Mediterranean Sea.

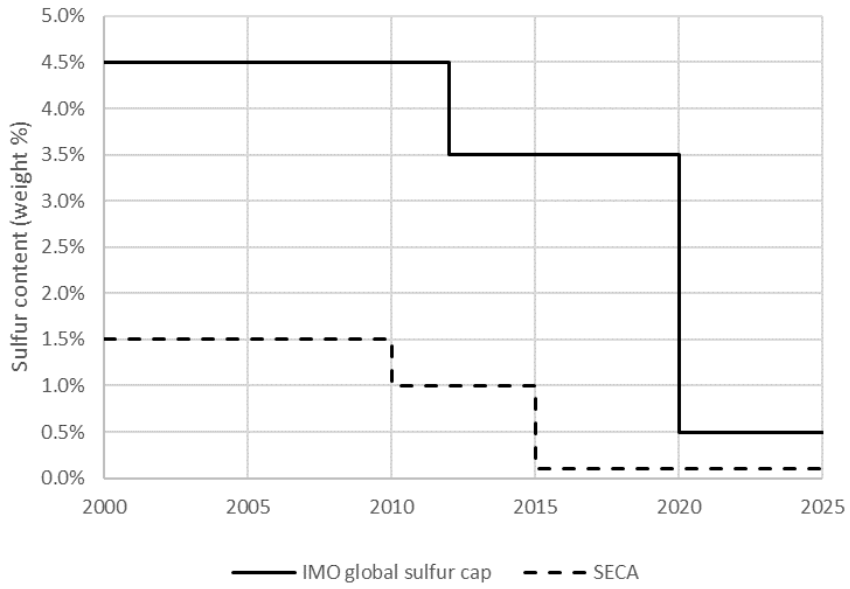


Figure 2. Sulfur limits of marine fuels for the period 2000-2025.

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 3. 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).

[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.

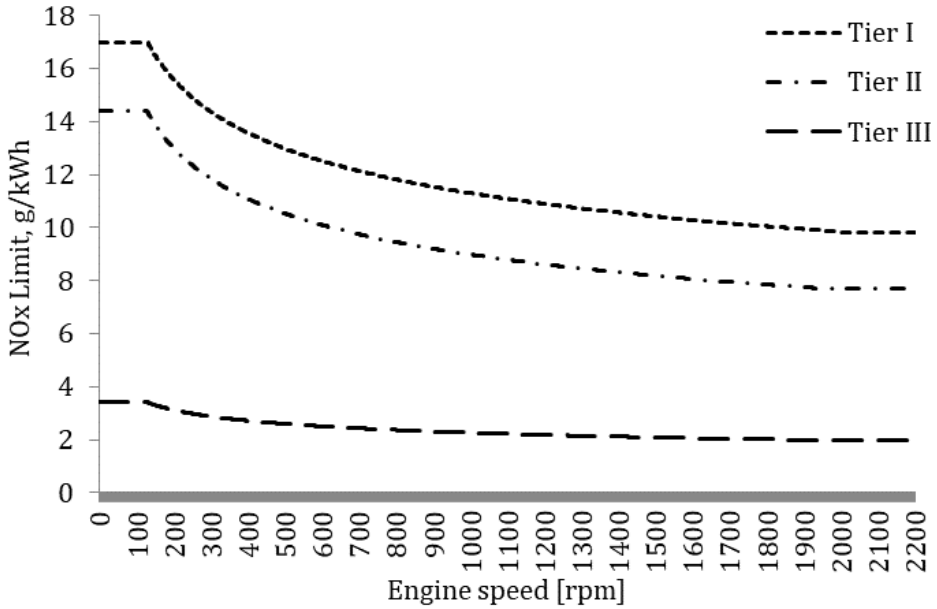


Figure 3. NO_X emission standards for marine engines in international shipping, varying by engine speed (rpm) and the ship's construction year.

2.4.2. European regulations: Inclusion in EUs Emission trading scheme (EU ETS) and FuelEU Maritime

Recently, several EU regulations have been adopted to reduce maritime GHG emissions (EP & EUCO, 2023d, 2023c, 2023b, 2023a). Shipping was added to the EU ETS in January 2024, as part of the European Green Deal (Christodoulou et al., 2021; Christodoulou & Cullinane, 2024). The EU ETS is an emissions trading scheme in which actors from many sectors must either purchase or are allocated allowances to be permitted to emit CO_2, the historic prices of allowances are illustrated in Figure 4. Established in 2005, the EU ETS currently includes direct CO_2 emissions and will also cover N_2O and CH_4 from 2026 (EP & 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.

In addition to the inclusion of shipping in the EU ETS, the EU has introduced FuelEU Maritime and the Alternative Fuels Infrastructure Regulation (AFIR) (EP & EUCO, 2023c, 2023d). FuelEU Maritime applies from 2025 and sets gradually stricter limits on the yearly average greenhouse gas intensity of energy used by ships above 5 000 GT calling at EEA ports. Unlike the EU ETS, which prices emissions, FuelEU Maritime targets the fuel and energy mix itself by requiring reductions in well-to-wake GHG intensity, including CO_2, CH_4, and N_2O (EP & EUCO, 2023d). The required reduction started at 2% in 2025 and increases gradually towards 80% by 2050, compared with a 2020 reference value, see Table 3. The regulation allows ship operators to comply through different combinations of fuels, energy-efficiency measures, and onboard technologies, provided that the required GHG intensity is achieved.

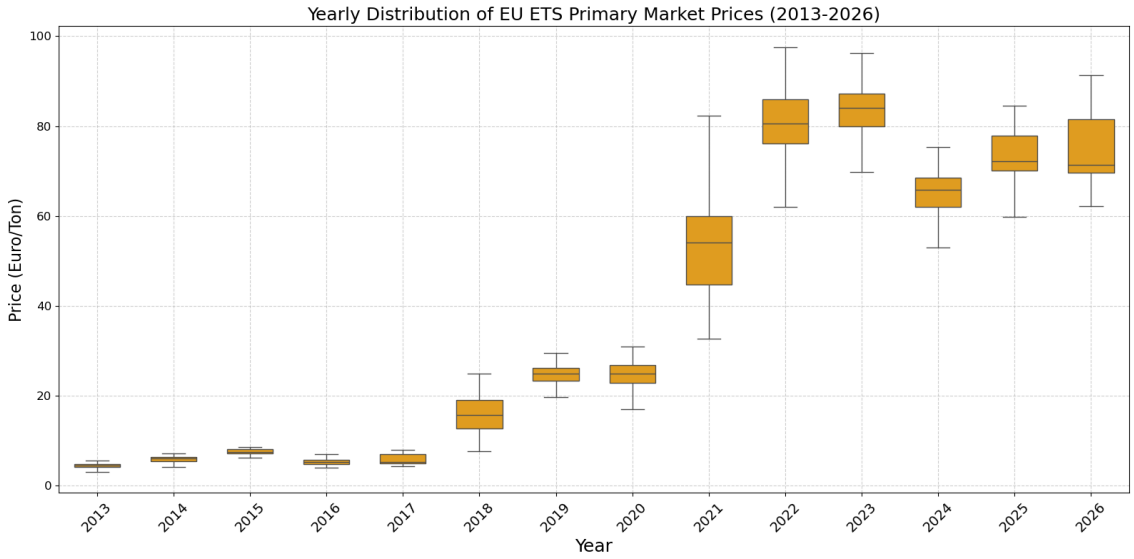


Figure 4. Boxplot illustrating daily historic EU ETS allowance price (ICAP, 2026). For 2026 only data for January-June are presented.

Table 3. FuelEU Maritime reductions target and corresponding GHG intensity

Year	Required reduction	Calculated maximum GHG intensity
2025	2.0%	89.3 gCO_{2e}/MJ
2030	6.0%	85.7 gCO_{2e}/MJ
2035	14.5%	77.9 gCO_{2e}/MJ
2040	31.0%	62.9 gCO_{2e}/MJ
2045	62.0%	34.6 gCO_{2e}/MJ
2050	80.0%	18.2 gCO_2e/MJ

FuelEU Maritime is based on a yearly compliance balance for each ship. The balance is calculated by comparing the ship's actual yearly average well-to-wake GHG intensity with the regulatory limit for that year and multiplying the difference by the amount of energy used within the scope of the regulation. A ship with a lower GHG intensity than the limit generates a compliance surplus, while a ship above the limit generates a deficit. The regulation also includes so called "flexibility mechanisms": a surplus can be saved for next year or pooled with other ships, while a deficit can be compensated through "pooling", limited borrowing from the following year, or by paying a penalty. Pooling means that ships are allowed to comply collectively rather than individually. For example, one ship using low-GHG fuels or electricity may generate a surplus that can offset the deficit of another ship using more GHG-intensive fuels. In practice, this creates a quota-like mechanism in which use of renewable fuels can acquire economic value within a fleet or between actors.

The denominator in FuelEU Maritime is the total energy used onboard within the regulatory scope, expressed in MJ. For fuels, this is calculated from the mass of fuel consumed multiplied by its lower calorific value. For electricity it applies to all electricity in MJ supplied from shore to ship. The resulting well-to-wake GHG intensity is therefore an energy-weighted average of all fuels and energy carriers used during the reporting year. However, following the reasoning outlined in Recital 44 of Regulation (EU) 2023/1805, electricity supplied through onshore power supply

(OPS) it is recommended to set the GHG intensity to zero (EP & EUCO, 2023d), reflecting its role in reducing emissions and air pollution in port. If a compliance shortage remains after the flexibility mechanisms have been applied, the company must pay a FuelEU penalty. The penalty is calculated from the remaining deficit and expressed in tonnes of VLSFO-equivalent energy. The penalty is set at 2 400 euro per tonne of VLSFO equivalent see appendix IV in EP & EUCO (2023d), with higher effective penalties possible in cases of repeated non-compliance.

AFIR complements FuelEU Maritime by addressing the infrastructure side (EP & EUCO, 2023c). For shipping, its most relevant requirements are about the deployment of onshore power supply (OPS) in major EU ports. This infrastructure requirement is linked to the FuelEU Maritime obligation for container and passenger ships to use OPS or equivalent zero-emission technologies while at berth. From 2030, this applies in ports covered by AFIR, and from 2035 in EU ports where OPS is available.

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 (EC, 2016, 2017, 2023a, 2023b; EP & EUCO, 2015, 2024). 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 5 (THETIS-MRV, 2023), which illustrates that emissions vary much across different ship segments, with container ships being the largest contributors.

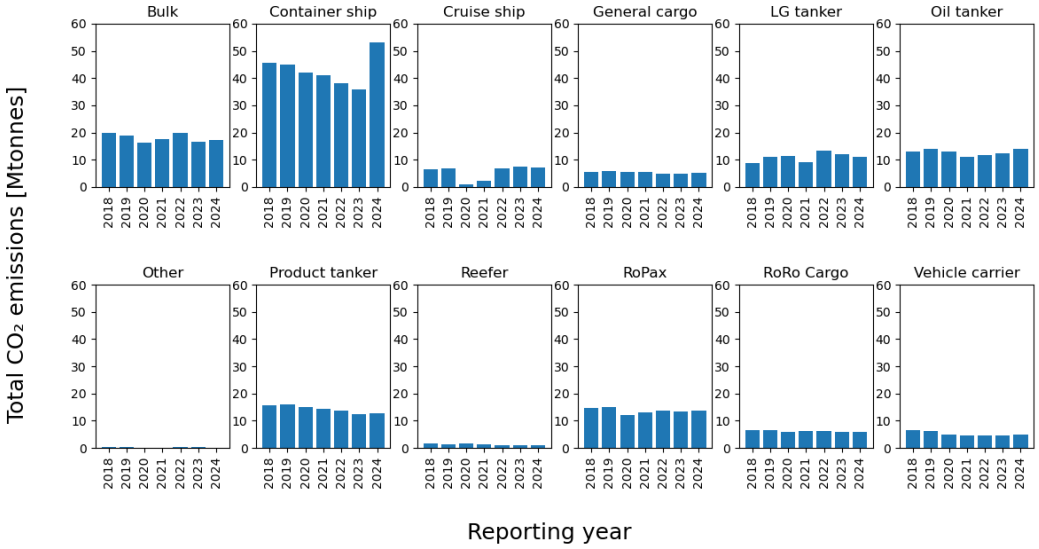


Figure 5. Total CO_2 emissions for the EU fleet (2018-2023) across 12 different ship segments, based on processed data from (THETIS-MRV, 2023). The figure illustrates the emission scope in Europe. The ship category 'Other' is currently not included in the EU ETS scope. However, some ships in this category have reported to MRV and are therefore included in this figure.

2.4.3. Environmentally differentiated port and fairway fees in Sweden

In Sweden, environmentally differentiated port and fairway fees are in place and has been in place since 1998, providing rebates based on ships' environmental performance (Lindé et al., 2019; Sköld, 2019; Vierth & Johansson, 2020). 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, 2016, 2017). 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).

2.5. 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.

2.5.1. Abatement strategies to reduce emissions of air pollutants

Shipping contributes substantially 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 (to air) 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 (Jalkanen et al., 2021; Lunde Hermansson et al., 2021, 2024). 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 (Fridell et al., 2020; Karjalainen et al., 2022; Winnes et al., 2020). 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 abatement strategy 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; Parsmo et al., 2024; Vaishnav et al., 2016). 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; Parsmo et al., 2024; Vaishnav et al., 2016).

2.5.2. 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. 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 (Berndes et al., 2016; Broch et al., 2013; Cowie et al., 2017; Reid et al., 2020; Searchinger et al., 2018; Sparovek et al., 2016). In the EU, sustainability criteria are addressed through the Renewable Energy Directive (EP & EUCO, 2018, 2023b). Several options exist for using biofuels in shipping, including hydrotreated vegetable oil (HVO), rapeseed methyl ester (RME), liquefied biomethane gas (bio-LMG), and bio-methanol (bio-MeOH) (Bach et al., 2021; Kanchiralla et al., 2023; Mukherjee et al., 2023). Biodiesel, such as HVO or RME, can be used in diesel engines either exclusively or blended with fossil diesel. Similarly, bio-LMG is compatible with ships using LMG 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 (Jivén et al., 2023; Kanchiralla, 2023; Kanchiralla et al., 2023). Hydrogen is produced using electrolysis, where electricity splits water into hydrogen, oxygen, and excess heat. The environmental performance of hydrogen therefore strongly 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 ship 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 LMG, 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) (Grahn et al., 2022; Kanchiralla, 2023; Styhre et al., 2024). 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 capture 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 ships currently have installed (S&P Global, 2023). These engines require the injection of pilot fuel, such as MGO.

Apart from the electricity price, the electrolyzers investment cost is one of the main cost components in electrofuel-production. There appears to be a gap between near-term electrolyzers cost estimates in the literature, which range from 420-1300 €$_{2025}$/kW (Korberg et al. 2021, Grahn et al. 2022, Styhre et al. 2024), and reported costs of approximately 2100-2200 €$_{2025}$/kW for current projects in Europe (European Hydrogen Observatory, 2026). These values may not be directly comparable, as the reported costs may include different components, such as the electrolyzers stack, balance of plant, power electronics, utilities, and installation. The difference may also reflect regional cost differences and optimistic assumptions about future cost reductions. The stack accounts for much of the potential cost reduction, while several other system components are already more mature and may therefore have less scope for further cost reductions. Krishnan et al. (2023) estimate that more than 73 billion euro (up to 2 300 billion euro) in cumulative investment would be required to reduce electrolyzers system costs to below 570 €$_{2023}$/kW.

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) (Jivén et al., 2020; Kanchiralla, 2023; Kanchiralla et al., 2023; Kersey et al., 2022; Kistner et al., 2024; Link et al., 2024; MAN, 2019; Stena Rederi Technical Division, 2018; WSDOT, 2020). 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). Battery systems for maritime applications tend to have higher costs per kWh than batteries used in road vehicles. However, comparisons between reported values are complicated by differences in technical assumptions and system boundaries. An important distinction is whether the reported cost e.g. refers to battery cells, a battery pack, or a complete containerized battery system. A containerized system may be the most relevant measure for ships, particularly “plug-in hybrid” ships, because it may include e.g. cooling and safety systems, power electronics, and integration into the ship. Considering these components and the high-power requirements of maritime applications, 370-500 €/kWh appears to represent a reasonable range of the current cost (Kanchiralla et al., 2025; Parsmo et al., 2024). Battery system costs are also influenced by the relationship between power and energy requirements, implying that systems with high power requirements may have a higher cost per unit of installed energy than systems designed primarily for energy storage. Another important aspect is that the selected design of the depth of discharge determines how much of the installed capacity is considered usable and therefore influences both the required installed capacity and the total system cost.

2.5.3. Abatement strategies to reduce emissions to water

Hazardous substances released into the sea originate from diverse ship activities, such as leakage of biocides from antifouling paints (Lagerström et al., 2022) and the discharge of scrubber water (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. By reducing biofouling, these coatings decrease hydrodynamic resistances, and fuel consumption while also minimizing the risk of spreading 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, such as silicone-based foul-release coatings (FRC) (Kim et al., 2025; Ytreberg et al., 2026). These coatings have recently gained more attention due to their ability to reduce water resistance without relying on harmful biocides (L. Johansson et al., 2020; Oliveira et al., 2022). Silicone coatings may also reduce fuel consumption and GHG emissions by improving ship hydrodynamics, though they usually come with higher initial application costs and specific maintenance requirements (Lagerström et al., 2022).

As mentioned in section 2.5.1, scrubbers produce wastewater that contains different hazardous substances, including metals and PAHs (Lunde Hermansson et al., 2021, 2023; Ytreberg et al., 2022). This scrubber washwater effluent can either be discharged directly into the sea in an open-loop system or treated in a closed-loop system, in which the resulting sludge can be offloaded to onshore facilities (Lunde Hermansson, 2024). However, despite being referred to as "closed-loop," some scrubber washwater effluent is still released into the ocean, although to a lesser extent. In a closed-loop system about 0.45 m^3/MWh_{out} is released compared to 90 (71.8–108) m^3/MWh_{out} for open-loop systems (Ytreberg et al., 2021). This closed-loop scrubber washwater effluent 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, enabling switching between open- and closed-loop operation 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 large space required. Another approach to avoiding scrubber-water discharge is to use more expensive low- or zero-sulfur fuels, thereby eliminating the need for a scrubber.

3. Theoretical framework

This chapter presents the concepts used to interpret the thesis results. The main point is that policy effectiveness cannot be assessed only from measured emissions and costs, but also depends on policy scope, interactions, system conditions, stakeholder perspectives, valuation choices, uncertainty, ambiguity, and normative assumptions.

3.1. Evaluating environmental policy instruments

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 (Christodoulou et al., 2019; Sterner & Coria, 2012). Command-and-control regulations involve 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, as the example used in Table 4, 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 4 categorize different implemented policy instruments for shipping relevant to this thesis.

Table 4. Example of categorization of relevant policy instruments for shipping

Example of taxonomy of policy instruments	Categorization used in this thesis	Examples	Source
Global emission standards	Command-and-control	NO_X (Tier) and IMO global sulfur cap	(MEPC, 2024, MEPC, 2014, MEPC, 2021)
Bans	Command-and-control	Tributyltin (TBT) in antifouling paint	(IMO, 2001)
Tradable emission permits	Market-based quantity	Inclusion of ships in EU ETS	(EP and EUCO, 2023a)
Tradable quotas	Market-based quantity	FuelEU Maritime	(EP and EUCO, 2023d)
Refundable emissions payments	Market-based price	Norwegian NO_X fund	(Parsmo et al., 2017)
Detailed regulation	Command-and-control	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)

Environmental policy instruments are commonly evaluated in terms of effectiveness and cost-effectiveness (EEA, 2001; Sterner & Coria, 2012; Vaz et al., 2001). Effectiveness refers to whether a policy instrument achieves its intended environmental objective, for example by reducing targeted emissions or environmental impacts. Cost-effectiveness refers to whether this reduction is achieved at relatively low cost, or at lower cost than alternative measures. A policy instrument can therefore be cost-effective per unit of reduction while still having limited overall effectiveness if the total reduction achieved is small. In applied environmental policy analysis, cost-effectiveness is often assessed using abatement costs (Einride, 2024). Abatement cost is generally defined as the cost of an abatement strategy per unit of environmental improvement achieved, such as per tonne of emissions reduced or per unit of avoided environmental impact.

Evaluating environmental policy instruments also involves considering whose costs and benefits are included in the assessment. This thesis distinguishes between a private-economic perspective (PoP) and a socio-economic perspective (SoP). The PoP considers costs and benefits from the shipowner's or operator's point of view and provides a basis for assessing investment incentives and the financial attractiveness of abatement strategies. The SoP considers costs and benefits to society, including environmental externalities, and provides a basis for assessing whether the benefits of an abatement strategy outweigh its costs. This distinction is adopted to structure the analysis in this thesis, while other ways of framing costs and/or benefits are also possible.

3.2. Policy interactions and system perspective

Evaluating policy instruments solely by observing improved environmental outcomes (effectiveness) and achieving the lowest possible abatement cost (cost-effectiveness/efficiency) in the short term is not always ideal, especially when assessing innovation. Costs can change with increased production (Jacobsson & Bergek, 2011; Krishnan et al., 2023), 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 the TIS framework, highlighting that policies or niche environments are critical for encouraging the growth of alternative solutions. 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. Central 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 ships (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 (IMO, 2001). 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 (Gritsenko, 2017; IMO, 2001; MEPC, 2014), which may be very effective when enforced. However, this type of regulation presents challenges further than reaching agreement and being time-consuming to implement, command-and-control regulations are also difficult to design in ways

that initially allow for learning on a smaller scale (Gritsenko, 2017; Sterner & Coria, 2012). 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."

3.3. Stakeholder participation

Participatory approaches are based on the "premise" that assessment and decision-making can be improved by integrating stakeholder knowledge, experiences, values, and perspectives alongside scientific expertise (Salter et al., 2010). Participation may range from consultation to collaborative co-production and can occur throughout different stages of the assessment process (Salter et al., 2010). Both for example McGookin et al. (2021) and Stalpers et al. (2008) argue that meaningful participation requires stakeholders to have the opportunity to influence the assessment rather than only provide information, thereby enhancing the relevance, legitimacy, and robustness of the results.

McGookin et al. (2021) conducted a systematic review of 59 studies that combined participatory approaches with energy systems modelling and planning. A central distinction in their review is between what they term *meaningful integration* and *shallow integration* of stakeholders. McGookin et al. (2021) argue that -meaningful integration occurs when participants are involved early in the research process, engagement is iterative, and stakeholders can influence problem framing, the analytical process, and the interpretation of results. In contrast, shallow integration refers to cases where stakeholders are consulted but have little or no influence on the direction of the analysis or the resulting conclusions.

One of the main findings of the review is that participatory approaches are argued to increase both legitimacy and robustness energy modelling exercises (McGookin et al., 2021). Legitimacy is increased when relevant actors are given the opportunity to contribute, diverse perspectives are included, and modelling assumptions and results are made transparent and open to discussion. Robustness may be strengthened by including a wider range of knowledge, experiences, values, and perspectives. This can reduce the risk of overlooking important aspects, using inappropriate assumptions, or ignoring local conditions. The review further stresses that participatory processes can help identify and discuss conflicts between objectives, such as trade-offs between environmental, social, technical, and economic goals. By making these trade-offs explicit rather than implicit, the resulting analyses may provide a more robust basis for decision-making. However, McGookin et al. (2021) also highlight the difficulty of translating these type of qualitative stakeholder perspectives into quantitative modelling assumptions. This process is often subjective and may reduce the transparency of how stakeholder input influences model results. Although these challenges cannot be fully eliminated, an iterative participatory process involving stakeholders with diverse expertise can make the translation process more inclusive and transparent. Repeated interaction also allows preliminary interpretations and assumptions to be revisited, thereby improving their consistency with the perspectives expressed by the participants.

3.4. External cost

External cost refers to the cost of producing a good or service that is not fully reflected in its market price (hence "external"). In this study, the term is used more narrowly to denote the

monetary value of external environmental effects on society, primarily damage costs from pollution, including related health impacts, but also potential co-benefits such as avoided emissions. In principle, externalities may also be non-environmental and can be either negative or positive, such as for example congestion (de Vries et al., 2024) or accidents (Arvidsson et al., 2018; EU, 2020). However, the focus of this study is on environmental external costs.

In practice, some of these environmental damage costs may be partly internalized through policy instruments, such as a CO_2 price (e.g., the EU ETS), implying that the cost is no longer entirely external. Nevertheless, for consistency in terminology, this study still refers to the underlying CO_2 damage cost as an external cost when describing and discussing results.

Because external costs, by definition, fall outside market prices, their assessment requires a broader socio-economic perspective. In this thesis, this perspective is represented by a hypothetical social planner who aims to evaluate impacts in terms of "overall welfare". Unlike individual decision-making, which is based on observed prices and individual constraints, the social planner considers costs and benefits from a broader societal perspective, including different population groups and time horizons. This affects, for example, the choice of discount rate, investment horizon, and the definition of what constitutes a cost.

A common way of incorporating external costs into policy evaluation is through cost-benefit analysis (Section 3.4.1), where environmental impacts are translated into monetary values and compared with other costs and benefits. This usually requires an impact pathway approach (Section 3.4.2), which traces the causal chain from emissions or pressures to environmental or health effects, and finally to monetary valuation where such economic valuation is possible (Section 3.4.3).

3.4.1. Cost-benefit analysis

A common application of external cost estimates is the so-called cost-benefit analysis (CBA), which aims to support decision-making by comparing the total costs and benefits of a project or policy. The basic purpose of CBA is to support decisions by comparing a project's or policy's total benefits and total costs (P. O. Johansson & Kriström, 2018), and in its simplest form, to recommend actions for which benefits exceed costs (or that maximize net benefits). One key motivation for monetizing external costs is therefore to include non-market impacts, such as environmental degradation and health effects, within the same decision-making structure as other economic decisions.

In CBA, impacts are typically assessed at the margin, meaning that valuation focuses on the welfare effect of relatively small changes in environmental quality or resource use. For example, rather than valuing the total loss of an ecosystem, analysis often considers the incremental impact of an additional unit of pollution on outcomes such as crop yields or land productivity (Costanza et al., 1997). While marginal values are conceptually defined as the change in welfare around a baseline, empirical applications often rely on unit values derived from average data, meta-analyses, or benefit transfer (see section 3.4.3) when context-specific marginal functions are not available.

3.4.2. Impact pathway approach

Estimating external costs requires understanding how emissions affect environmental quality, human health, and ecosystems. The magnitude of these effects depends on the pollutant, the

receiving environment, and the populations or ecological systems exposed, as different pollutants cause environmental and health damage through different causal mechanisms. For example, emissions of NO_X and primary particulate matter may both affect human health, but their contributions depend on atmospheric transformation, dispersion, population exposure, and the relationships between exposure and health outcomes (de Vries et al., 2024; EU, 2020; Holland et al., 2013; M. A. Huijbregts et al., 2017; IIASA, 2024; OECD, 2012).

A commonly used way of representing these relationships is through an impact pathway approach. In this thesis, the term *impact pathway approach* is used in a broad sense to describe the sequence of causal relationships linking emissions or environmental pressures to environmental damage. The terminology and separation of analytical steps vary between approaches and impact categories (Barrage & Nordhaus, 2024; Fantke et al., 2017; Holland et al., 2013; M. A. Huijbregts et al., 2017; IIASA, 2024; Owsianiak et al., 2023; Smeets & Weterings, 1999; Steen, 2016). However, the general principle is that environmental impacts, and in some cases their monetary value (Section 3.4.3), are derived by linking emissions to subsequent environmental or health effects. In air-pollution assessment, this typically involves atmospheric dispersion, exposure, physical impacts, and valuation (Holland et al., 2013). Impact-pathway analysis can also be applied within LCA, where emissions are linked through fate, exposure, and effects to potential environmental impacts (Krewitt et al., 1998). A simplified representation of this progression used in this thesis is given in Figure 6.

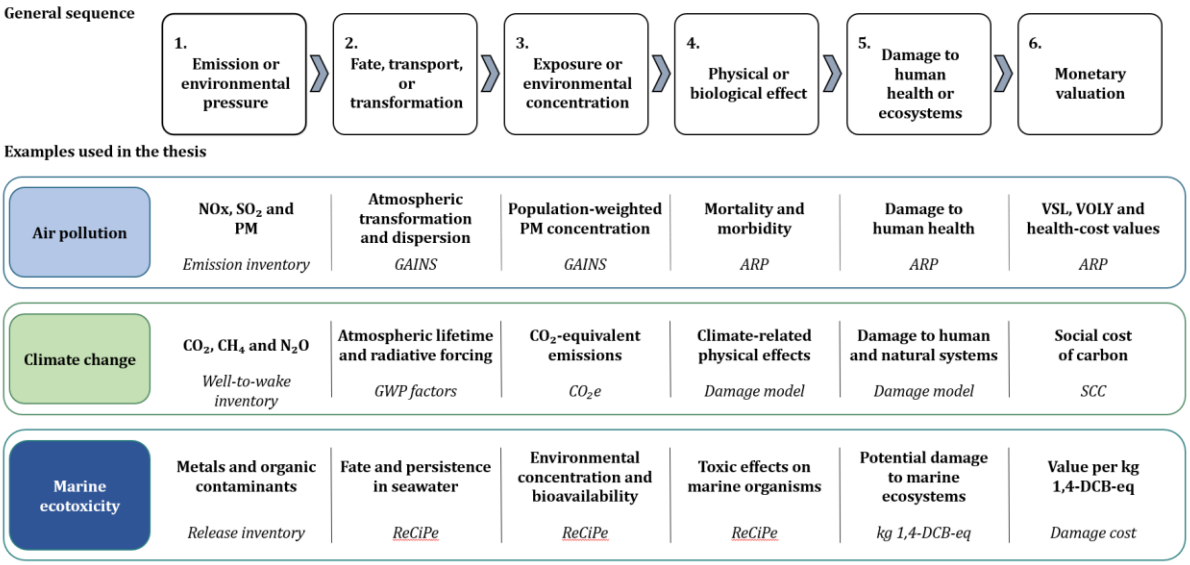


Figure 6. Simplified impact pathways linking shipping-related emissions and environmental pressures to impact indicators, damage, and monetary valuation. The analytical steps differ between impact categories, and not all steps are modelled explicitly in each pathway.

The three main impact categories examined in this thesis illustrate different representations of the impact pathway. For climate change, emissions of CO_2, CH_4, and N_2O are commonly converted into CO_2 equivalents using global warming potentials. GWP represents the integrated radiative forcing caused by a greenhouse-gas emission relative to CO_2 over a specified time horizon. It therefore represents an intermediate climate indicator rather than the complete pathway from emissions to economic damage (M. A. Huijbregts et al., 2016, 2017). A monetary climate damage estimate requires additional relationships linking emissions and atmospheric concentrations to temperature change, physical and economic consequences, and valuation. Integrated assessment

models may combine these relationships to estimate the social cost of a specific greenhouse gas. For example, Azar et al. (2023) extend an integrated assessment model to include methane emissions, the atmospheric methane cycle, climate damages, discounting, and methane abatement costs in the estimation of the social cost of methane.

For human-health impacts from air pollution, the pathway is spatially dependent. Emissions of primary $PM_{2.5}$ and precursor pollutants such as NO_X and SO_2 affect ambient $PM_{2.5}$ concentrations through dispersion and atmospheric transformations. Changes in population exposure are then linked to health outcomes using concentration-response relationships (Holland et al., 2013), after which health impacts can be monetized. The GAINS methodology, for example, quantifies premature mortality associated with long-term exposure to $PM_{2.5}$ by combining ambient concentrations, relative-risk functions, population data, and baseline mortality information (Amann et al., 2011). The resulting damage cost therefore depends not only on the mass emitted, but also on emission location, atmospheric conditions, population exposure, the health endpoints included, and the valuation applied to those endpoints.

Marine ecotoxicity involves a different and particularly "substance-intensive" pathway. Emissions of metals and organic contaminants are linked to their environmental fate, exposure, toxicity, and potential effects on marine species (Dong et al., 2016; Fantke et al., 2017; M. A. Huijbregts et al., 2016, 2017; Owsianiak et al., 2023). Because individual substances differ substantially in persistence, bioavailability, and toxicity, characterization factors are used to express their potential impacts in relation to a reference substance. This allows emissions of different contaminants to be aggregated into a common toxicity-potential indicator, such as kg 1,4-dichlorobenzene equivalents (M. A. Huijbregts et al., 2016, 2017). A separate damage cost can then be applied to monetize this indicator as for example derived in Noring et al. (2016) de Vries et al. (2024).

The use of characterization factors reduces the dimensions of the calculation, but does not imply that all emitted substances, locations, or ecological effects are represented equally well. The marine ecotoxicity pathway is limited by the substances for which characterization factors are available and by the spatial and ecological resolution of those factors. Both ReCiPe (M. A. Huijbregts et al., 2016, 2017) and Dong et al (2016) identifies spatial variation as particularly relevant to the ecological impact modelling of metals because fate, exposure, effects, and metal speciation varies between environments. The subsequent monetary valuation introduces an additional analytical step because the toxicity-potential indicator must be connected to an ecological change that can be valued.

One of the most difficult issues to consider when estimating environmental impacts is how to treat time in the impact pathway. The selected time horizon can substantially influence the relative importance of environmental impacts and may alter both the results and the conclusions of a study (Lueddeckens et al., 2020). For climate impacts, a shorter time horizon will, for example, increase the importance of methane relative to CO_2. For marine ecotoxicity, a longer time horizon will increase the relative importance of persistent substances, particularly metals, compared with less persistent substances such as PAHs. The importance of metals in toxicity assessment has been shown to depend strongly on the selected time horizon (M. Huijbregts, 1999; M. A. Huijbregts et al., 2017).

From a policy perspective, a 100-year time horizon is frequently used for climate impacts through GWP100, since climate change is considered a long-term environmental impact. However, this should not be interpreted as a general "truth". In their systematic review, Lueddeckens et al. (2020) found that the length of the time horizon is commonly described in the reviewed literature as a subjective or even arbitrary choice.

In the review different arguments are raised for using different time horizons. A long-time horizon may overestimate impacts if emissions distributed over a long period are treated as if they occurred at the beginning of the assessment period. It may also reduce the willingness of acting now and is generally associated with greater uncertainty, since new abatement strategies may emerge in the future. A short time horizon on the other hand, may shift environmental problems to the future and give insufficient consideration to intergenerational equality (Lueddeckens et al., 2020). According to the review, several authors argue that the selection of the time horizon should, at least partly, be a political decision or be left to the decision-maker rather than the researcher in a life cycle assessment.

Different time horizons are also used indirectly in different impact pathways. For example, NO_X emissions are converted into changes in annual mean $PM_{2.5}$ concentrations using spatially differentiated source-receptor coefficients derived from atmospheric chemistry and transport modelling (Amann et al., 2011; IIASA, 2024). These coefficients represent the response of annual mean $PM_{2.5}$ concentrations to changes in annual NO_X emissions around a reference emission level. Thus, unlike GWP100 and the time-horizon-dependent characterization of marine ecotoxicity, this characterization step does not use an explicit multi-year integration horizon.

3.4.3. Methods and practical challenges related to monetary valuation

Environmental goods and services that give rise to external costs are typically characterized as public goods, meaning that they are non-rivalrous and non-excludable. As a result, their value is not reflected in market prices, which make the use of non-market valuation methods necessary within a welfare-economic context (Börger et al., 2014). These methods aim to estimate individuals' willingness to pay (WTP) for environmental improvements or to avoid environmental degradation. Two main approaches are commonly used for this purpose: revealed preference and stated preference methods (de Vries et al., 2024; Mendelsohn, 2019a; Steen, 2016). Revealed preference methods extract values from observed behavior in related markets, such as housing prices or travel decisions, and therefore capture values associated with actual economic choices. However, these approaches are limited to "use values". In contrast, stated preference methods are based on survey techniques in which individuals express their WTP for hypothetical changes in environmental quality, allowing for the estimation of both use and non-use values.

Non-use values represent an important component of total economic value, as individuals may derive utility from the existence or preservation of environmental resources, independent of direct use (Börger et al., 2014). Because such values are not observable in market behavior, they can usually not be captured using revealed preference methods.

Stated preference approaches are, however, subject to several methodological challenges. One well-documented issue is insensitivity to scope (Whitty, 2012), where stated WTP does not increase proportionally with the magnitude of the environmental improvement. This behavior deviates from standard economic assumptions and may be related to cognitive limitations, survey

design, or alternative preference structures (Whitty, 2012). Stated preference methods may also be affected by biases related to questionnaire design and respondent behavior.

In many applied contexts, conducting new valuation studies is not feasible due to data or resource constraints. Benefit transfer is therefore frequently used as a second-best approach, where valuation estimates from existing studies are transferred to a new policy context (Liu et al., 2011). The validity of benefit transfer depends on the similarity between study and policy sites, including environmental, socio-economic, and cultural conditions. A central challenge in benefit transfer is the presence of transfer errors, defined as the deviation between the transferred estimate and the unknown true value at the policy site. These errors arise both from the underlying valuation studies and from the transfer process itself, including measurement and generalization (Liu et al., 2011). Spatial variation is especially important, as WTP often declines with distance from the environmental resource, a phenomenon referred to as distance decay (Johnston & Moeltner, 2014). In addition, cultural and contextual differences may influence valuation outcomes if not explicitly accounted for (Hynes et al., 2013).

Classification of monetary valuation methods

While monetary valuation methods are commonly categorized into revealed preference, stated preference, and, in some cases, cost-based approaches (de Vries et al., 2024; Mendelsohn, 2019a; Steen, 2016), as discussed above, this classification does not fully capture an important practical distinction: the type of evidence underlying valuation estimates. In particular, some estimates are grounded in empirically observed or modelled relationships, whereas others rely on responses to hypothetical scenarios. To address this, the classification developed in this section focuses explicitly on the evidence base of the valuation approach. It considers only the final valuation step, that is, how physical impacts are translated into monetary values, rather than the preceding impact modelling. To highlight this, a nuanced approach to classify valuation approaches is illustrated in Table 5.

In Table 5 methods are grouped according to their underlying evidence base: (i) approaches grounded in empirical relationships, including both directly observed data and model-based estimates, (ii) approaches based on stated preferences elicited in hypothetical settings, and (iii) approaches derived from cost-based proxies such as restoration or replacement costs. This classification aims to clarify what kind of evidence underpins different valuation approaches and helps explain why results may differ across studies.

Empirically anchored approaches refer to methods that are grounded in empirical relationships, either through observed behavior, measured outcomes, or model-based representations of real-world processes. This category includes revealed preference methods, such as hedonic pricing and travel-cost models, as well as cost-of-illness and human capital approaches. It also includes model-based approaches such as the social cost of carbon, which rely on empirically estimated relationships (e.g. between emissions, climate, and economic impacts) embedded in integrated assessment models. While such approaches involve modelling, they remain anchored in empirical evidence through the underlying data and estimated relationships. These methods therefore capture observed economic impacts or modelled welfare effects but may require substantial data inputs and rely on structural assumptions.

Hypothetical valuation approaches, in contrast, rely on stated preferences derived from survey-based methods such as contingent valuation and choice experiment (Kroes & Sheldon, 1988;

Mendelsohn, 2019a). Another practical approach to extending hypothetical testing is through deliberative processes, in which reflective dialogue and argumentation among participants form part of the valuation process (Klinke & Renn, 2002). These methods are particularly valuable for capturing non-use values and other effects that are not reflected in observed behavior. However, they depend on respondents' ability and willingness to express preferences in hypothetical scenarios and may therefore be sensitive to survey design, framing, and strategic or socially desirable responses. In addition, such approaches are not directly constrained by actual budget limitations, which may affect their realism in certain contexts.

Cost-based proxy approaches differ from the valuation methods above in that they do not attempt to measure the underlying welfare loss associated with environmental damage. Instead, they approximate value using the costs of intervention, such as restoration (e.g. remediation of contaminated sediments), replacement (e.g. substitute habitat creation), or abatement (e.g. the cost of measures needed to meet a target). These approaches are often practical and widely applied, particularly when other data is unavailable, but should be interpreted with caution. They may overestimate damages if they implicitly assume that full restoration is socially optimal, and may underestimate damages if restoration is incomplete or if institutional factors lead to insufficient compensation. Cost-based approaches thereby reflect resource expenditures rather than welfare changes, and should therefore be understood as proxies for value rather than direct valuation measures.

Table 5. Classification of monetary valuation approaches by evidence base and underlying measurement, based on the reviewed literature (Åström & Källmark, 2023; Azar et al., 2023; Barrage & Nordhaus, 2024; Cole, 2011; Cole & Moksnes, 2016; de Vries et al., 2024; EU, 2020; Kroes & Sheldon, 1988; Mendelsohn, 2019b; Noring et al., 2016; Steen, 2016).

Category	Methods	Evidence base	What is measured	Strengths	Limitations
Empirically anchored	Revealed preference; Social Cost of Carbon (SCC); Cost-of-illness; Human capital	Empirically grounded relationships (observed data and model-based estimates of exposures and outcomes)	Observed economic impacts or modelled welfare effects	Grounded in empirical relationships	Requires data or modelling assumptions
Hypothetical	Stated preference	Survey responses	Willingness-to-pay	Captures non-use values	Sensitive to survey design and biases
Cost-based proxies	Restoration; Replacement; Abatement	Observed or estimated intervention costs	Cost of intervention (proxy for avoided damage)	Practical; commonly available	Does not measure welfare directly

3.5. Knowledge, values, and uncertainty in policy evaluation

Evaluating environmental policy instruments requires more than estimating emissions, abatement costs, and policy-induced incentives. Such evaluations also depend on how environmental impacts are valued, which societal objectives are considered relevant, and how uncertain or disputed information is interpreted. How knowledge, values, uncertainty, and ambiguity influence the interpretation of policy effectiveness are therefore discussed in this section.

3.5.1. Descriptive and normative knowledge in the evaluation of environmental policy instruments

This study is interdisciplinary in character, combining insights from several domains to understand environmental policy instruments in shipping. From a philosophy of science perspective, policy evaluation involves both descriptive knowledge and normative elements. Descriptive knowledge concerns how policies, emissions, technologies, and economic behavior function and interact. Normative elements appear when the resulting outcomes are assessed against specific goals or evaluative criteria. Research on environmental policy instruments therefore relates to applied or design-oriented science (Niiniluoto, 1993; H. A. ; Simon, 1996).

In Niiniluoto (1993) framework, basic research and what he terms predictive science represent descriptive approaches, while design science represents a normative approach concerned with how specified goals can be achieved. Basic research aims to increase understanding of fundamental principles. Niiniluoto defines predictive science as research that applies existing knowledge to practical questions about expected outcomes. Both basic and predictive research can typically be expressed in the following form:

"X cause A[2] *in situation B" (1)*

Where X and A can, for example, be events or properties. Like basic and predictive sciences, design science also produces knowledge and collects it systematically. However, design science describes what "ought to be" (it is normative) and creates and evaluates "artifacts" to meet specific criteria or goals. The knowledge generated by design science targets real-world problems. Niiniluoto (1993) argues that if X is chosen by humans (not by nature), it is considered design science and can be expressed in the form Niiniluoto (1993):

"If you want A, and you believe that you are in a situation B, then you ought to do X." (2)

Understanding and evaluating environmental policy instruments involves both descriptive and normative elements. The descriptive elements concern relationships and mechanisms, such as the radiative properties of greenhouse gases and how policy design affects costs, investment conditions, and environmental outcomes. Models may also be used to estimate outcomes under specified assumptions. The normative element enters when these outcomes are evaluated against a particular goal or criterion.

In Niiniluoto's (1993) framework, design-oriented research ought to satisfy a "special condition of social relevance". This means that the goal represented by A in Equation (2) should address a socially relevant problem rather than reflect a goal selected solely by the researcher. Science can provide knowledge about the consequences of different actions and the conditions under which a goal may be achieved, but it does not determine the goal itself. In this thesis, social relevance is supported by relating to the evaluation of environmental goals, regulations, and strategies adopted through societal and political processes, as described in Section 2.3.

[2] Or "tends to cause with a probability."

3.5.2. Ethical considerations and the valuation of external costs

A fundamental premise in external cost research is that the monetary values used in calculations reflect underlying normative assumptions. These assumptions concern both the worldview embedded in the analysis (e.g., what counts as welfare-relevant), and the ethical considerations used to justify trade-offs. An important ethical challenge in this context relates to whether such trade-offs are meaningful at all when they require expressing fundamentally different kinds of values on a single scale (Aldred, 2006; Craswell, 1998; Reiff, 2014), such as health effects from air pollution, long-term climate damages, and ecological impacts from toxic substances in marine environments, which correspond to the main environmental impact categories examined in this thesis. However, an additional perspective less emphasized in valuation debates is psychological: moral judgments are often formed intuitively and only later rationalized (Haidt, 2001), which for example complicates the common modelling assumption that individuals and firms behave as fully rational maximisers (Sen, 1977; H. A. Simon, 1976).

To clarify why ethical considerations, matter for external-cost valuation, it is useful to distinguish between outcome-based and rule-based (deontological) moral reasoning. Consequentialist reasoning evaluates actions by their expected outcomes (e.g., comparing costs and benefits), whereas deontological reasoning starts from duties, rights, or rules about what should or should not be done, even when outcomes are uncertain. In environmental policy, these perspectives often interact: rules can set non-negotiable constraints or targets, while monetary valuation is used to compare alternative ways of meeting them. Reiff (2014) suggests that these two ethical perspectives are often combined in practice:

- *"While consequentialist methods of moral reasoning rely heavily on comparisons regardless of whether we are comparing the moral value of acts, or of the states of affairs they produce, or both, there is no reason why we must abandon consequentialism just because the required comparisons cannot always be made, for consequentialist moral reasoning may be able to exist inside a larger over-arching deontological structure that supplies the rule of decision-making when the normal tools of consequentialist reasoning are unavailable."*

A practical way to interpret this "hybrid" view is that environmental decision-making often combines rule-based and outcome-based reasoning. Regulatory frameworks can set normative constraints or duties (a deontological element), for example that it is right to reduce emissions that are harmful to humans, as in the establishment of emission control areas (NECAs). Within such constraints, consequential reasoning can still be used to compare alternative measures by quantifying expected outcomes, for example by monetizing emission reductions and associated health impacts and weighing these benefits against the costs of technologies and compliance. Seen this way, the "hybrid" ethical stance still relies on a characteristic step whenever outcome-based comparisons are used: translating diverse impacts and objectives into a form that can be compared and aggregated within the same decision construction.

The issue of placing qualitatively different kinds of values on a single scale, and, in many applications, subsequently expressing them in monetary terms, has been widely debated and is commonly referred to as *incommensurability* (Aldred, 2006; Craswell, 1998; Reiff, 2014). The term is used in slightly different ways across this literature. Craswell (1998) frames incommensurability primarily as a problem of justification: even if alternatives can be ordered on some metric, the question is whether any *single* scale can provide a legitimate justification for

choosing what ranks highest. Aldred (2006) focuses more closely on monetary commensurability in CBA and defines monetary incommensurability as cases where no unique monetary valuation exists for a change in an environmental good. In his formulation, commensurability requires that the alternatives can be measured precisely on a monetary scale, whereas incommensurability arises when no single exact monetary amount can represent the change on the environment, even if the alternatives may still be rankable. Reiff (2014) uses a broader definition, treating values as incommensurable when one is neither better than, worse than, nor equal to another, and he argues that commensurability and comparability stand or fall together (opposing what Alfred suggests). Incommensurability is directly relevant in the external-cost and shipping context, because the task is to express heterogeneous environmental impacts alongside company revenues and costs on a common (typically monetary) scale. These perspectives point to a pragmatic question: does CBA make sense for its intended purpose in a given context, and what is lost (or gained) when heterogeneous values are translated into a single monetary metric?

A related issue concerns assumptions about rational decision-making (Sen, 1977; H. A. Simon, 1976). Traditional economic models often assume that actors maximize individual outcomes (revenue minus costs). However, both Simon (1976) and Sen (1977) partly challenge narrow notions of rationality: Simon emphasizes "bounded rationality", where decision-makers face limited information and computational capacity and therefore chase "satisficing" (good-enough) solutions rather than true optimization, while Sen highlights that choices may also reflect moral commitments, duties, and social identities. One implication is that behavior may be better understood in terms of procedural rationality (how decisions are made under constraints) rather than only substantive rationality (which choice is made) (Simon, 1976).

3.5.3. Distinguishing between ambiguity and uncertainty

Previous sections have shown that the evaluation of environmental policy instruments involves both descriptive and normative elements. In this respect, the distinction made by Klinke and Renn (2002) between physical and social or psychological dimensions in risk evaluation may be useful. They emphasize that both dimensions matter, while also arguing that they should be separated analytically. Therefore, variation in outcomes that occurs because observations, causal relations, or policy implications are interpreted differently, or because different normative criteria are applied, is treated as *ambiguity* in this thesis. By contrast, uncertainty concerns either incomplete knowledge about the system (epistemic) or variability that is inherent in the system itself (aleatoric). This distinction (see Table 6) is analytical rather than fundamental, because ambiguity and uncertainty may interact in practical policy evaluation.

Uncertainty analysis is commonly associated with concepts such as accuracy, precision, variability, confidence intervals, probability density functions, and random and systematic error. Quantitative uncertainty can be analyzed using methods such as error propagation or Monte Carlo simulation, as recommended in for example the IPCC Guidelines for National Greenhouse Gas Inventories (Frey et al., 2006). Both aleatory and some forms of epistemic uncertainty may be represented quantitatively, provided that there is a defensible basis for specifying ranges or probability distributions. However, not all epistemic uncertainty can be represented adequately as numerical spread and such methods should therefore be interpreted with caution, because they may quantify the spread of results without resolving systematic bias in the underlying data or model assumptions (Frey et al., 2006). A narrow uncertainty range therefore does not necessarily imply that the representation of the system is accurate. For example, if literature-based specific fuel oil consumption factors (SFOC) are derived from measurement on older ships, such as for

example in (Cooper, 2001), while engine efficiency has improved over time, the resulting estimates may be systematically higher (biased) even if the reported spread is small.

Ambiguity, by contrast, relates to assessment of variation in outcomes that arises because the world is interpreted in different ways or because different normative values are applied in the evaluation (Klinke & Renn, 2002). This may concern different interpretations of what counts as an effective policy, such as whether effectiveness should be understood in terms of marginal or total emissions reductions, or the use of different discount rates and time horizons when assessing intergenerational justice. In such cases, sensitivity analysis or scenario analysis may be useful, not to estimate probabilities, but to examine how conclusions depend on alternative but reasonable assumptions.

In addition to uncertainty and ambiguity, there are situations in which analysts do not know, or do not agree on, the appropriate model structure, the relevant probability distributions, or how the desirability of alternative outcomes should be evaluated. Such situations may be described as cases of deep uncertainty (Marchau et al., 2019). Within such situations, uncertainty may relate to indeterminacy, that is, to cause-effect relationships that cannot be specified with sufficient confidence to support reliable prediction, for example because they are non-linear, or insufficiently understood (Klinke & Renn, 2002). At the more extreme end, deep uncertainty may also relate to total ignorance, in which analysts do not know the full extent of what they do not know (Walker et al., 2003). A practical implication of deep uncertainty is that probability-based methods may be difficult to justify, because the relevant probability distributions are not known or are contested. In such cases, scenario analysis, sensitivity analysis, or other approaches based on alternative plausible assumptions may be more appropriate than probabilistic uncertainty propagation.

Table 6. ***Analytical distinction between ambiguity and uncertainty based on descriptions in the literature*** **(Frey et al., 2006; Klinke & Renn, 2002)**

Main category	Subcategory	What the variation in outcomes relates to	Examples
Ambiguity	Interpretive	Variation because the same observations, evidence, or causal relations are interpreted differently	Different framings, contested evidence
	Normative	Variation because different values, objectives, or evaluative criteria are applied	Justice concerns conflicting priorities
Uncertainty	Epistemic*	Variation because knowledge about the system is incomplete or imperfect	Parameter uncertainty, model uncertainty and structural uncertainty
	Aleatory	Variation because the system itself exhibits inherent variability	Natural variability, stochastic processes

*Bias is not treated as a separate subtype of uncertainty, but as a potential source of systematic inaccuracy related to epistemic limitations. It may arise from non-representative data, flawed conceptualization, model misspecification, or systematic measurement error. Its direction and magnitude may not be known, and can therefore not necessarily be represented by expanding a numerical uncertainty range.

4. Material and methods

This chapter describes the methodological approach used in this thesis. It combines environmental assessment, private-economic analysis, socio-economic evaluation, and stakeholder participation. These methods are used to examine how policy instruments affect shipowners' investment conditions, how abatement costs compare with avoided external costs, and how actors in the maritime and energy systems interpret policy design, infrastructure, fuel availability, and uncertainty.

4.1. Brief description of the methods used in the five studies

The research questions (**RQ1–RQ3**) are addressed through the stepwise integration of five manuscripts, referred to as **Paper I–Paper V**. Together, these papers form an evaluation structure: **Paper I** starts from individual policy instruments and shipowner incentives, **Paper II** extends the analysis to fleet-level abatement costs and scenarios, **Paper III** adds consequences for marine environmental impacts and primes external cost, **Paper IV** contributes with stakeholder and system-level perspectives, and **Paper V** expands the assessment through external costs and a societal perspective. The methods used in each paper are briefly described below.

Paper I adopts the PoP in 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 of 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 investment or use of the examined abatement strategies, potential emissions reductions were not quantified in this paper, instead, emissions reduction is further explored in **Paper II, Paper III** and **Paper V.**

The cost-benefit analysis method in **Paper I** was conducted using model ships that represent different 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 used 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.

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 adopting the PoP. 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** and **Paper V**.

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. 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.

Paper III combines the PoP and SoP in 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 **Paper I, II and IV**, **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.

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. Ship activity and emissions from the Ship Traffic Emission Assessment Model (STEAM) was utilized in the break-even analysis, covering all ships using scrubbers globally (Jalkanen et al., 2021; L. Johansson et al., 2017).

Paper IV is an ex-ante study that examines the potential role of liquid fuels in future low-carbon energy systems by combining energy systems modelling with stakeholder participation. The method used **in Paper IV** is an integration of global energy systems modelling with a participatory approach involving stakeholders connected to a local port cluster. The analysis is based on qualitative data from interviews and workshops, combined with quantitative insights from the Global Energy Transition (GET) model. The interaction between modelling and stakeholder input allows for the exploration of how assumptions, uncertainties, and local conditions influence the interpretation of model results. The paper highlights that the role of different fuels, including

liquid fuels, depends not only on their modeled cost and performance, but also on how actors perceive feasibility, risks, and system constraints. In this context, uncertainties related to, for example policy design, legitimacy, time perspectives, and lock-in become central. However, in this thesis, **Paper IV** is used primarily for its stakeholder-based insights rather than for its global energy system modelling results. More specifically, the synthesis draws on the interview and workshop material to examine how actors in a local energy port cluster interpret future fuel transitions, policy signals, and infrastructure development. This material is used to complement, **Papers I–III**, which primarily focus on techno-economic policy evaluation from a shipowner perspective.

Paper V is an ex-ante study evaluating maritime environmental policy instruments from a societal perspective by comparing abatement costs with external environmental costs in future European shipping scenarios. The paper combines scenario modelling of fleet development, fuel use, emissions to air and water, and policy-driven abatement strategies with impact assessment and monetary valuation of climate change, air pollution, and marine ecotoxicity. In contrast to the earlier papers (**I-III**), which mainly focus on individual policy instruments, shipowner incentives, or specific environmental pressures, **Paper V** brings these elements together in a socio-economic cost-benefit assessment. It therefore contributes to the thesis by showing how the interpretation of policy effectiveness changes when multiple environmental impacts, societal costs, uncertainty, and ambiguity are considered simultaneously.

A brief overview of the policies, abatement strategies, emissions, methods and data applied in the five papers is outlined in Table 7. A more detailed description of the methods applied can be found in each paper.

Table 7 Overview of the method and materials used in Paper I-V.

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 PEMFC. 4. Battery electric (full).	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 emission/fuel consumption. Propulsion and auxiliary output (engine work) (SFOC/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
Paper IV	Not focus but elaborate on carbon storage, ban on combustion cars, green quotas and a trade ban of resources.	Energy carriers	Climate change	Swishing to alternative liquid fuels (all transport modes)	Literature and stakeholders	GET model. Stakeholder participation	Scenario analysis and stakeholder participation.
Paper V	Accelerated FuelEU Maritime compliance and transition to biocide-free foul-release coatings.	Energy carriers, anti-fouling paints and scrubbers	Climate change, Marine ecotoxicity, Human heath impacts.	Renewable alternative fuels and biocide free FRC	EU MRV (THETIS-MRV, 2023), Sea-web (S&P Global) (S&P Global, 2023) and literature.	MRV-based fleet scenario modelling combining fuel classification, fleet renewal, renewable-fuel allocation, FuelEU compliance calculations, antifouling release estimates, and emission factors for greenhouse gases, air pollutants, scrubber water, and antifouling substances.	Cost-benefit analysis, annuitization of cost, Carbon abatement cost, and scenario analysis.

4.2. Individual policy-instrument perspective (RQ1)

Four 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 four 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**), a low-carbon fuel standard though FuelEU Maritime (**Paper V**) and emissions standards through the IMO global sulfur cap (discussed indirectly in **Paper III** and partly in **Paper II**).

4.2.1. Scope of the four 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 four policy cases in Table 8. 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 further assessed in this thesis which is further described section 4.2.3.

Table 8. Summary of the scope of the four policies.

Scope	**Case 1**: Fairway discount	**Case 2**: EU ETS and **Case 3**: FuelEU Maritime	**Case 4**: 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

4.2.2. Definition of fleets

The term “fleet” could be used in different ways. In this thesis, it refers to multiple ships grouped together due to their relevance to the policy instruments that are 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, LMG 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.

4.2.3. The national perspective: Sweden

Environmentally differentiated fairway fees are assessed by applying a discount based on the number of port 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 estimation of the Swedish emissions of CO_2 was based on the Swedish port call statistics (SMA, 2023b). 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).

$$Swedish\ emissions = \sum_{port\ call,ship} CO_2\ per\ NM_{ship} \cdot Distance_{port\ call} \quad (3)$$

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 9.

Table 9. Key assumption regarding the calculation of Swedish emissions of CO_2 and Swedish port calls statistics in 2022.

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.
Distance calculation	Distance Calculation: Emissions are calculated for 50% of the distance between two ports.
Distance	Distances were based on SeaRoute (2023-2024), ShipTracker (2017),
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, i.e. estimates of the amount of fuel used onboard (GJ_{in}) and engine work (MWh_{out}), are rough estimates derived from MRV data (see **Paper II**), assuming that ships use only two types of fuel.

4.3. Policy-system perspective (RQ2)

To address RQ2: *"do interactions among environmental policy instruments, abatement strategies and different stakeholders influence investment decisions and environmental outcomes in shipping?"*, this thesis use a qualitative stakeholder-based interpretation (from **Paper IV**) and a quantitative policy-interaction analysis. The purpose is to assess not only how policy instruments interact in terms of costs and emissions, but also how such interactions are shaped by actors, investment conditions, infrastructure and fuel supply.

The quantitative part uses the abatement-cost approach developed in **Paper II** to examine how selected policy instruments interact when applied sequentially to the European fleet. The interaction effects are evaluated by comparing changes in abatement costs and emissions reduction potential after each policy step. The analysis is carried out for three sequential policy measures: first, the Energy Efficiency Design Index (EEDI), which improves the energy efficiency of newly built ships and is included as part of the baseline, second, a hypothetical global ban on scrubber-water discharge, represented by assuming that ships use MGO or VLSFO instead of HFO

with scrubbers; and third, an extension of the EU ETS to cover 100% of emissions from inbound and outbound traffic.

The qualitative part builds primarily on **Paper IV**. The policy-system perspective examines how environmental policy instruments interact across emission types, abatement strategies, and actor responses. The interview and workshop material are used to analyze how actors in a local energy port cluster interpret future fuel transitions, policy signals, and infrastructure development. In **Paper IV**, stakeholder participation was organized in three phases: semi-structured interviews followed by two workshops. The interviews were conducted with stakeholders connected to the Port of Gothenburg energy cluster and included actors such as bunker suppliers, fuel producers, gas infrastructure actors, port representatives, and storage companies. In total, 16 stakeholders participated in at least one phase of the process. The interviews lasted approximately 40–90 minutes, were recorded, transcribed, and analyzed thematically. The workshops were then used to discuss interview insights, modelling assumptions, preliminary model results, and scenario narratives.

In this thesis, **Paper IV** is used to complement the abatement cost focused analyses by showing how actors perceive policy instruments, investment risks, fuel availability, infrastructure needs, and uncertainties. This is relevant because technically or economically feasible measures are not necessarily implemented if they depend on actors outside the shipping company, such as ports, fuel producers, refineries, storage operators, or other parts of the energy system. The stakeholder perspective therefore expands the analysis from individual shipowner incentives to the larger system in which maritime fuel and technology transitions take place.

In addition to the interviews included in the main analysis, three supplementary interviews were conducted. These were undertaken after it became apparent that decisions concerning two major port related investments, a large processing plant for heavy fuel fractions and a proposed shore-to-land LMG transfer connection, had been influenced by factors not initially captured in the study. The supplementary interviews included (i) a representative from an environmental NGO that actively opposed the refinery expansion, (ii) aprotester involved in the protest against the new LMG infrastructure and (iii) a key political decision-maker. Although these interviews were not included in the formal analysis of **Paper IV**, they are relevant to this thesis and are therefore used as illustrative examples. Where relevant, insights from three student theses conducted within the broader research process of this thesis are also used to contextualize early responses from shipping companies (Ahlström & Johnsson, 2023; Emanuelsson & Fransson, 2023; Fagerbaek Guldstrand & Wall, 2024). The interview questions in these projects were developed in close connection with the analytical focus of this thesis, but the material is used here only as supporting context.

4.4. Evaluative perspective (RQ3)

RQ3 examines how the interpretation of policy effectiveness changes when the private investment perspective is complemented by a socio-economic perspective. The analysis compares the private cost of abatement with the economic incentives created by policy instruments and compares the socio-economic cost of the same measures with their avoided external costs. These comparisons address related but different questions: if current policy instruments provide a competitive private investment case, and if an abatement measure may be socially beneficial even when it is not privately profitable.

The distinction between PoP and SoP is central for interpreting policy effectiveness under uncertainty and ambiguity. The choice of interest rate and investment horizon was also one of the issues debated in the interviews and workshop in **Paper IV**, illustrating that these parameters are not only technical assumptions but also reflect different actor perspectives on risk, time, and investment conditions. From a private perspective, effectiveness is evaluated in relation to whether policy instruments alter the economic incentives faced by ship owners and operators. From the social perspective, effectiveness is evaluated in relation to whether abatement strategies reduce external costs sufficiently to justify their technical costs. This synthesis therefore combines the cost perspectives used in the appended papers with the theoretical concepts introduced in Chapter 3, particularly external costs, uncertainty, ambiguity, and evaluative assumptions.

In the baseline of the PoP, annualized costs are calculated using an interest rate of 10% and an investment lifetime of 10 years (or 30 years for production facility). The PoP is used to assess how abatement options compare with market-based policy instruments, such as taxes, subsidies, charges, or other economic incentives. The intention of this type of analysis is to help to determine whether a policy instrument is likely to be "strong" enough to make an abatement strategy financially attractive for shipping companies. In SoP, annualized costs are calculated using a discount rate of 3% and the technical equipment lifetime, assumed to correspond to the ship, production facility or lifetime of 25–40 years (used in Equation 5). The SoP is not used to represent market behavior. Instead, it is compared with the external costs (Equation 4) avoided by reducing emissions (Equation 5). This comparison addresses whether an abatement measure can be considered a good investment from the perspective of society: if the social cost of reducing emissions is lower than the avoided external damage cost, the measure may be interpreted as socially beneficial even if it is not profitable on the market with or without policy support.

The general approach used to estimate external cost, emission reductions and abatement costs in **RQ3** is described by Equations 3-4:

$$Annual\ abatement = yearly\ baseline\ emission - estimated\ emission\ with\ abatement\ strategy = \left(EF_{baseline} - EF_{abatement\ strategy}\right) \cdot AD \quad (3)$$

Annual abatement was calculated as the difference between the estimated baseline emissions and the estimated emissions after applying an abatement strategy. Emissions were estimated using activity data (AD) and emission factors (EF) where AD represents the level of activity, EF represents emissions per unit of activity. Annual abatement represents the physical reduction in emissions or releases achieved by an abatement strategy relative to the baseline. To assess its societal relevance, this reduction must be linked to the environmental impact pathways described in Section 3.4.2 and monetized through the damage cost estimates introduced in Section 3.4 and further discussed in Section 3.4.3. The total external benefit of an abatement strategy is therefore calculated by combining the annual abatement with a characterization factor and a monetary damage cost:

$$Total\ external\ benefit = Annual\ abatement \cdot Charactarisation\ factor \cdot Damage\ cost \quad (4)$$

This means that the estimated external benefit depends on the physical reduction achieved, the modelled relationship between emissions and environmental impacts, and the monetary value assigned to the avoided damage.

The external costs were estimated for three impact pathways: climate change, air pollution and marine ecotoxicity in the same way as they were calculated in **Paper V** but for hypothetical scenario in 2023 instead of future scenarios at a fleet level. However, since ship-specific AIS were unavailable, emissions were assumed to be distributed across sea basins according to the overall geographical activity distribution of the analyzed fleet for each ship.

5. Results and discussion

5.1. Current design of individual-policy-instruments: four case studies

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

5.1.1. 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". While Paper I was based on seven model ships, the same type of reasoning can also be extended to the fleet level by combining cost and emissions data for ships calling at Swedish ports. In this thesis, Swedish GHG emissions are based on the estimates by Styhre et al. (2024), with minor modifications. Figure 7 shows the resulting WTW GHG emissions for the Swedish fleet, disaggregated by ship segment and by the number of fairway fees paid in 2022. The estimate includes 50% of emissions from incoming and outgoing voyages and 100% of domestic emissions, corresponding to 4.58 million tonnes CO_{2e}. The figure shows that RoPax and RoRo ships account for a substantial share of emissions and include many ships with frequent calls at Swedish ports, which is consistent with the assumptions used for the model ships in **Paper I**.

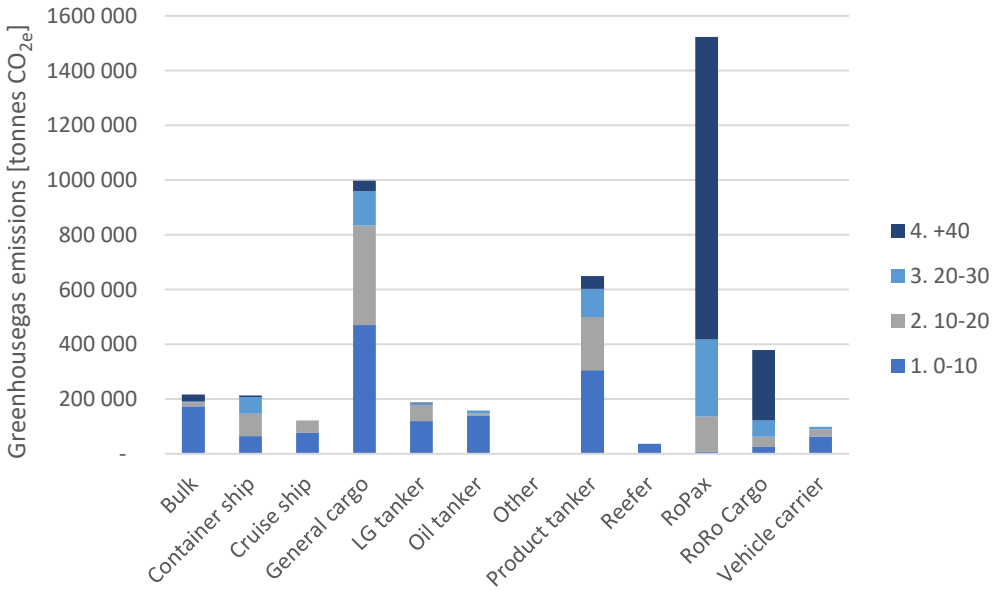


Figure 7. Swedish fleet GHG emissions (WTW) estimate including 50 % of incoming and outgoing ships and 100% of domestic emissions. The colors indicate how many times each ship paid for the fairway fee in 2022. The figure includes all ships that paid the fairway fee, i.e., passenger and cargo ships that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

Figure 8 illustrates how the abatement cost for e-MeOH changes when the Swedish environmentally differentiated fairway discount is included. The analysis is based on ships entering Swedish ports that are also included in the European scenario analysis, which means that

smaller ships between 300 and 5,000 GT and ships not calling at Swedish ports are excluded. The results show that adding the fairway discount has only a marginal effect on the estimated abatement cost. This supports the conclusion from **Paper I** that the Swedish environmentally differentiated fairway fee is not designed to support CO_{2e} abatement.

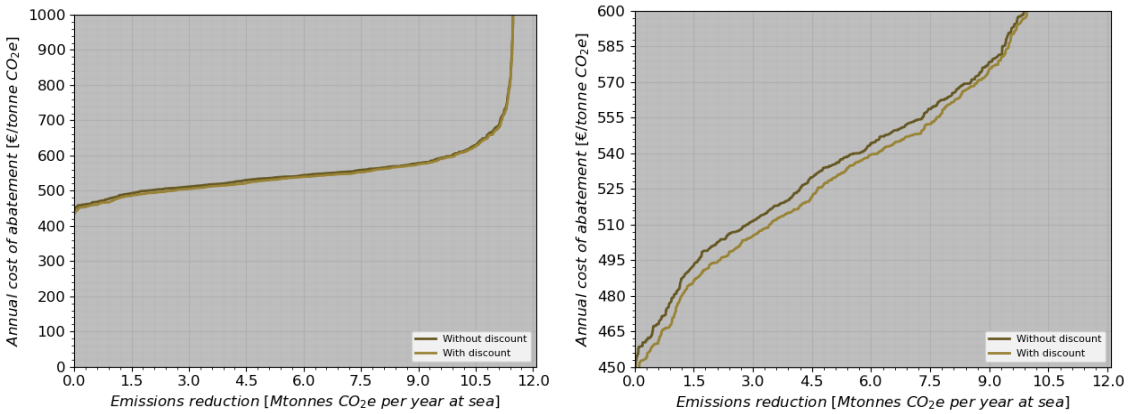


Figure 8. Abatement cost for e-MeOH compared to CO_{2e} reduction potential for one existing policy (the environmentally differentiated discount for Swedish fairways), for ships entering a Swedish port in 2035. The figure includes all emissions reduction potential for the extended scope pf EU ETS. The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

Taken together, the fleet-level analysis and the results from **Paper I** indicate that the discounts alone are unlikely to affect shipowners' willingness to invest in the investigated abatement strategies, including selective catalytic reduction (SCR), biocide-free antifouling coatings, a small battery system (1 MWh), and onshore power supply (OPS). The main reason is that the discounts are too low relative to the investment costs. Port discounts were also limited; in four of the five studied ports, the discount amounted to only 10% of the port fee. This suggests that the policy, in its current design, is unlikely to be effective in reducing emissions through technical or operational investments, which is directly relevant to **RQ1**.

This interpretation is consistent with Trafikanalys (2017), which argued before the reform of the environmentally differentiated fairway dues that the new system would weaken previous environmental incentives by using a broader index and differentiating a smaller share of the fees. **Paper I** also suggests that a more targeted allocation of the discount could have made the instrument more relevant for specific abatement measures. For example, if the available discount had been directed towards a single technology or emission category, the rebate could approach the cost of some measures, such as switching from copper-based antifouling paint to biocide-free silicone-based fouling release coatings. Such coatings may also be attractive to shipowners, as they have demonstrated equal or superior effectiveness compared with copper-based paints in preventing hull fouling (Lagerström et al., 2022). However, the current Swedish fairway discount is not structured as an investment support scheme. It is instead applied continuously based on port calls, and repeated changes to the system may create uncertainty for shipowners considering investments based on this policy. Ultimately, as argued in **Paper I**, the total discounts provided (€3.6 million in 2018 and €5.8 million in 2019) are insufficient on their own to motivate cleaner technology investments when distributed across many shipowners and many possible abatement strategies.

5.1.2. 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 can encourage shipping companies to invest in cleaner fuels or energy carriers and thus reduce GHG emissions. **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 *"the current design of individual environmental policy instruments impact shipping companies to invest in technical and operational measures"*, it is also relevant to assess the present conditions. If the high renewable fuel production costs in **Paper II** (**Paper V** for biomass) are assumed to better represent current price levels, in combination with higher prices for batteries (500 euros/kWh) (**Paper I** and **Paper V**), electrolyzer (2100 euro/kW) (European Hydrogen Observatory, 2026), 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 9. 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_{2e} emissions from each ship, where 50% of CO_{2e} emissions from ingoing and outgoing travel and 100% of domestic is included. One can, for example, see that for bio-MeOH there are many ships, with annual emissions reduction potential up to about 73 Mtonnes, where the abatement cost is below 450 €/tonne CO_2, while for batteries the abatement cost is below 450 €/tonne CO_2 only for ships emitting less than approximately 3 Mtonne CO_2 per year. The result suggests that current EU ETS prices (60-90 € per tonne CO_2, see Figure 4) are likely too low to incentivize abatement (**RQ1**), except for ships operating on shorter, frequent routes where batteries could be a feasible option.

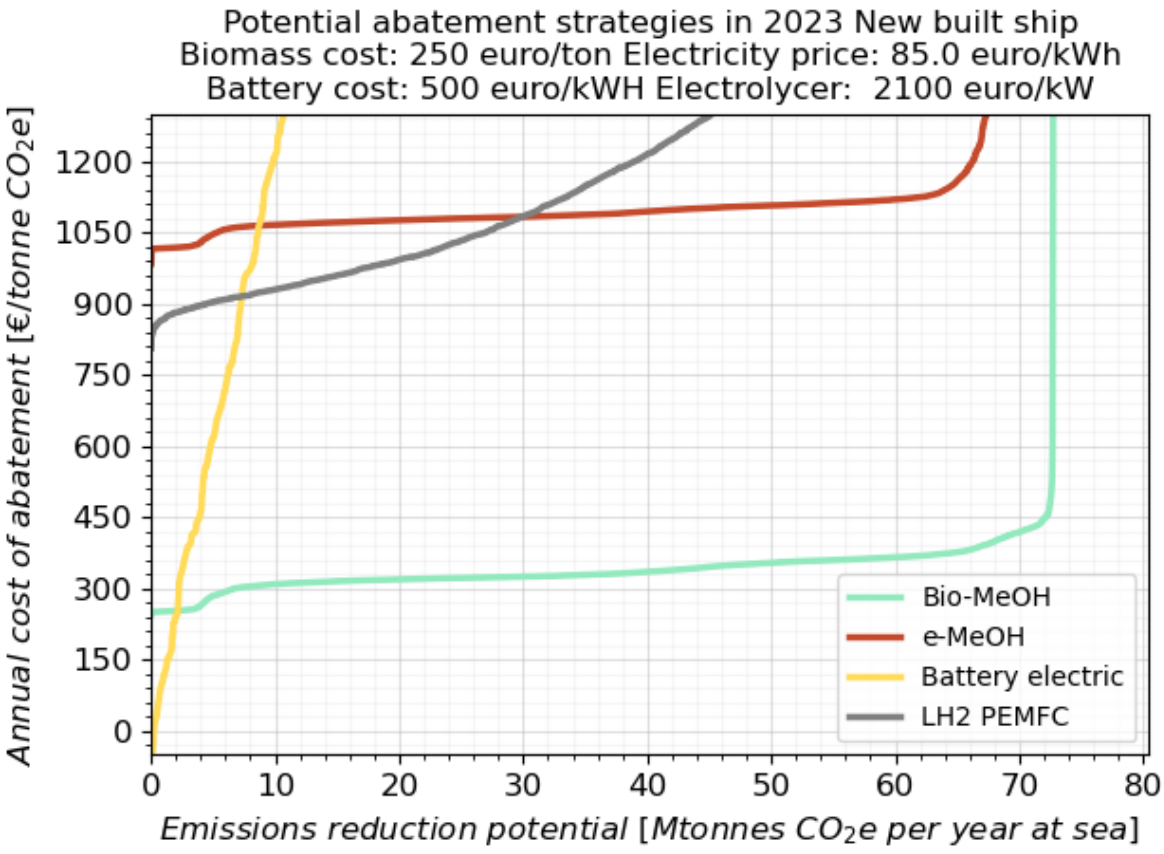


Figure 9. Estimated abatement cost in the European fleet based on fossil fuel prices in 2023 and high cost of producing renewable alternatives and high cost of battery, electrolyzers and PEMFC.

The analysis in Figure 9 should not be interpreted as identifying which abatement strategies should be prioritized from a long-term societal perspective. Rather, abatement cost curves indicate the approximate incentive level needed for different strategies to become attractive to shipowners, for example through subsidies, refundable emission payments, or feed-in tariffs. They also show how costs vary across ships, which is useful when considering differentiated policy design and the timing of support.

However, if policy relies mainly on short-term cost efficiency, as in a broad market-based instrument such as the EU ETS, there is a risk of lock-in and suboptimal long-term outcomes. New technologies often face market-entry barriers and may struggle to compete with cheaper or more mature alternatives (Azar & Sandén, 2011; Geels, 2005). In **Paper II**, bio-MeOH generally had the lowest abatement cost, except for RoPax ships on shorter routes where electrification was more favorable. A market-led transition would therefore likely begin with electrification for suitable short-distance ships and biofuels for some other ships. Since sustainable biofuel potential is limited and more costly alternatives such as, bio fuel using more expensive feedstocks, hydrogen and e-fuels require time for learning, infrastructure, and supply-chain development, broader price signals may need to be complemented by more targeted and long-term support. Feed-in tariffs for solar and wind power provide one example of such support for initially costly technologies (Jacobsson & Lauber, 2004). At the same time, the transition should account for upstream emissions, since electricity production still has a non-negligible carbon intensity, averaging about 200 g CO_2e/kWh (EEA, 2024). This tension between least-cost compliance, technology development, and fuel availability is also central to the next case, FuelEU Maritime, which directly regulates the greenhouse gas intensity of the energy used onboard.

5.1.3. FuelEU Maritime (Case 3)

FuelEU Maritime is included in the baseline scenario in **Paper V** and represents a different type of policy instrument than the EU ETS. Rather than pricing emissions directly, it regulates the yearly average GHG intensity of the energy used onboard. The potential impact of this regulation is illustrated in Figure 10. Since biofuel was among the lowest-cost renewable fuel options in both **Paper II, Paper IV** and **Paper V**, bio-MeOH is used here as an illustrative compliance option.

Figure 10 shows how the amount of fossil fuel that can remain within the FuelEU Maritime limit decreases over time, while the amount of bio-MeOH required to meet the regulation increases. Under both the low- and high-demand assumptions, the required amount of bio-MeOH remains relatively limited until 2040. The larger shift occurs after 2040, especially in 2045 and 2050, when the GHG-intensity requirement becomes substantially stricter. FuelEU Maritime therefore allows relatively high fossil-fuel use during the early compliance period, while the stricter post-2040 targets require substantially larger shares of low-GHG fuels.

The figure should be interpreted as an illustrative case rather than a prediction of the future fuel mix. It assumes that the required GHG-intensity reduction is achieved through bio-MeOH and does not include direct electricity use or other low-GHG options. If electrification, shore power, hydrogen, or other compliance options are used, the required amount of liquid renewable fuel would likely be lower. However, the general timing issue remains: if compliance is mainly delayed

until the stricter targets after 2040, demand for low-GHG fuels and related infrastructure may increase sharply late in the transition period.

At the same time, the design of FuelEU Maritime may be less effective in creating early demand for more expensive fuels that require new production capacity, such as e-fuels, or more expensive biobased options. The possibility to comply initially through limited shares of lower-cost renewable fuels (or even fossil fuels with lower CO_{2e} intensity such as LMG), and later through pooling or other flexibility mechanisms, may reduce the incentive to invest early in costly production pathways. This could become problematic because both onboard technology investments and fuel production capacity require long time horizons and substantial capital. FuelEU Maritime is in this way also an illustration of the same tension as in the EU ETS case: policies that support least-cost compliance in the short term may not necessarily create the early market conditions needed for more expensive long-term alternatives. This is also why **Paper V** includes an alternative faster-transition scenario for FuelEU Maritime.

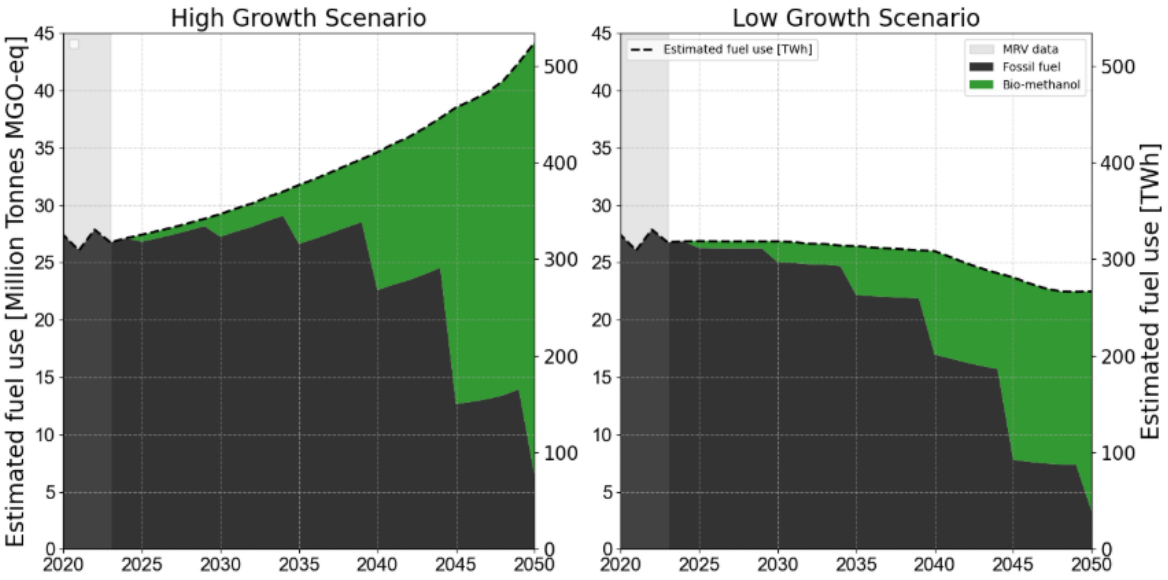


Figure 10. Estimated effect of FuelEU Maritime on fossil fuel use and bio-MeOH demand under low- and high-demand assumptions. As the required GHG-intensity reduction increases over time, the share of fossil fuel that can remain within the compliance limit decreases, while the amount of bio-MeOH required to meet the regulation increases.

5.1.4. The rise of scrubbers: IMO global sulfur cap (Case 4)

The revised IMO global sulfur standard entered into force in 2020 and required ships to reduce sulfur emissions either by using compliant low-sulfur fuels or by installing exhaust gas cleaning systems. The regulation has been effective in reducing SO_X emissions to air, with measurable reductions in atmospheric SO_X concentrations after implementation, as shown for example by Tauchi et al. (2022). However, one important consequence of allowing scrubbers as a compliance option was that many shipowners continued using HFO together with scrubbers instead of switching to low-sulfur fuels such as MGO. This matters because the use of MGO is associated with lower emissions of substances such as PAHs and metals than HFO (Lunde Hermansson et al., 2021).

The economic assessment in **Paper III** was motivated by the increasing number of local and national restrictions on scrubber-water discharge (BIMCO, 2024; EGCSA, 2024). These

restrictions have been contested by parts of the shipping sector, partly because shipowners argue that scrubbers were installed in "good faith" as an accepted compliance option under the sulfur regulation and that later discharge bans may undermine those investments (**Paper III**).

This concern needs to be considered alongside the environmental consequences of scrubber use. Scrubber-water discharge is a major source of PAHs and metals to the marine environment in areas with dense shipping activity, including the Baltic Sea (Ytreberg et al., 2022). **In Paper III**, the Baltic Sea, which is designated by IMO as a Particularly Sensitive Sea Area (PSSA), was estimated to have received 3.2 billion m^3 of open-loop scrubber water between 2015 and 2022. The environmental damage-cost assessment in **Paper III** was not directly comparable with the private cost-benefit analysis of scrubber use, because different methodological approaches were applied, which is was later partly address in **Paper V**. It was also incomplete, as it only included nine metals and ten PAHs for which ReCiPe characterization factors were available (M. A. Huijbregts et al., 2016, 2017). Since 69 substances have been identified in scrubber water (Lunde Hermansson, 2024), and since other external effects such as GHG emissions and marine acidification were excluded, the estimated damage costs are likely conservative.

The results from **Paper III** show that 51% of shipowners had already reached break-even by the end of 2022 in the central cost case, meaning that fuel-cost savings had compensated for the scrubber installation cost. This is consistent with the fact that more than 5 000 ships globally have installed scrubbers instead of switching to low-sulfur alternatives. The 51% estimate may also be conservative, since the analysis did not account for the possibility that shipowners could store fuel when prices were low and use it when price differences were larger. For the Baltic Sea, the marine environmental damage cost from scrubber-water discharge over the period 2014–2022 was estimated at more than €680 million.

Paper III therefore illustrates a case in which an environmental policy instrument was effective in reducing the targeted emissions, SO_X to air, by requiring shipowners to act. At the same time, the policy design allowed scrubbers as a compliance pathway, which contributed to a new waste stream of scrubber water containing hazardous substances. The case shows that a policy can be effective with respect to its primary target while still creating unintended environmental consequences in another impact category. It also indicates that the main driver behind the adoption of scrubbers was economic: by using HFO instead of more expensive low-sulfur fuels, shipowners could reduce compliance costs, while part of the environmental burden was shifted from air to water. This is further explored from a policy-system perspective in **RQ2**.

5.2. Policy-system perspective: Interactions between different environmental policies and actors (RQ2)

This section addresses **RQ2** by examining policy interactions from a policy-system perspective. The analysis first builds on a quantitative fleet-level analysis based on the abatement-cost approach in **Paper II** and the scrubber-related results in Paper III. In this analysis, selected policies are added stepwise to illustrate how their combined effects influence abatement costs and emissions reduction potential for the European fleet: first EEDI-related efficiency improvements, then a hypothetical global ban on scrubber-water discharge, and finally an extension of the EU ETS to cover 100% of inbound and outbound traffic. It then complements these quantitative interpretation interviews and workshops in **Paper IV** to discuss how stakeholders perceive policy instruments, fuel availability, investment conditions, infrastructure, and uncertainty.

5.2.1. European fleet scenario analysis of different policy instruments: abatement cost curves

Figure 11 provides a complementary way of illustrating policy interactions compared to the narrative above by quantifying the impact using abatement cost curves for the shipping sector. The analysis applies the same methodological approach as in **Paper II** and shows the renewable central-cost scenario for e-MeOH in 2035, assuming a hypothetical retrofit of the European fleet to e-MeOH. The curves show how the resulting abatement cost changes when policies are added sequentially. First, the EEDI is included in the baseline, primarily reducing total GHG emissions through improved energy efficiency. Second, a global scrubber ban is added, which raises the reference fuel cost by requiring ships that would otherwise use HFO with scrubbers to switch to more expensive low-sulfur fuels such as MGO. This lowers the resulting abatement cost of switching to e-MeOH. Third, the EU ETS is extended to cover 100% of inbound and outbound traffic, which further reduces the estimated abatement cost and increases the emissions reduction potential included within the policy scope. However, even in this final case, the abatement cost remains above recent EU ETS allowance prices of about 50–75 €/tonne CO_2, because the policy mainly affects the annualized investment cost and the emissions covered, not the variable fuel-cost difference.

The figure therefore illustrates that policy instruments interact both economically and environmentally. Measures that increase the cost of continued fossil fuel use, such as a scrubber ban or an expanded EU ETS, can reduce the relative cost of low-GHG alternatives. Conversely, allowing scrubbers lowers fuel costs by enabling continued HFO use, which increases the abatement cost of switching away from fossil fuels. Even so, the analysis shows that this impact is relatively small difference for CO_2 abatement cost at a fleet level (since only some ship having scrubber installed). These interactions also affect more than one environmental dimension: fuel choice and scrubber use influence GHG emissions, air pollutants, and discharges to water. As discussed in **Paper II**, **Paper III** and **Paper V**, HFO use also affects PM and CO_2 emissions, and scrubbers increase onboard energy demand, which can raise fuel consumption and direct operational CO_2 emissions. In addition, regulations operate at different geographical levels and are administered by different bodies, meaning that shipowners may face very different combinations of costs, incentives, and environmental effects depending on their routes and operations.

Policy instruments should not be interpreted only in isolation. Their combined effect depends on how they alter relative fuel prices, which emissions are included, which abatement options remain available, and which actors are affected. For RQ2, this illustrates how interactions between policies, emission types, and abatement strategies can shape both investment incentives and environmental outcomes.

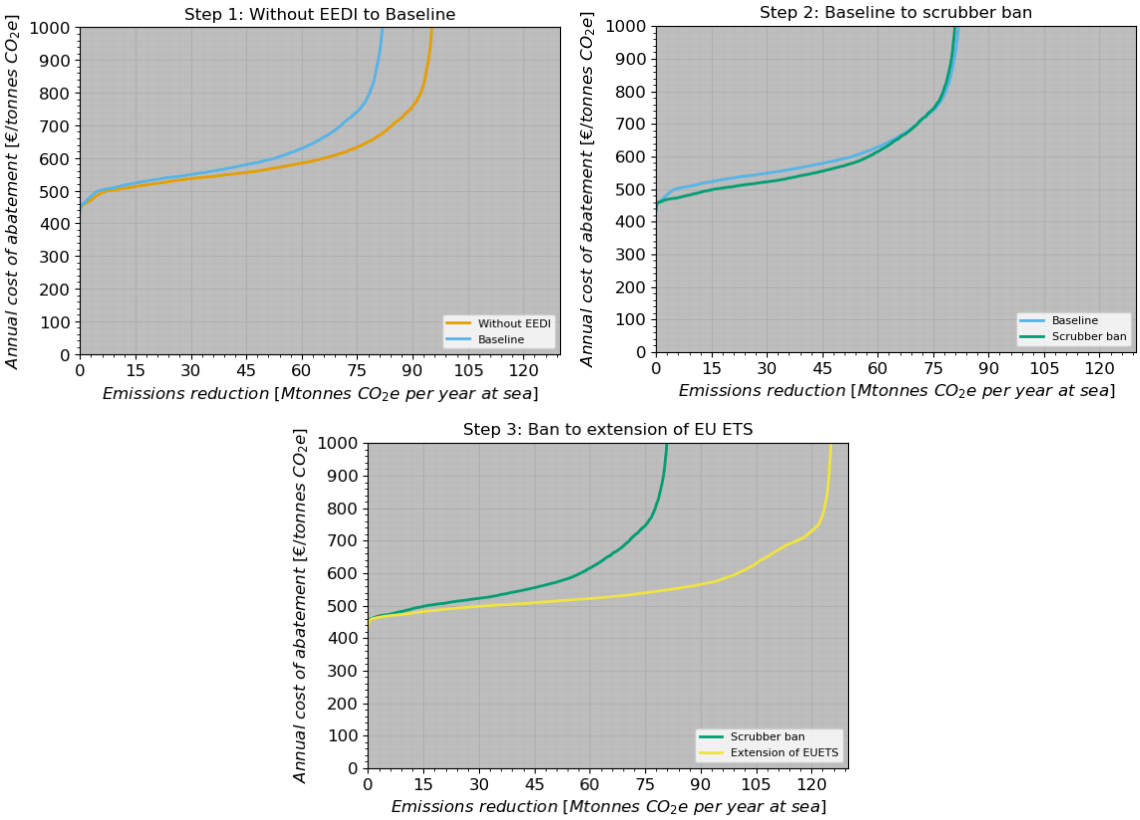


Figure 11. Estimated resulting abatement cost and emissions reduction potential for retrofitting e-MeOH ships and the CO_{2e} reduction potential for one existing policy (EEDI) and two proposed policies (a global scrubber ban and an extension of the EU ETS) for the European fleet in 2035. The three figures 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.

5.2.2. Stakeholder participation

One important contribution of the interviews and workshops was that they identified practical and system-related aspects of the fuel transition that were not fully visible in a purely model-based or literature-based analysis. Because the participants were active in the port cluster and had substantial operational experience (**Paper IV**), they were able to point to constraints and dependencies that would otherwise risk being overlooked. These included for example the role of refinery processes, policy instrument influencing fuel availability, the diversity of end-users connected to the port cluster, and infrastructure-related issues. The stakeholder interaction was not a quantitative validation of the model. It was used to assess assumptions and identify system conditions that influence whether technically feasible abatement strategies can be implemented in practice.

Policy instruments influencing fuel availability

The GET modelling results in **Paper IV** indicate that substantial volumes of liquid fuels may still be needed globally after 2050, although the demand is expected to decline more strongly in some sectors than in others. This long-term role of liquid fuels was also discussed in interviews and workshops, but the stakeholder perspectives were not uniform. Some actors expected a large reduction in liquid-fuel volumes handled within the port cluster, with one interviewee referring to internal assessments of reductions of around 70% after 2045. This was mainly linked to

expected electrification in road transport, since a large share of the current fuel demand handled in the port-cluster is connected to road vehicles in Northern Europe.

At the same time, the workshop discussions and voting exercises (see supplementary material in **Paper IV**) showed that many participants still expected liquid fuels to remain important for sectors that are more difficult to electrify, especially shipping, aviation, and to some extent heavy road transport. The stakeholder material therefore suggests that future liquid-fuel demand is not only a question of total volumes, but also of how demand shifts between sectors and fuel types. This is important for shipping, since the availability and price of alternative marine fuels will depend on developments in the broader transport and energy system, as well as on maritime regulation.

In this context, several participants emphasized the importance of stable and well-designed policy instruments. The interviews and workshop discussions included examples where policy design had created uncertainty or unintended consequences, such as the Swedish "pump legislation", which required certain fuels to be available at filling stations for road vehicle. Other instruments, such as the Swedish greenhouse gas reduction quota, were described as important for creating demand for renewable fuels in road transport, while the Swedish CO_2 tax was not perceived as sufficient on its own to drive a large-scale shift towards biofuels in transport sector. This is relevant for shipping because FuelEU Maritime can be interpreted as creating a similar quota-like demand for low-GHG fuels (also fossil fuels) in the maritime sector.

Related to McGookin et al.'s (2021) critique that many participatory studies fail to adequately represent relevant stakeholders, **Paper IV** included representatives from almost all business operators in the Energy Cluster in Gothenburg, and all companies with energy infrastructure in the port were invited to participate. In addition to the interviews included in the main analysis, three supplementary interviews were conducted. They demonstrate how investment decisions are often influenced by factors extending beyond technical rationality and short-term economic considerations, including public opinion, political processes, and societal values.

A new LMG terminal was built in the Port of Gothenburg, and plans were made to connect the terminal to the Swedish natural gas grid through a new pipeline. The fossil-LMG as intended to be transported by ship and used both as a marine fuel and as an energy source for industrial applications. In addition, the port viewed the proposed grid connection as a means of facilitating the future transfer and distribution of liquefied biomethane (bio-LMG), thereby supporting a gradual transition from fossil-based to renewable gaseous fuels. However, the project faced strong opposition from climate activists, who physically blocked access to the energy terminal in protest. The protesters argued that any new investment in fossil-fuel infrastructure was incompatible with climate goals. The demonstrations received considerable media attention, bringing the issue onto the political agenda. Subsequently, the Swedish government rejected the proposed connection to the national gas grid, arguing that it was not aligned with Sweden's long-term climate objectives and the principles of the Swedish climate policy framework.

The second case concerned the proposed expansion of a refinery on the west coast of Sweden. According to one of the participants, the investment was partly motivated by the introduction of the Sulfur Directive, as the company wanted to upgrade heavy fuel oil with a high sulfur content, previously used by ships, into higher-quality products. During the permitting process, at least one environmental NGO opposed the project in court. According to one of the interviewees and the

court document, the NGO's primary concern related to potential increases in local environmental impacts, particularly air pollution, although the project's potential contribution to higher greenhouse gas emissions was also raised as concern. According to the interviewee, the case initially received limited media attention but were later covered by a journalist. However, the issue was subsequently framed in the media as a case that would substantially increase Sweden's national CO_2 emissions. As a result, the project attracted widespread national attention and became the subject of public and political debate. Similar to the LMG infrastructure case, questions were raised regarding whether such an investment was compatible with Sweden's long-term climate ambitions and climate policy framework. Before a final court decision was reached, however, the company decided to withdraw its application. The official explanation provided by the company was that the project was no longer considered economically feasible. It was never established whether the decision was influenced solely by economic factors or whether the growing public and political debate surrounding the project's compatibility with Sweden's climate objectives also played a role.

These two examples can illustrate why fuel availability is difficult to represent in only abatement-cost calculations. Furthermore, future alternative fuel prices are uncertain because long-term market data are limited, production costs are often used as proxies, and market outcomes may be affected by non-linear dynamics such as innovation, sudden increases, and price spikes (as discussed in for example **Paper II**). This is particularly relevant for FuelEU Maritime, where increasingly strict GHG-intensity requirements after 2040 could create rapid growth in demand for low-GHG fuels and technologies. Such developments are difficult to model with precision and may be understood as part of the deep uncertainty surrounding future fuel markets (Marchau et al., 2019). Table 10 summarizes the policy instruments that were frequently mentioned in interviews and workshops and that were perceived as relevant for fuel demand, fuel availability, or investment conditions.

Table 10 Policy instruments mentioned in the interviews and workshops as relevant for fuel demand, fuel availability, or investment conditions.

Policy instruments	Geographical coverage	Sector
SECA and IMO sulfur cap	Regional and global	Marine fuel quality and scrubber use
Land Use, Land Use Change and Forestry (LULUCF)	UN/EU	Forestry, agriculture, and biomass availability
EU ETS	EU	Energy-intensive sectors; shipping from 2024
Direct regulations and public procurement	Local, regional, and national	Technology deployment and infrastructure demand
Swedish CO_2 tax	Sweden	Fossil fuel use, with sectoral exemptions
Greenhouse gas reduction quota (Swedish: reduktionsplikt)	Sweden	Road transport; renewable fuel demand
Swedish Climate Policy Framework, including the Climate Act	Sweden	National climate-policy direction
Pump legislation	Sweden	Road-fuel distribution infrastructure
Local carrots and sticks	Local	Shipping incentives, e.g. port and fairway discounts (**Paper I**)

Ship related regional incentives and abatements strategies

The stakeholder material also illustrates that investment decisions are shaped by long-time horizons and by learning processes that begin before policy requirements are formally

introduced. For example, representatives from the Port of Gothenburg described investments in alternative technologies as a long-standing part of the port's development strategy. OPS for RoPax ferries was already operational in 2010, well before later EU requirements on the use of shore-side electricity for some ship categories (described in **Paper II** and **Paper V**). OPS for RoPax ferries was operational in Gothenburg by 2010, well before later EU requirements for shore-side electricity. Such early deployment provides opportunities to identify technical, operational, and institutional barriers before wider regulatory implementation (Geels, 2005). The long interval between early deployment and broader regulation also illustrates the timescales involved in technology diffusion.

As discussed in Section 5.1.2, electrification for suitable short-distance ships and biofuels for some other ships appear to be among the more immediate transition options. This pattern is already visible in battery-electric ferries in Norway (Bach et al., 2020), and Sweden (**Paper I**), but the stakeholder material shows that similar tendencies can also be observed more concretely in the Gothenburg port cluster through ongoing initiatives related to OPS, biofuels, and operational measures. Several current initiatives identified in the Gothenburg port cluster further suggest that abatement strategies differ in cost and emission-reduction potential, a well as in the type of coordination they require. Measures such as OPS expansion, biofuel use, and slow steaming or just-in-time arrival all involve multiple actors and rely on different combinations of infrastructure, fuel supply, operational routines, and commercial incentives. OPS for product tankers, for example, has been assessed, piloted, and is now operational, but only used commercially to a limited extent so far. This indicates that technical feasibility does not automatically translate into large-scale adoption. Similarly, biofuel use was also confirmed to have occurred at a smaller scale already before the introduction of the EU ETS and FuelEU Maritime. According to interviews with local container and product tanker companies, biofuels had been used directly, as blend-in, and through mass-balance arrangements in the region (Ahlström & Johnsson, 2023; Emanuelsson & Fransson, 2023; Fagerbaek Guldstrand & Wall, 2024). This was also supported by interviews with a bunker company and port actors in **Paper IV**. In addition, bunkering of bio-LMG had been explored, and a liquefaction plant linked to the port is scheduled to become operational in 2027. These examples indicate that policy effectiveness depends on whether a measure is economically attractive in a model, whether the relevant actors, infrastructures, and supply chains are sufficiently aligned for investments to occur and for emissions to be reduced in practice.

Time horizon and discount rate

The stakeholder material also showed that actors differed substantially in how they defined short- and long-term investments. For some private actors, a ten-year investment horizon could already be considered long term, whereas port-related infrastructure investments could be assessed over several decades, in some cases up to around 50 years. This difference became particularly visible in the workshop exercise in connection to **Paper IV** where participants were asked to indicate investment horizons and return requirements for major investments. The discussion that followed showed that some participants were surprised by the size of the differences between actors.

The same pattern was also visible in relation to return requirements and discount rates. In the workshop exercise, some actors indicated rates below 5%, while others used rates above 10% when assessing major investments. This variation shows that actors do differ in how far into the future they evaluate an investment, how strongly they discount future costs and benefits. For

capital-intensive abatement measures and fuel infrastructure, such differences can substantially change whether an investment appears economically feasible.

These differences matter because the chosen time horizon and discount rate strongly influence the annualized cost of an abatement measure and, therefore, whether it appears economically feasible. They also illustrate that investment assumptions are not only technical parameters. They reflect different actor positions, ownership structures, risk perceptions, and expectations about future markets and regulations. Variation in investment horizons and return requirements thereby introduces ambiguity into policy evaluation: different actors may apply different evaluative criteria to the same technology or policy, leading to different conclusions about what is economically reasonable or desirable (Klinke & Renn, 2002). In the context of this thesis, this difference is illustrated for the European fleet in Section 5.3.1, which examines socio-economic costs (RQ3).

5.3. Perspective on policy evaluation: external costs and uncertainty

Section 5.3.1 addresses RQ3 by examining how the interpretation of policy effectiveness changes when the assessment is extended from a private economic to a socio-economic cost perspective, including external costs. Sections 5.3.2–5.3.4 then examines uncertainties related to ship activity and emissions, environmental impact modelling and monetary valuation. The final section, i.e. 5.3.5, distinguishes these sources of uncertainty from ambiguity arising from alternative objectives, system boundaries and time perspectives.

5.3.1. Socio-economic cost perspective vs private economic cost perspective

Figure 12 shows the estimated external costs associated with European fleet, including climate change, marine ecotoxicity from scrubber-water discharges, and air pollution. These external costs were calculated using the same methodology as in **Paper V**, but for the year 2023 instead of 2040. The external costs are normalized by each ship's annual CO_2 emissions, and the ships are ranked from the highest to the lowest total external cost per tonne of CO_2. The x-axis shows the cumulative annual CO_2 emissions of the ships in this order.

The results show that the total normalized external cost varies considerably across the fleet, from approximately 200 to 640 € per tonne of CO_2. Marine ecotoxicity (resulting from scrubber discharge water) represents the largest individual cost category for ships corresponding to approximately the first 26 Mt of cumulative CO_2 emissions. Beyond this point, the marine ecotoxicity cost drops to zero because the remaining ships do not discharge scrubber water in the modelled scenario.. For most ships, the external cost of air pollution is of a similar magnitude to the climate-change cost. However, the air-pollution cost decreases among the ships with the lowest total external costs, partly because several of these ships use LMG and therefore have lower emissions of some air pollutants (NO_X, $PM_{2.5}$ and SO_X). The ship-level air-pollution results should nevertheless be interpreted with caution. Geographical information was insufficient to determine the share of each ship's emissions occurring inside and outside emission control areas. This particularly affects the allocation of ship-specific NO_X and SO_2 emissions and means that the air-pollution costs are more representative at the fleet level (as presented in **Paper II** and **V**) than for individual ships.

Figure 12 also shows that the 2023 EU ETS allowance price covered only part of the estimated external cost of fleet emissions. The difference is particularly large when air pollution, marine

ecotoxicity, and upstream greenhouse-gas emissions are included, these impacts are not reflected in the allowance price. However, the external cost in Figure 12 should not be interpreted as the benefit for each abatement technology, because the investigated alternatives affect the emission pathways differently (as discussed in **Paper II**). Using battery-electric propulsion and PEMFCs on ships will reduce or eliminate direct emissions when considering only the ship operation as system boundary, but may retain upstream emissions from energy production. Methanol dual-fuel engines retain some direct emissions, while the upstream climate impact depends on the methanol production pathway. For bio-methanol, the normalized avoided external cost (i.e. "benefit") ranges from about 180 to 510 € per tonne of CO This range is used in Figure 13.

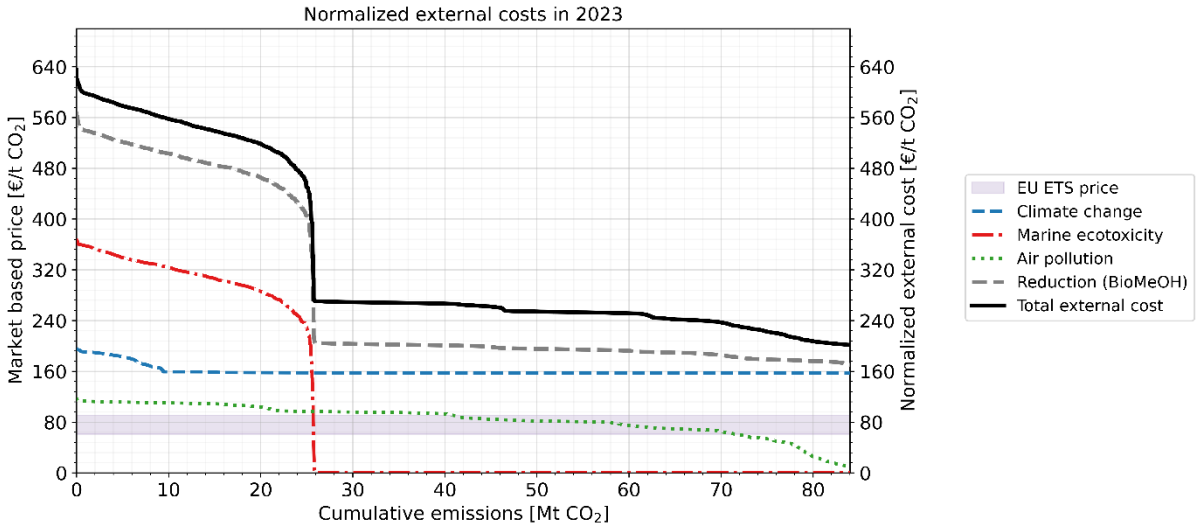


Figure 12. Normalized external costs associated with fleet emissions in 2023 for central estimates from Paper V (Y2 axis). Emissions are ordered according respective external cost for each ship per tonne of CO_2, illustrating the variation in climate, air pollution, and marine ecotoxicity costs across the fleet. The shaded area represents the 2023 EU ETS price (Y1-axis).

Figure 13 combines the results presented in Figure 9 and Figure 12 and combines results from all papers (**Paper I-Paper V**). The EU ETS price range, shown by the purple shaded area, can be compared with the abatement costs calculated from the private-economic perspective (PoP), because both represent economic signals faced by market actors. This comparison indicates whether the carbon price is sufficient to make the fuel switch economically attractive under the assumed market conditions. Similarly, the abatement costs calculated from the socio-economic perspective (SoP) can be compared with the estimated external costs avoided using bio-methanol, shown by the green shaded area. This comparison instead indicates if the resources required for the abatement strategy are smaller than the estimated environmental benefits to society. The two comparisons therefore address different questions: the private economic perspective concerns if market actors have an economic incentive to implement the measure, while the socio-economic perspective assesses if the implementation may increase overall "welfare".

The comparison indicates that emission abatement using bio-methanol may be socio-economically beneficial for a large part of the fleet under the assumptions applied. Where the Bio-MeOH-SoP curve lies below the lower boundary of the external-cost range, the estimated abatement cost is lower than even the lower external-cost estimate, i.e. it would be beneficial to abate for a society perspective. Where the curve lies within the shaded area, the net-benefit depends on the external-cost estimate applied. Only for the final and most expensive part of the

abatement potential does the SoP cost increase rapidly and exceed the central range of estimated external costs.

The figure also shows a substantial difference between the market-based and socio-economic perspectives. From the private perspective, the abatement cost associated with bio-methanol is considerably higher than the 2023 EU ETS price for almost the entire emission-reduction potential. From the social perspective, the abatement cost is lower because different assumptions are used regarding factors in this case the discount rate and economic lifetime, as discussed in **Paper IV** and Section 5.2.2. In addition, the estimated external costs include environmental impacts and upstream greenhouse-gas emissions that are not fully represented by the EU ETS price. The year 2023 is used as a reference year to represent the fleet, prices, emissions, and technologies under the conditions examined in this thesis. Compared with Paper II, where 2035 was used as the future year and Paper V that used 2040. The figure should therefore be interpreted as a theoretical abatement potential rather than as an implementation pathway for a single year. Replacing or rebuilding the relevant ships and expanding the production of bio-methanol would require a longer transition period.

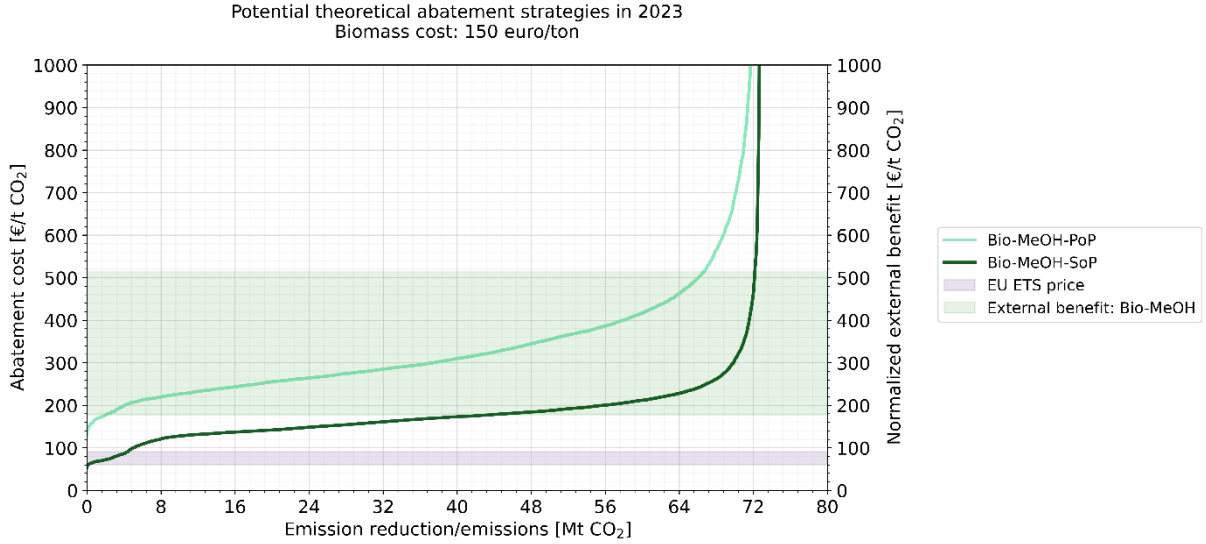


Figure 13. Theoretical CO_2-abatement cost curves under 2023 reference-year conditions bio-methanol (Bio-MeOH), calculated from private (PoP) and social (SoP) perspectives using the central cost assumptions. The x-axis shows cumulative annual CO_2-emission reduction potential across the fleet, and the y-axis shows the corresponding abatement cost per tonne of CO_2. The purple shaded area represents the 5th to 95th percentile range of the 2023 EU ETS spot allowance prices. The green shaded area represents the 5th to 95th percentile range of the external benefit through the use of bio-methanol, normalized by the associated CO_2-emission reduction.

5.3.2. Uncertainty in activity data and emission factors

Uncertainty in ship activity affects estimates of both technology costs and external costs, as activity data are used in a substantial proportion of the calculations. Uncertainty relates to things we don't know and natural variability, such as time at/berth, geographical coverage, port frequency, size of engines and ship itself, fuel consumption/power demand, transported amount of goods, cost of installing and operating an abatement strategy but also to assessing what is the future activities.

Uncertainty in ship activity affects estimates of emissions, technology costs, and external costs, since activity data support a substantial share of the calculations. As demonstrated in **Papers II**

and **V**, epistemic uncertainty relates to incomplete or aggregated information on fuel use, engine characteristics, time at berth, geographical activity, voyage length, power demand, and transported cargo, while aleatory uncertainty relates to variation among ships, routes, and operating conditions. These uncertainties are important when estimating emissions, but their relative importance may decrease when the analysis is extended to environmental impacts and external costs, where uncertainties in impact assessment and monetary valuation can have a greater influence on the results, as discussed in section 5.3.3 and **Paper V**.

The influence of activity-related uncertainty also depends on the outcome considered. Assumptions about voyage energy requirements strongly affect the estimated costs of battery-electric and hydrogen-based options (**Paper II**), whereas fuel-type attribution has a particularly large influence on estimated scrubber-water discharges (**Paper II** and **Paper III**). Future fleet activity, fuel uptake, vessel replacement, and technology costs involve deeper structural uncertainty because neither the developments themselves nor their probability distributions can be estimated reliably. They are therefore addressed through alternative scenarios and sensitivity analyses rather than interpreted as probabilistic confidence intervals (**Papers II** and **V**).

An opportunity for improving shipping policy assessments concerns the empirical basis of emission factors and their uncertainty ranges. As discussed in **Paper II** emission factors used in AIS-based models such as SHIPAIR, STEAM and MASTER are frequently reported and applied without explicit uncertainty ranges, despite their use in research, emission inventories, and policy assessments (Armstrong, 2022; Basso et al., n.d.; Faber et al., 2020; Jalkanen et al., 2016, 2021; Windmark et al., 2017). The EMEP/EEA emission inventory guidebook provides an exception by reporting indicative uncertainty ranges for shipping emission factors (Trozzi et al., 2019). However, several of these ranges originate from Entec (2002) and represent expert-judgement bounds rather than statistical confidence intervals derived from a general and representative set of ship measurements. Their continued use indicates that the empirical basis for ship related emission-factor´s uncertainty has not been updated at the same rate as other details are increasing in ship-emission models.

The uncertainty involved also needs to be interpreted more carefully than as a single numerical range. A numerical range may make sense for real-world variability of aleatory uncertainty type (related to for example engine loads, operating modes, technologies, and maintenance conditions). But when there is bias such as a systematic deviation between regulation-based emission factors for example a widening of symmetric uncertainty interval may not be relevant for the analysis (Frey et al., 2006). This is particularly relevant for NO_X, for which malfunctioning, tampering, or deviations from idealized compliance conditions may result in emissions that differ systematically from regulatory expectations (Comer et al., 2023; Fridell et al., 2023). This affects the interpretation of policy effectiveness because the estimated outputs do not necessarily indicate accurate emission estimates. If the central emission factors are biased, error propagation using the reported uncertainty ranges may produce a precise distribution around an inaccurate result. Updated and operationally representative measurements are therefore necessary to improve central estimates, distinguish real-world variability from knowledge gaps, and identify potential systematic bias.

5.3.3. Uncertainty along the environmental impact pathways

The uncertainty in the marine ecotoxicity damage cost used in **Paper V** only reflects uncertainty in the estimated TBT content of the sediment assumed to be removed through dredging. The range

is derived from the 95% confidence interval of measured TBT concentrations and does not include uncertainty in the monetary valuation or characterization factor. A further limitation concerns the lack of spatial differentiation in the marine ecotoxicity characterization. The ReCiPe characterization factors applied in this thesis in **Papers III and V** use a single factor for each substance and perspective and therefore do not account for geographical differences in substance fate, bioavailability and toxicity. Such differences can influence metal fate, bioavailability, and toxicity through variations in seawater residence time, salinity, pH, dissolved organic carbon, and suspended particulate matter. Using spatially differentiated comparative toxicity potentials for 64 Large Marine Ecosystems representing coastal and continental shelf waters worldwide Dong et al. (2016) found that estimated marine ecotoxicity potentials varied by approximately four orders of magnitude for copper and three orders of magnitude for zinc. These comparative toxicity potentials are not directly comparable with the ReCiPe characterization factors because the methods use different indicators and modelling assumptions. In addition, the analysis by Dong et al. (2016) is limited to metals. Nevertheless, the reported ranges illustrate the potential importance of emission location and receiving-environment conditions for estimated marine ecotoxicity.

For air pollution, spatial variation in atmospheric transport, chemical transformation, and population exposure is incorporated into the GAINS model through source-receptor relationships. These relationships describe how emissions of precursor pollutants from a source region affect $PM_{2.5}$ concentrations in different receptor grid cells. The resulting concentrations are combined with the spatial distribution of the population to estimate population-weighted $PM_{2.5}$ exposure. However, there is also uncertainty in the atmospheric transport and chemical transformation of NO_X and SO_X into secondary $PM_{2.5}$, the resulting population exposure, and the estimated health response. These uncertainties have not been quantified in this thesis. Furthermore, the low, central, and high estimates obtained from the ARP model reflect alternative monetary valuations of premature mortality identified in the literature. They do not include uncertainty in the estimated health effects themselves, such as the relationship between long-term $PM_{2.5}$ exposure and chronic mortality. The uncertainty ranges applied in **Paper V** (for example in Figure 7) therefore cover only part of the impact pathway.

Climate characterization (GWP100) is also subject to uncertainty, for example, the GWP100 of methane is estimated to have an uncertainty range of about 40% around the central characterization factor (Armour et al., 2021). While this uncertainty is not small, it is substantially smaller than the several-orders-of-magnitude spatial variation estimated for marine ecotoxicity potentials. Also, under the current emission estimates, CH_4 and N_2O make relatively small contributions to total CO_2-equivalent for the total shipping fleet compared with CO_2. Their characterization uncertainty therefore has a limited influence on the total estimated climate impact at the fleet level. The contribution of methane could, however, become more important with increased use of methane-based fuels, particularly where methane slip is substantial. Great uncertainty ranges are associated with the net climate effects of short-lived climate forcers, such as NO_X and SO_X (Armour et al., 2021; Åström & Källmark, 2023; Szopa et al., 2023). These effects are not included in the main assessment and are only briefly considered in **Paper V**.

Climate damage cost estimates (here SCC) are also subject to substantial uncertainty. In the alternative damage scenario presented by Barrage and Nordhaus (2024), climate damages at 3 °C warming corresponds to 9% of global output (GDP), compared with 3.1% in the base case. The alternative estimate, based on Howard and Sterner (2017, 2020), uses damage approximately three times larger than in the base case. The resulting social cost of carbon in 2025 increases by 87%, from 78 to 146 USD per tonne of CO_2 2025. This increase is smaller than the change in the

damage coefficient because the social cost of carbon represents the discounted marginal effect of an additional tonne of CO_2 within the DICE model, rather than a direct scaling of total damage. These alternative damage functions are better interpreted as alternative scenarios reflecting the deep uncertainty about future climate damage than as a parametric uncertainty range suitable for conventional error propagation or Monte Carlo analysis.

Overall, also the knowledge base underlying the characterization and valuation of marine ecotoxicity appears much less established than that available for climate change and air pollution. Climate impacts can be characterized and valued using several metrics, models, and damage-cost approaches, although substantial disagreement remains concerning, for example, time horizons, discounting, climate-damage functions, and the treatment of short-lived climate forces. Similarly, the health impacts of air pollution have been studied using different atmospheric dispersion, exposure-response, and valuation models at both local and regional scales. External-cost estimates for climate change and air pollution are also already applied in policy context and, in some cases, partly internalized through policy instruments, such as carbon pricing.

The evidence base for marine ecotoxicity on the other hand is considerably more limited than for climate change and air pollution. Some marine ecotoxicity characterization models exist, but they differ in substance coverage, spatial resolution, modelling assumptions, and impact indicators. ReCiPe was used in this thesis because it provides marine characterization factors for a comparatively broad range of metals and organic substances and expresses their potential impacts in a common midpoint indicator, kg 1,4-DCB-eq. This enables substances released from different shipping-related sources to be aggregated and compared within the same assessment. The evidence base is even more restricted for monetary valuation. Only two independent approaches providing monetary values applicable to marine ecotoxicity expressed in kg 1,4-DCB-eq were identified: the TBT-based willingness-to-pay valuation developed by Noring et al. (2016) and the biodiversity-based valuation in the Environmental Prices Handbook (de Vries et al., 2024). Values appearing in subsequent shipping studies largely represent applications or adaptations of these approaches rather than additional independent valuation evidence (Martínez-López et al., 2022; Ytreberg et al., 2021).

5.3.4. Evidence and uncertainty in monetary damage valuation

The evidential basis of monetary damage valuation is particularly important when external costs are compared with each other or with abatement costs. Monetary valuations may include several types of evidence (see Section 3.4.3), and different types of evidence are often combined within external cost estimates. However, this combination is not always transparent in the reported results. The suitability of a monetary damage estimate for policy evaluation therefore not only depends on its quantified uncertainty, as discussed in the previous section, but also on whether the estimated value is appropriately connected to the magnitude and context of the environmental change under assessment, including its geographical location and ecological characteristics. The valuation methods applied in **Paper V** and this thesis (Table 5) should therefore be understood as providing different and partly complementary forms of evidence, rather than forming a simple hierarchy in which one method is necessarily better than another.

Stated preference methods address an important limitation of empirically anchored approaches because they can capture non-use values that are not expressed through observed market behavior (Börger et al., 2014). This is particularly relevant for marine ecosystems, which may be valued for their continued existence, or protection independently of their direct use. Excluding such values would not make the assessment neutral but could systematically underrepresent

environmental effects that are not reflected in markets or associated with direct use. However, the applicability of stated preference estimates depends on how respondents understand and respond to the environmental change presented to them. A central consideration is whether willingness to pay responds to the scope of that change. The WTP responses may sometimes not increase proportionally with the magnitude of an environmental improvement. If WTP changes little when the magnitude of the improvement changes substantially, the resulting estimate may be weakly connected to the physical scope of the effect (Whitty, 2012). The estimate may then instead partly reflect a general preference for environmental protection, the framing of the valuation scenario, or respondents' familiarity with the issue. Such responses may still provide meaningful evidence of public concern or support for environmental protection, but they are more difficult to interpret and apply as unit damage costs for different quantities of environmental impact.

These concerns are particularly relevant to the marine ecotoxicity estimates applied in **Papers III** and **V**, where the monetary estimate is constructed through several linked analytical stages. The impact pathway extends from emissions of individual substances through environmental fate, exposure, toxic effects, and ecosystem damage to final monetary valuation. The assessment includes only substances for which the required characterization factors are available, while the final monetary values are transferred from previous valuation studies. Uncertainties and potential mismatches may therefore arise at several stages, including between the emitted substances and the available characterization factors, between characterized toxicity potentials and ecological damage, and between the environmental change valued in the original study and that represented in the present assessment. Even if the preceding stages of the impact pathway were represented accurately, it would remain necessary to consider what respondents' stated WTP represents. Familiarity and public attention may influence how respondents interpret the environmental change presented to them. Respondents may, for example, value damage associated with TBT, as examined by Noring et al. (2016), differently from comparable ecological damage associated with less familiar hazardous substances. This example does not imply that familiarity necessarily determines WTP, but illustrates that a transferred monetary estimate may be connected both to the ecological change and to the specific substance, information, and context presented in the original valuation study. The relevant criterion is therefore not whether one method reveals respondents' "true" values, but whether the resulting estimate is sufficiently connected to the environmental change and decision context in which it is applied.

These limitations matter because monetary estimates are used not only to describe environmental damage, but also, as in this thesis, to compare policy instruments and abatement strategies. Such comparisons address a conditional normative question: given a specified environmental objective and set of conditions, which policy instrument or abatement strategy ought to be selected (Niiniluoto, 1993)? The answer depends both on normative choices concerning which consequences to include and how to value them, and on the evidence underlying the estimated costs and effects. Monetization places external costs and abatement costs on a common numerical scale (Aldred, 2006; Craswell, 1998; Reiff, 2014), but does not make their evidential foundations, assumptions, or uncertainties equivalent. The use of a common monetary unit may therefore mask differences in how closely each estimate is connected to the environmental change it is intended to represent.

The marine ecotoxicity estimates should therefore not be interpreted as establishing a single well-defined monetary value, as also illustrated in **Paper V**. The estimates indicate that scrubber-water

discharge may generate societal damage that is not reflected in shipowners' private costs, but the exact magnitude remains dependent on assumptions and data throughout the impact pathway. When external costs are compared with abatement costs, greater analytical weight should be given to estimates that are demonstrably responsive to the magnitude, location, and ecological context of the environmental change. Stated preference estimates remain important where non-use values would otherwise be excluded, but their application requires explicit consideration of scope sensitivity, benefit transfer, validity, and possible systematic bias. Further development of marine ecotoxicity damage costs requires improvements throughout the impact pathway. This includes more substance- and location-specific characterization factors, clearer links between characterized toxicity potentials and ecological damage, and updated stated preference studies in which the environmental change is understandable to respondents and sensitivity to scope is explicitly tested. New valuation studies alone are insufficient if the environmental outcomes presented to respondents cannot be linked to the emissions and characterization factors used in the assessment.

5.3.5. Ambiguity and the interpretation of policy effectiveness

Lueddeckens et al. (2020) note that choices of time horizon are sometimes described in the literature as subjective or arbitrary. However, describing choices associated with ambiguity as arbitrary risks overlooking the theoretical, ethical, and methodological arguments on which they are based. The environmental objectives presented in Table 2 are, for example, normative rather than arbitrary because they express judgements about which environmental outcomes should be pursued and prioritized. Such judgements cannot be resolved solely through additional empirical evidence. Therefore, the interpretation of policy effectiveness depends partly on the evaluative perspective adopted. It is therefore important to make the assumptions underlying different evaluative perspectives and their implications for the results transparent. In **Paper V**, the impact of ambiguity is addressed by presenting results under alternative time horizons for marine ecotoxicity, discounting assumptions for climate impacts, and by comparing the value of life years lost (VOLY) and value of a statistical life (VSL) approaches to the valuation of mortality. These alternative valuation metrics are further discussed in this chapter.

The choice between VOLY and VSL illustrates ambiguity in the monetary valuation of health impacts. VSL values small changes in mortality risk based on individuals' willingness to pay to reduce the risk of premature death, while VOLY values the associated change in life expectancy. VOLY therefore explicitly differentiates between impacts involving different numbers of life-years lost. VSL does not directly account for remaining life expectancy and therefore assigns the same value to avoided mortality risks affecting people with substantially different expected remaining lifetimes, such as an older person and a younger person. In **Paper V**, using VSL instead of VOLY increases the estimated external cost of air pollution by approximately a factor of four. The central estimates for VOLY and VSL in the ARP model are derived from the same WTP survey (the *NEWEXT* survey) (Holland et al., 2013). Consequently, the differences between the estimates do not arise from the use of separate valuation studies conducted with different respondents or in different geographical contexts, but from how the same underlying survey data are processed and applied.

Another source of ambiguity concerns how future environmental impacts are included in the assessment. In **Paper V**, external costs are exemplified using alternative time horizons and social discount rates. Extending the time horizon for marine ecotoxicity from 20 to 100 years increases the estimated external costs associated with scrubber-water discharges and antifouling paints by

factors of approximately four and six, respectively. Climate-related external costs are similarly sensitive to the social discount rate. The estimated costs increase from approximately 10 B€yr^{-1} at a discount rate of 3% to 20 B€yr^{-1} at 2% and more than 40 B€yr^{-1} at 1%. These differences do not primarily reflect uncertainty about an empirically measurable parameter. Rather, the time horizon determines which future impacts are included, while the discount rate determines the weight assigned to those impacts. The choice of discount rate therefore relates directly to intergenerational justice. The results are consistent with what is argued in Lueddeckens et al. (2020), who describe the selection of time horizons and discount rates as value-dependent choices for which no clear consensus exists. Their influence should therefore be made transparent through sensitivity or scenario analysis rather than represented only by a single central estimate.

In the central case in **Paper V** and in Figure 13**,** a social discount rate of 3% is applied. This rate should not be interpreted as a 3% pure rate of time preference. The social discount rate can be separated into a pure rate of time preference and a consumption-related component that depends on expected consumption growth and the declining marginal utility of consumption (derived from the Ramsey equation) (Stern, 2006). As an example: the Stern Review adopts a near-zero pure rate of time preference (0.1 %), principally on the ethical grounds that the welfare of future generations should not receive less weight (Stern, 2006). This does not imply a zero total social discount rate. Future consumption may still be discounted if future generations are expected to be wealthier, and the marginal utility of additional consumption consequently declines.

A near-zero pure rate of time preference may be normatively justified if the welfare of future generations is not considered less important only because it occurs later. However, the resulting valuation of future environmental damage may imply substantial costs for households and companies if it is translated directly into current policy. While such a policy is practically and politically feasible it also depends on how its costs are distributed, when they occur, and the constraints faced by decision-makers. A normatively preferred outcome therefore does not by itself identify an implementable policy strategy. Consequently, a theoretically preferred strategy will not necessarily provide practical guidance for actual decisions unless its implications can be presented in an understandable form and translated into alternatives that decision-makers can realistically consider under existing institutional, informational, and financial constraints (Simon, 1976).

Following Niiniluoto (1993), a recommendation may therefore be valid relative to its **stated ethical objective** but remain incomplete if the conditions connecting that **objective to real-world** resource allocation are not made explicit. If the objective is to select a socially beneficial abatement strategy, an assessment based on long-term social welfare may identify the preferred strategy.

Another important aspect of time is that different time horizons for the characterization factors are applied to marine ecotoxicity and climate change in this thesis. A 20-year horizon is used for marine ecotoxicity, while climate impacts are characterized using GWP100. The use of different time horizons for climate change and marine ecotoxicity can be justified by differences in the impact pathways and the policy contexts in which the results are applied. The shorter time horizon for marine ecotoxicity reflects the policy objective of achieving Good Environmental Status and is also more consistent with the relatively short-term change of environmental impact valued in the Good Environmental Status (GES) scenario of Noring et al. (2016). For climate

change, a longer horizon is relevant because CO_2 emissions have long-term effects and climate policies are generally evaluated over longer periods.

The use of impact-category-specific time horizons is not unique. Different impact pathways already contain different temporal representations. For example, the characterization of NO_X emissions through changes in annual mean $PM_{2.5}$ concentrations represents the response to changes in annual emissions and does not use an explicit multi-year integration horizon. Compare that to GWP_{100} that integrates radiative forcing over 100 years. The selected temporal representation therefore reflects the purpose and temporal characteristics of each impact pathway rather than a requirement to apply an identical time horizon across all impact categories. The resulting external costs can still be compared and aggregated, but the different temporal assumptions should be made explicit when interpreting the results.

The findings of this thesis indicate that policy evaluations should distinguish between uncertainty in data and models and ambiguity arising from alternative objectives, system boundaries, time horizons, and valuation perspectives. Evaluations should clearly state what reported uncertainty ranges include and exclude, and examine whether alternative assumptions affect the conclusions. This would make the conditions under which a policy is considered effective more transparent.

6. Conclusions

6.1. What does each policy instrument do on its own (RQ1)?

The four cases show how different policy mechanisms influence shipowners by changing either the private cost of available abatement strategies, the environmental performance that must be achieved, or the set of compliance options from which shipowners can choose. The broad distinction between market-based (EU ETS, FuelEU and discounts) and direct regulation (sulfur directive) is useful, but insufficient for explaining the resulting response. The response is shaped by the "strength" of the economic incentive or compliance requirement, its timing and predictability, the ships and operations included within its scope, and the technical and operational alternatives available for compliance.

The environmentally differentiated port and fairway fees provide an "encouraging" but limited reward (**Paper I**), which is small relative to the costs of the assessed measures and therefore unlikely to trigger substantial investment on its own. The EU ETS in **Paper II,** on the other hand, increases the cost of GHG emissions, but its effect varies across ships and technologies because abatement costs and technical applicability are heterogeneous. FuelEU Maritime (**Paper V**) creates a progressively stricter requirement for the GHG intensity of onboard energy, but its design with a stepwise trajectory, influence whether compliance is achieved through incremental fuel substitution, operational changes, or longer-term technology investment. In contrast, the IMO sulfur limit imposes a binding requirement that necessitates a response, while allowing shipowners to choose between alternative compliance strategies (**Paper III**).

From an effectiveness perspective, the cases demonstrate different relationships between policy design and environmental outcomes. The sulfur limit imposes an emission requirement and has resulted in reductions in the targeted SO_2 emissions. FuelEU Maritime establishes a required GHG-intensity trajectory, although the resulting fuel mix and technological response depend on how shipowners use the available compliance mechanisms. The EU ETS and the Swedish environmental discounts both improve the relative economic position of lower-emission alternatives, but their ability to induce additional investment depends on whether the resulting price signal is sufficiently large relative to ship-specific abatement costs.

This comparison indicates that policy "strength" alone does not determine the resulting technological response. Broad and flexible instruments may accelerate cost-effective compliance, but they also tend to favor options that are already relatively mature or inexpensive in the short term. This pattern is observed for the EU ETS (**Paper II**), FuelEU Maritime (**Paper V**), and environmentally differentiated discounts (**Paper I**). Policy instruments can direct attention towards specific technologies or environmental pressures, but their influence remains limited when the financial incentive is small, unstable, or available only to a restricted share of operations. In the short term, compliance may be achieved through low-hanging-fruit options, such as drop-in or blended biodiesel, or through the introduction of relatively small quantities of liquefied biomethane. However, the analyses presented here indicate that substantially larger quantities of alternative fuels will be required from 2040 onwards, increasing from approximately 10 to 13 million tonnes of MGO_{eq} in 2040 to 15 to 26 million tonnes in 2045 and 20 to 32 million tonnes in 2050. The policy instrument provides limited incentives for the rapid expansion of alternative-fuel production implied by these estimates. Additional constraints may arise because existing ships may lack the onboard technologies required to use these fuels.

The current design of individual environmental policy instruments influences shipowners by changing the relative costs, requirements, and available choices associated with abatement. Whether this influence results in investment depends on whether the policy **sufficiently** improves the private investment case for the relevant ship and operation. The four cases further show that a policy instrument may successfully induce compliance without necessarily promoting the option that reduces the largest range of environmental pressures.

6.2. What happens when policies interact in a system (RQ2)?

The results identify three forms of interaction: 1. interactions among policy instruments, 2. interactions across environmental pressures, and 3. interactions between policy implementation in the larger technological and institutional system.

Firstly, the quantitative fleet-level analysis (based on work in **Paper II**) provides a complementary illustration of the policy-system perspective by illustrating how policy instruments can alter each other's effects on investment incentives and environmental outcomes. When EEDI-related efficiency improvements, a hypothetical global ban on scrubber-water discharge, and an extended scope of the EU ETS are introduced sequentially, they change both the reference conditions against which the alternative fuel (in this case illustrated for e-MeOH) is evaluated and the emissions included within the policy scope. The scrubber ban increases the cost of continued fossil-fuel use for affected ships, thereby reducing the relative abatement cost of switching to e-MeOH, while the extended EU ETS further alters the economic incentive and increases the emission-reduction potential covered by the policy. However, the relatively limited fleet-level effect of the scrubber ban on CO_2 abatement costs also shows that the magnitude of a policy interaction depends on how many ships and operations are affected. The analysis therefore demonstrates that policy interactions are not necessarily additive: their combined effect depends on policy scope, the available compliance options, the reference technology or fuel, and the composition of the regulated fleet.

Secondly, the interaction also extends across environmental dimensions. Allowing scrubbers can reduce SO_2 emissions to air while maintaining the use of HFO and generating discharges to water (**Paper III** and **Paper V**), while restricting scrubber use may alter fuel costs, direct GHG emissions, and the relative attractiveness of low-GHG alternatives (**Paper II**). The environmental outcome of a policy combination therefore depends on the targeted emission, how the available compliance strategies redistribute emissions and costs. Similar relationships arise when policies targeting GHG emissions (**Paper V**) influence the choice of fuels and technologies and thereby indirectly other environmental outcomes. The interaction across environmental pressures therefore arises not because the pollutants directly affect one another, but **because the same fuel, technology, or operational decision affects several emission pathways simultaneously**. Policy combinations can consequently reduce one environmental pressure while increasing, maintaining, or relocating another.

The policy-system perspective goes beyond comparisons of numerical emissions and costs by considering also how policy outcomes are shaped through interactions among actors, institutions, technologies, and geographical levels. The stakeholder material shows that investment decisions depend on the formal design and economic strength of a policy instrument, as well as on fuel availability, infrastructure, permitting processes, public acceptance, and coordination among actors who control different parts of the transition. These interactions can therefore alter both the implementation of policy instruments and their resulting environmental outcomes.

The stakeholder material (**Paper IV**) illustrates features that are consistent with polycentric governance (Gritsenko, 2017; Ostrom, 2017), in which multiple public, private, and civil-society actors influence governance at different levels and scales. Companies may respond proactively to expected regulation or attempt to influence its development, while NGOs, activists, political actors, ports, and infrastructure providers may affect investment conditions, public debate, and policy decisions. The interviews also identify local niches in which novel technologies and operational arrangements are tested. From a technological innovation systems perspective, these initiatives are relevant because local experimentation can support technical and institutional learning before solutions are adopted more broadly (Bach et al., 2020, 2021; Geels, 2005). Polycentric governance and technological innovation systems therefore capture related but distinct mechanisms: the former concerns the distribution of authority and influence, whereas the latter helps explain how technologies develop through experimentation, learning, and interaction among actors.

Despite their different roles and interests, the interviewees broadly accepted net-zero emissions around 2050 as a central objective. However, they differed in their interpretations of feasible transition pathways, the allocation of responsibility, and if current developments were sufficient to achieve this objective. This distinction is important because agreement on an overall goal does not imply agreement on which policies or technologies should be prioritized. Since policy-instrument evaluation is normative and concerned with how societal objectives ought to be achieved (Niiniluoto, 1993), stakeholder participation can strengthen the social relevance and legitimacy of the assessment when participants are given a meaningful opportunity to influence its framing, assumptions, or interpretation (McGookin et al., 2021; Salter et al., 2010; Stalpers et al., 2008).

The results show that combinations of environmental policy instruments influence investment decisions and environmental outcomes not only through changes in costs and emission incentives. Outcomes are also influenced by interactions with infrastructure, fuel supply, technological learning, public acceptance, and political processes. Policy effectiveness therefore depends partly on the alignment and interaction of policy instruments, but also on coordination among actors operating at different levels and on the capacity of the broader policy system to support experimentation and learning over time.

6.3. How should policy effectiveness be interpreted (RQ3)?

Including external costs adds to the interpretation of several results related to **RQ1** and **RQ2**. The policy instrument investigated in **Papers I**, **II**, and partly **V** FuelEU Maritime, may currently and in the coming years be too weak to make the analyzed abatement strategies attractive from a private economic perspective. However, the results of this thesis and **Paper V** indicate that the estimated environmental benefits may exceed the socio-economic costs of these measures. Under the assumptions applied in the central cost case, the private abatement cost remains above the EU ETS allowance price for almost the entire theoretical emission reduction potential, while the socio-economic abatement cost is lower than the estimated avoided external costs for a substantial share of the fleet. The absence of a competitive investment case therefore does not imply that an abatement strategy is inefficient from a societal perspective. Instead, the difference between the private and socio-economic results indicates that the environmental damage is not fully internalized in the costs faced by shipowners. Stronger or complementary policy instruments may therefore be socio-economically justified in relation to the current European environmental objectives. However, this conclusion rests on several analytical steps and assumptions concerning activity data, costs, emission reductions, impact characterization, and monetary valuation. The

following discussion distinguishes between variation related to uncertainty in these inputs and relationships, and variation arising from alternative normative perspectives and interpretations.

The uncertainty ranges reported in this thesis cover only some parts of the calculations and should not be interpreted as representing the total uncertainty of the results. For example, the ranges applied in **Paper V** do not cover all uncertainties in the emission estimates, characterization factors, impact pathways, and monetary valuations. It is also important to distinguish between variability among ships and uncertainty that is common to the whole fleet. Uncertainty in a common emission factor or model does not decrease simply because the same factor is applied to thousands of ships. If the central estimate is systematically inaccurate, error propagation or Monte Carlo simulation may instead produce a narrow range around an inaccurate result. Some uncertainties, such as alternative climate damage functions, may be better examined through scenarios than through probability distributions. Also, the data availability differs between the environmental impacts considered. It is also worth noting that the impact pathways for climate change and air pollution are relatively well studied, although important uncertainties remain, whereas the data available for marine ecotoxicity is more limited.

Differences in the results also appear from ambiguity concerning the objective, time perspective, and valuation principle used in the assessment. In **Paper V**, this is illustrated by the choice between VOLY and VSL, alternative time horizons for marine ecotoxicity, and different social discount rates for climate damage. Using VSL instead of VOLY increases the estimated external cost of air pollution by approximately a factor of four, even though both estimates are derived from the same underlying WTP survey. Increasing the time horizon or applying a lower discount rate similarly increases the weight given to future environmental damage. These differences cannot be solved only through additional empirical data because the alternatives represent different ways of defining and valuing the relevant consequences (normative values). A policy or abatement strategy may therefore appear effective depending on the environmental objective, valuation principle, and time perspective applied. These choices should be stated and justified in relation to the purpose of the evaluation, and alternative assumptions should be presented when they affect the results. Despite these differences, the results in **Paper V** indicate that stronger or complementary policy instruments may be socio-economically justified for some abatement strategies across several of the valuation perspectives and assumptions examined.

Future improvements can be made at several stages along the impact pathway. However, in the policy evaluation context the results suggest that research resources should increasingly be directed towards understanding and valuing environmental impacts, particularly marine ecotoxicity, rather than only towards further refinement of shipping activity data. Further development of marine ecotoxicity damage costs requires stronger empirical links and greater resolution throughout the impact pathway. This includes more substance- and location-specific characterization factors, clearer links between characterized toxicity potentials and ecological damage, and updated stated preference studies in which the environmental change is understandable to respondents and sensitivity to scope is explicitly tested. New valuation studies alone are insufficient if the environmental outcomes presented to respondents cannot be linked to the emissions, characterization factors, and ecological effects represented in the policy assessment.

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