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Cost-Benefit Analysis of Alternative Greywater Management Scenarios from "Cradle to Grave"

A research project out within the Swedish Transport Administration's industry program Sustainable Shipping, operated by Lighthouse, published December 2025

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Cost-Benefit Analysis of Alternative Greywater Management Scenarios from "Cradle to Grave".

A comprehensive assessment of the Net Societal Benefit of ten ship-generated greywater management pathways in the Baltic Region, from generation to final disposal, using the Trelleborg wastewater management system as a case study. The outcome would inform policy and support sustainable Shipping in the Baltic Region.

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Summary

This study evaluates the economic and environmental performance of alternative greywater management strategies for passenger ships operating in the Baltic, using the Port of Trelleborg as a representative case. Greywater (GW), comprising water from showers, sinks, galleys, and laundries, remain unregulated under MARPOL Annex IV, despite its substantial loads of nutrients, organic matter, microplastics and metals. As shipping traffic grows, especially in enclosed and sensitive marine areas such as the Baltic Sea, the lack of international regulation creates uncertainty for ports and shipowners on how GW should be handled. This study provides the first integrated cost-benefit analysis (CBA) of ship-to-port greywater reception, land-based treatment, and reuse in the Baltic Sea context.

Ten management scenarios were examined, ranging from direct discharge at sea (SC1) to advanced land-based treatment with full or partial reuse (SC5B$_1$ – SC5B$_5$). The analysis quantifies (1) societal costs, including capital and operating expenditure for shipboard systems, port reception facilities (PRF), and municipal wastewater treatment plants (MWTP/PWTP); (2) stakeholder costs and benefits for ships, ports, and municipalities; and (3) environmental benefits (EB) expressed monetarily using pollutant shadow prices for nutrients, organic matter, suspended solids, metals, and microplastics. Reuse benefits were also estimated based on avoided potable-water production and distribution.

The societal cost analysis shows that all treatment options are more expensive than direct discharge at sea. This is expected, since SC1 avoids high costs associated with the use of port reception facilities and GW treatment infrastructure. Among treatment-only scenarios, SC3 (PRF → MWTP) performs better economically than SC4 (PRF → PWTP → MWTP), but neither achieves a positive net social benefit due to limited environmental gains relative to cost. Adding reuse significantly improves performance: all reuse scenarios achieve higher total environmental benefits and lower net welfare losses than non-reuse scenarios. The best-performing option is SC5B$_1$ (100% reuse), which transforms treated greywater into a substitute for potable water used in toilet flushing and other non-potable uses. However, even SC5B$_1$ remains slightly negative in Net Societal Benefit (NSB) under current Swedish freshwater prices, highlighting the low economic value of freshwater in northern Europe.

Stakeholder results reveal a structural asymmetry: ships face the largest costs under all treatment and reuse scenarios, while ports experience modest cost changes and municipalities benefit mainly from avoided loading rather than direct financial gains. This misalignment means that socially preferable options are unlikely to be adopted without targeted incentives, regulatory intervention, or revised port pricing schemes under the EU PRF Directive. Environmental benefits are highest for scenarios involving advanced treatment and reuse, with COD, nitrogen, zinc and microplastics dominating the shadow-price valuation.

Sensitivity analysis shows that the economic viability of reuse is most dependent on the assumed shadow price of freshwater. Doubling the freshwater value nearly halves the welfare loss of SC5B$_1$, while changing reuse-treatment cost by ±20% has only a minor effect. This confirms that reuse becomes increasingly attractive in contexts with higher scarcity, higher municipal tariffs, or greater environmental penalties for water abstraction.

Overall, the results demonstrate that a shift toward land-based treatment and reuse can support Baltic Sea protection goals under HELCOM's Baltic Sea Action Plan (BSAP) and EU Marine Strategy Framework Directive (MSFD). However, cost-sharing mechanisms, incentives for early adopters, and harmonized regional policies will be essential for implementation. The study concludes that circular water management at ports is both technically feasible and environmentally beneficial, and, with appropriate policy instruments, can become an economically viable part of sustainable maritime wastewater governance.

Sammanfattning

Denna studie utvärderar den ekonomiska och miljömässiga prestandan hos alternativa strategier för hantering av gråvatten (GW) från passagerarfartyg som trafikerar Östersjön, med Trelleborgs hamn som representativt fall. Gråvatten, som består av vatten från duschar, handfat, kök och tvättutrymmen, är fortfarande oreglerat enligt MARPOL bilaga IV, trots dess betydande belastning av näringsämnen, organiskt material, mikroplaster och metaller. I takt med att sjöfartstrafiken ökar, särskilt i slutna och känsliga havsområden som Östersjön, skapar avsaknaden av internationell reglering osäkerhet för hamnar och redare kring hur GW bör hanteras. Denna studie presenterar den första integrerade kostnads–nyttoanalysen (CBA) för hantering av gråvatten från fartyg i hamn, landbaserad rening och återanvändning i Östersjön. Tio olika scenarier utvärderades, från direkt utsläpp till havet (SC1) till avancerade landbaserade behandlingslösningar med fullständig eller delvis återanvändning av vattnet ($SC5B_1$-$SC5B_5$). Analysen kvantifierade (1) samhällskostnader, vilket inkluderar investerings- och driftskostnader för fartygsbaserade system, mottagningsanläggningar för gråvatten i hamnarna (PRF), kommunala avloppsreningsverk (MWTP) samt hamnbaserade reningsanläggningar (PWTP) (2) kostnader och nyttor för berörda aktörer, fartyg, hamnar och kommuner, samt (3) miljönyttor (EB) uttryckta i monetära termer med hjälp av skuggpriser för utsläpp av föroreningar i form av näringsämnen, organiskt material, suspenderade ämnen, metaller och mikroplaster. Nyttan med återanvändning uppskattades också utifrån undvikt produktion och distribution av dricksvatten.

Den samhällsekonomiska kostnadsanalysen visar att alla behandlingsalternativ är dyrare än direkt utsläpp till havs, vilket speglar både den begränsade användningen av hamninfrastruktur och de högre reningskraven för GW vid scenariot med utsläpp direkt till havs. Bland scenarierna med enbart rening presterar SC3 (PRF → MWTP) ekonomiskt bättre än SC4 (PRF → PWTP → MWTP), men inget av dem uppnår en positiv samhällsekonomisk nettonytta på grund av begränsade miljövinster i förhållande till kostnaden. Införande av återanvändning förbättrar resultatet avsevärt då alla återanvändningsscenarier ger större samlade miljönyttor och lägre välfärdsförluster än scenarier utan återanvändning. Det alternativ som presterar bäst är $SC5B_1$ (100 % återanvändning), där renat gråvatten ersätter dricksvatten som används för toalettspolning. Även $SC5B_1$ är dock svagt negativt i samhällsekonomisk nettonytta (NSB) givet dagens svenska priser på färskvatten vilket belyser det låga ekonomiska värdet av färskvatten i norra Europa. Intressentanalysen visar en strukturell asymmetri, där fartygen står för de största kostnaderna i alla behandlings- och återanvändningsscenarier, medan hamnarna endast påverkas av måttliga kostnadsförändringar och kommunerna främst gynnas genom minskad belastning snarare än direkta finansiella vinster. Denna obalans innebär att samhällsekonomiskt önskvärda alternativ sannolikt inte kommer att införas utan riktade incitament, reglerande styrmedel eller reviderade hamntaxor inom ramen för EU:s PRF-direktiv. Miljönyttorna är störst i scenarier med avancerad rening och återanvändning, där COD, kväve, zink och mikroplaster dominerar skuggsprisvärderingen.

Känslighetsanalysen visar att den ekonomiska lönsamheten för återanvändning är mest beroende av det antagna skuggpriset på sötvatten. En fördubbling av färskvattenets värde minskar välfärdsförlusten i $SC5B_1$ med nästan hälften, medan en förändring av återanvändningsrelaterade reningskostnader med ±20 % endast har en liten effekt. Detta bekräftar att återanvändning blir allt mer attraktiv i sammanhang med högre vattenbrist, högre kommunala taxor eller större miljömässiga sanktioner för vattenuttag.

Sammantaget visar resultaten att en övergång till landbaserad rening och återanvändning kan stödja Östersjöns miljömål enligt HELCOM:s Baltic Sea Action Plan (BSAP) och EU:s havsmiljödirektiv (MSFD). För att detta ska kunna genomföras krävs dock kostnadsdelningsmekanismer, incitament för tidiga aktörer samt harmoniserade regionala policyramverk. Studien drar slutsatsen att cirkulär vattenhantering i hamnar både är tekniskt genomförbar och miljömässigt fördelaktig och, med lämpliga styrmedel, kan bli en ekonomiskt hållbar del av en långsiktigt hållbar avloppsvattenhantering inom sjöfarten.

Acronyms and Abbreviations

ADEC	Alaska Department of Environmental Conservation
ASTP	Advanced Sewage Treatment Plant
ASTS	Advanced Sewage Treatment System
BOD	Biochemical Oxygen Demand
BSAP	Baltic Sea Action Plan
Cd	Cadmium
CLIA	Cruise Lines International Association
COD	Chemical Oxygen Demand
Cr	Chromium
Cu	Copper
EMSA	European Maritime Safety Agency
EU	European Union
GW	Greywater
HELCOM	Helsinki Commission
IMO	International Maritime Organization
MEPC	Marine Environment Protection Committee of IMO
Mn	Manganese
MSFD	Marine Strategy Framework Directive
MWTP	Municipal Wastewater Treatment Plant
N	Nitrogen
Ni	Nickel
NSB	Net Societal Benefit
P	Phosphorus
Pb	Lead
PET	Polyethylene terephthalate
PP	Polypropylene
PRF	Port Reception Facility
PSSA	Particularly Sensitive Sea Area
PWTP	Port-based Wastewater Treatment Plant
Revaq	Renare Vatten med Avloppsslam som Kretslopp
TSS	Total Suspended Solids
UWWD	Urban Wastewater Treatment Directive
Zn	Zinc

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1 - Introduction

1.1 - Shipping and its environmental impacts

Shipping plays a vital role in global trade and passenger mobility, with myriads of goods transported by sea and millions of passengers carried annually across regions such as the Baltic Sea (IMO, 2020). While shipping is often considered an energy-efficient mode of transport per tonne-km, its environmental footprint is significant and multifaceted. Conventional concerns have long centered on air emissions, including sulphur oxide (SO_x), nitrogen oxides (NO_x), particulate matter, and greenhouse gases (GHG), which contribute to acidification, eutrophication, climate change, and adverse health outcomes (Corbette et al., 2007; Smith et al., 2015). In response, international regulations such as MARPOL Annex VI have imposed stricter emission control areas (ECAs), including the Baltic Sea, where sulphur content in fuel is capped at 0.1%. Beyond air pollution, shipping activities generate a range of marine and coastal impacts, notably through ballast water discharges introducing invasive species, underwater noise affecting marine mammals, accidental oil and chemical spills, and solid waste disposal, wastewater (bilge water, scrubber water, grey water, blackwater), antifouling paints and stern tube oil leakage (HELCOM, 2018a; UNCTAD, 2022; Jalkanen et al., 2023). For instance, about 505,000 m^3 bilge water, 312 million m^3 scrubber water, 5.4 million m^3 greywater, 0.5 – 1.4 million m^3 sewage (black water), 4,740 m^3 leaked stern tube oil, 569 tonnes antifouling paint were potentially discharged into the Baltic Sea in 2022 (Jalkanen et al., 2023).

More recently, attention has shifted towards the role of wastewater discharges, including blackwater, greywater, bilge water, and scrubber water, as emerging contributors to marine pollution (Mujingni et al., 2024). Greywater derived from sinks, showers, galleys, and laundry, can contain nutrients, detergents, organics, pathogens, and synthetic microfibers, posing risks to sensitive marine ecosystems (Baresel & Olshammar, 2019; Folbert et al., 2022; Mujingni et al., 2024). This is of special concern in semi-enclosed and low-flushing seas like the Baltic, where pollutant accumulation is amplified and ecological thresholds are under stress (HELCOM, 2021a). In addition to ecological effects, wastewater discharges can undermine economic sectors dependent on clean marine environments, including fisheries, aquaculture, and coastal tourism (Campanale et al., 2020). Thus, shipping, while indispensable for global connectivity, remains a critical sector where sustainable management of environmental impacts is essential to align with international environmental goals such as the HELCOM Baltic Sea Action Plan (BSAP) (HELCOM 2021b), the EU Marine Strategy Framework Directive (MSFD), and the Sustainable Development Goals (SDGs).

1.2 - Greywater in Baltic Sea shipping

Greywater, defined as sanitary wastewater generated from sinks, showers, laundries, and galleys, is increasingly recognized as a significant but under-regulated waste stream from ships. In addition to these conventional sources of greywater, a U.S. Environmental Protection Agency (US-EPA) study identified other sources to include wastewater from bars, pantry, sink, salon and spa drains, laundry floor drains, dry cleaning condensate, refrigeration and air conditioning condensates, garbage room drains, and medical facility sinks and drains (US-EPA, 2011). Due to the variability in sources, greywater composition differs significantly between ships, making its definition and characterization complex. Recent monitoring highlighted that ships discharged about 5.4 million m^3 of greywater to the Baltic Sea in 2022, with passenger ships responsible for over 84% of volumes (Jalkanen et al., 2023). Unlike blackwater, which is subject to stricter MARPOL Annex IV controls, greywater discharges remain largely unregulated even though it also contains nutrients, metals, organic matter, detergent, oils, pathogens, synthetic fibers, microplastics and pharmaceuticals (Kalnina et al., 2022; Mujingni et al., 2024; Mujingni et al., 2025). Recent studies in the Baltic Sea region have identified a broad spectrum of contaminants in ship-generated greywater, including metals, nutrients, total suspended solids (TSS), organic matter, oxygen-consuming substances (OMOCS), microplastics, pharmaceuticals, and organic compounds (Mujingni et al., 2024). The discharge of untreated greywater poses potentially significant environmental risks (Ytreberg et al., 2020). Notably, metal concentrations in all three primary greywater streams (accommodation, laundry, and galley) exhibit high hazard potential, with Hazard Indices (HI) exceeding safe thresholds by several orders of magnitude. Furthermore, phosphorus, TSS, and OMOCS (COD-Cr and BOD_5) in greywater often surpass the discharge limits set by the International Maritime Organization's (IMO) Marine Environment Protection Committee (MEPC) 227(64) regulation for sewage effluent from Advanced Wastewater Treatment Plants (AWTP) (Mujingni et al., 2024). In terms of toxicity potential among liquid waste streams from ships, greywater ranks third, following sewage (second) and closed-loop scrubber effluent (first) (Ytreberg et al., 2021). Currently Denmark, Finland and Sweden have banned the discharge of open-loop scrubber water from 2025 (Gustavsson & Westerberg, 2025) in their territorial sea, and the ban for all scrubber effluents will take effect from 2029 (Bergman, 2024). Onboard laundry has been identified as a hotspot for microplastics and phosphorus emissions, with concentrations of microfibers in laundry greywater reaching hundreds of thousands of particles per cubic meter on RoPax vessels, transport ships, cruise ships and research ship in the Baltic Sea (Mikkola, 2020; Kalnina et al., 2022; Mujingni et al., 2024; Jang et al., 2024). These emissions contribute to the accumulation of pollutants in a semi-enclosed marine environment already under pressure from eutrophication, hazardous substances, and climate change (HELCOM, 2021a).

The Baltic Sea presents a unique context where the scale of RoPax and cruise traffic intersects with ecological vulnerability. Trelleborg, for instance, is a very busy RoPax hub in northern Europe, with multiple daily connections to Germany, Poland, and Denmark, and therefore generates substantial volumes of greywater that either get discharged at sea or handled through port reception facilities (PRFs) (Folbert et al., 2022). Despite the potential environmental impacts of greywater discharge, and while some ports in the region, including Trelleborg, have invested in pre-treatment plants (PWTP) and municipal wastewater integration, a harmonized management approach backed by regulations is still lacking. As a result, greywater management in the Baltic is highly fragmented.

Greywater generated on passenger ships can either be discharged directly into the marine environment or delivered to port reception facilities (PRF) for subsequent treatment at MWTP before final discharge. On cruise ships, greywater is often used to dilute blackwater before treatment in onboard AWTPs, as dilution enhances the efficiency of the treatment process. Additionally, some vessels mix greywater with comminuted food waste before discharging it into the sea at regulated distances from shore (Kalnina et al., 2022). Consequently, current greywater handling practices rely primarily on voluntary industry and national initiatives, company policies, and environmental stewardship efforts rather than standardized regulatory frameworks. This diversity underscores both regulatory gap and the opportunity for innovation and highlights the need for coordinated strategies such as developing cost-effective and environmentally beneficial greywater management scenarios that can reduce pollutant discharges and contribute to HELCOM's Regional Action Plan on Marine Litter and nutrient reduction targets, to ensure the overall sustainable management of greywater in the region. Against this backdrop, the present study models compare ten alternative scenarios for ship-generated greywater management Trelleborg, evaluating their costs and environmental benefits from a "cradle-to-grave" perspective.

1.3 - Aim and objectives of the project

The overall aim of this project is to assess the costs and environmental benefits of alternative ship-generated greywater management scenarios in the Baltic Sea using the Trelleborg case study. The objectives are fivefold:

1. To study the characteristics of greywater generated by RoPax ships and model ten Greywater (GW) management scenarios using Trelleborg as a case study.
2. To compute the financial costs (societal and stakeholder costs) of managing GW in each scenario.
3. To derive shadow prices of targeted contaminants from Municipal Wastewater Treatment Plants (MWTP) and calculate the environmental benefits of each scenario
4. Apply a cost-benefit analysis (CBA) framework to compute the net societal benefit (NSB) of each scenario and rank them.
5. To provide recommendations for stakeholders on the most sustainable pathways for managing ship-generated greywater in the Baltic Sea region and, some implications to policy.

With the aims established, the structure of the paper is outlined. It begins with a review of the background literature and policy context for greywater management. Next, data gathering and analytical methods are described. Finally, externalities are monetized via shadow pricing and a cost–benefit analysis of ten scenarios: onboard treatment versus port reception options including port treatment for discharge, reuse or pre-treatment and municipality treatment with or without pre-treatment is performed, and results are presented. The study concludes with implications for industry and policy, and actionable recommendations.

1.4 - Scope and Limits

This study focuses on RoPax vessels calling at the Trelleborg Port and the management of their greywater streams. The analysis applies a "cradle-to-grave" system boundary from shipboard generation to final discharge to the sea or reuse, passing through other processes as represented in the different scenarios. While the study provides cost and environmental benefit estimates grounded in empirical data and literature, certain limitations remain. First, shadow prices for pollutants such as microplastics are still developing, and proxy values (if available) must be used with caution. Secondly, results are specific to the Trelleborg system and its infrastructure, although they provide transferable lessons for other systems in other cities, countries and regions. Thirdly, uncertainties regarding future regulation, technology costs, and ship traffic growth are addressed through sensitivity analysis, but residual uncertainties remain.

1.5 - Methodological choice: Cost-Benefit Analysis with shadow price modelling

The methodology applied in this study is Cost-Benefit Analysis (CBA), which compares the monetary value of costs with the monetized environmental benefits of reducing greywater-related pollution. Research on ship-generated GW management is limited within the Baltic Region and recent arguments (for instance, Friends of the Earth (2023)) presented to the IMO by environmental NGOs and member States have expressed the need to regulate GW together with black water (BW). Moreover, the lack of an international legal regime for GW management in the shipping industry has raised questions on the efficiency of available treatment technologies in removing the identified pollutants in GW and the need for sustainable management strategies. While these concerns prompt avenues for debates, the cost component and the environmental benefit of available GW management options are still unclear, therefore the net profit of using one management strategy over the other from "cradle-to-grave" is unknown. As environmental quality is a priority for authorities, there's the need to find the most appropriate strategy to protect the marine environment (water resources) from cost and environmental benefit perspectives. Within the context of water resource management, greywater discharged from either onboard holding tanks and treatment plants or from MWTPs has associated environmental benefits known in economic terms as positive externalities. While the economic valuation of these externalities is important in justifying the economic feasibility of wastewater management schemes, positive externalities have no market value, rendering their quantification cumbersome (Molino-Senante et al., 2011).

This project strengthens debates on sustainable greywater management by computing the Net Profit (NP) of GW management from "cradle-to-grave" based on Cost-Benefit Analysis (CBA) performed on five modelled ship-generated GW management scenarios. The cost for handling GW in PRF, PWTP and MWTP, as well as treatment results concerning the main pollutants driving the hazard potential of GW identified as zinc (Zn), copper (Cu), manganese (Mn), nitrogen (N), phosphorus (P), total suspended solids (TSS), COD-Cr (chemical oxygen demand), 5-day biochemical oxygen demand (BOD_5), polyethylene terephthalate (PET), polypropylene (PP) (Mujingni et al., 2024) are calculated. The reduction of pollutants is given a shadow price to reflect the benefit for avoidance of negative environmental impact. The shadow prices, the average volume of effluent, and the number of pollutants eliminated using available technologies, are used to calculate the environmental benefit. CBA is performed to obtain Net Profit (NP) which is the difference between the environmental benefits and the costs of managing GW from generation to final disposal. The project seeks to evaluate and compare the NP in the ten modeled scenarios and rank them in order of decreasing NP to find the most sustainable strategy in handling GW from ships. This study would widen environmental researchers' competences in this field, as, in the absence of sufficient evidence-based research findings, decisions made may be misleading. Moreover, the results would enable policymakers to make informed decisions on which level in the GW management chain efforts should be targeted to protect the Baltic Sea and promote sustainable shipping.

1.6 - Background Literature and Policy Context

1.6.1 - Ship-generated greywater characteristics and pollutant loads

Greywater constitutes the largest volume of sanitary wastewater among other numerous wastewaters generated on board ships. An estimated volume of 5.4 million m^3 of GW was generated by ships in the Baltic Sea (Baltic ships) in 2022, 84% of which was collectively generated by RoPax and cruise vessels (Jalkanen et al., 2023). GW volume is as much as four times the volume of blackwater generated on RoPax ships (Mujingni et al., 2024) and per capita generation rates on board passenger ships range from 157 – 235 L/person/day (Mikkola, 2020). GW is wastewater generated mainly from the showers and sinks in the cabins, dishwashers and sinks in the kitchen and restaurants, and laundry machines and sinks in the laundry rooms. As such, it is known to have three main sub flows, notably accommodation, laundry and galley GW streams. The percentage contribution from the accommodation, laundry and galley sub flows have been estimated as 64%, 19% and 17%, respectively, on cruise ships (Mikkola, 2020) and 61%, 8% and 9% on ferries (Juneau, 2021). On average, the accommodation GW sub flow is the highest volume, followed by laundry, and galley GW is the least.

Ship-generated greywater, while less regulated than blackwater, can contain substantial pollutant concentrations. Ship-generated greywater contains nutrients (nitrogen, phosphorus), organic matter, detergents, metals, pathogens, fats, oils, and emerging contaminants such as pharmaceuticals, per- and polyfluoroalkyl substances (PFAS), and microplastics (MPs) (Baresel & Olshammar, 2019; Kalmina et al., 2022; Mujingni et al., 2024; Ytreberg et al., 2022). When discharged into the sea, it is potentially toxic to marine life. For instance, greywater has the 3rd highest toxicity potential after sewage (2nd) and open loop scrubbers (1st) (Ytreberg et al., 2021). Moreover, five RoPax ships that operated in the Baltic Sea could potentially load 16.6 tonnes TSS, 2.15 tonnes N, 1.58 tonnes P, 112 tonnes COD-Cr, 62.4 tonnes BOD_7, 25.9 billion MPs and 5.38 tonnes fat, into the Baltic Sea (Mujingni et al., 2024).

A study on contaminants from Baltic ships in 2022 showed that GW potentially contributed about 179 – 188 tons of phosphorus, of which 68 tons was from GW and the rest principally originating from sewage (0 – 9 tons) and food waste (110 tons) into the Balti Sea. Moreover, of about 402 – 447 tons of nitrogen discharged, 232 tons mainly originated from greywater, and the rest from sewage (68-113 tons) and food waste (101 tons) (Jalkanen et al., 2023). Specifically, the RoPax ships in the same study, being the highest contributor of greywater discharge that year in terms of volume (64.5%), discharged 143 tonnes N and 61.8 tonnes P into the sea via GW. Among metals, Zn and Cu are the highest contributors to the environmental risk of GW, with a percentage contribution of 94% (Zn- 67% and Cu – 27%) to the total cumulative risk (Ytreberg et al., 2020). These results match Mujingni et al. (2024) in which Zn, Cu, and Mn were identified as the highest contributors to the hazard potential of GW, contributing 95% (Zn – 63%, Cu- 20%, Mn – 12%) to the Hazard Index (HI). Furthermore, the geometric means of COD-Cr (≈ 640 mg/L) and BOD_5 (≈ 290 mg/L) were several times higher than the MEPC 227(64) sewage effluent requirement for COD-Cr: 125 mg/L and BOD_5: 25 mg/L (Mujingni et al., 2024). A prior project also revealed the presence of several microplastic polymers in GW and identified Polyethylene terephthalate (PET) and Polypropylene (PP) as the most prominent MPs, with a contribution of 74% (PET – 58% and PP – 16%) to the total MP occurrence (Mujingni et al., manuscript ongoing). Besides these pollutants, pharmaceuticals were also identified in GW from the ships.

1.6.2 - Environmental and socio-economic impacts of untreated discharges

The ecological and socio-economic implications of untreated or insufficiently treated greywater discharges are enormous. The most prominent is nutrient enrichment which contributes to eutrophication, algal blooms, and oxygen depletion, affecting benthic organisms including fisheries and biodiversity (HELCOM, 2021b). Organic matter and pathogens deteriorate bathing quality and pose risk to public health, especially in densely trafficked coastal areas (Folbert et al., 2022). Microplastics and PFAS represent persistent pollutants with poorly understood but potentially significant risks to marine food webs and human exposure through seafood (Campanale et al., 2020). Socio-economic consequences extend to tourism, aquaculture, and port reputations, as well as the rising treatment burden on municipal wastewater systems that receive shipborne discharges.

1.6.3 - Current practices, initiatives and regulatory frameworks

Current practices for ship-generated greywater management in the Baltic are shaped by a patchwork of regional instruments, evolving national rules, and industry standards, with a clear gap at the global level. GW management practices in the maritime sector are highly variable. Options include direct discharge at sea outside 3 nautical miles, onboard treatment in AWTPs with sewage, mixing with food waste and discharge beyond 12 nm, land-based delivery via PRFs and pre-treatment in PWTPs and treatment in MWTPs (Kalnina et al., 2021). While some shipping companies have voluntarily installed AWTPs to meet stringent environmental performance standards (e.g. for operation in Alaska), most vessels operating in the Baltic still rely on direct discharge or indirect discharge via PRFs where infrastructure exists (HELCOM, 2023). For Baltic passenger ships that mix GW with sewage and treat in on board AWTPs, the requirements of MEPC 227 (64) of 2012 for effluent standards apply. This includes the attainment of minimum geomean effluent concentrations of 125 mg/L COD, 25 mg/L BOD_5, 35 mg/L

TSS, 20 mg/L nitrogen and 1 mg/L phosphorus, before discharging into the Baltic Sea (IMO, 2012). Moreover, ships that mix GW with ground food waste discharge the streams together untreated, in accordance with MARPOL Annex V Special Area discharge provisions. These varying ship-generated GW management strategies perpetuate the entry of contaminants into the Baltic Sea.

Trelleborg and Ystad represent rare cases where ship-generated GW is systematically received and pre-treated at a PWTP before channeling into the municipal system. However, many Baltic ports lack such infrastructure, leading to inconsistency in greywater handling across the region. Regionally, the Baltic Sea is a MARPOL Annex IV Special Area for passenger-ship sewage (blackwater) (IMO, 2017) which since June 2019 for newbuilds and June 2021 for existing ships, has required either discharge to PRFs or onboard treatment to the stricter MEPC.227(64) standard (IMO, 2012). However, these provisions explicitly cover sewage and do not regulate greywater, implying that greywater can still be legally discharged at sea under IMO rules (Jalkanen et al., 2023). This regulatory gap in MARPOL Annex IV has been repeatedly highlighted as a weakness in marine environmental protection (IMO, 2020). In the Baltic Sea, which is also a Particularly Sensitive Sea Area (PSSA), HELCOM has set ambitious targets under the Baltic Sea Action Plan (BSAP) to reduce nutrient inputs, hazardous substances, and marine litter. However, specific greywater discharge restrictions are not yet uniformly implemented. The EU Marine Strategy Framework Directive (MSFD) (2008/56/EC) and the Water Framework Directive (2000/60/EC) establish overarching requirements for achieving Good Environmental Status (GES), indirectly placing pressure on Member States to address shipborne greywater.

Furthermore, MARPOL Annex IV and EU Directive 2019/883 oblige Member States to provide "adequate" PRFs and cost-recovery schemes ("no-special-fee") for ship-generated waste including sewage (EU, 2019), and HELCOM has issued technical guidance to help Baltic ports implement wastewater handling under the Special Area regime (HELCOM, 2019). Yet neither instrument directly mandates greywater delivery unless it is co-mingled with sewage. Ship owners have attested that the current most efficient GW management strategy is to deliver GW to PRFs to be treated further on land. However, the reception of GW by the ports is a concern for municipalities in Sweden because, while MWTPs can efficiently remove nutrients, organic matter and microplastics, they are not designed to efficiently remove some contaminants such as metals and micropollutants such as pharmaceuticals and organic compounds (Svensk Vatten, 2019). This validates the potential of MWTPs to be vectors of the said contaminants from ship-generated GW to the marine environment. In Sweden, according to the local municipal regulations (ABVA), the manager of the MWTP is not obliged to receive wastewater whose content differs significantly from domestic wastewater (Press et al., 2020). Consequently, ports receiving GW from ships are expected to ensure that the GW channeled to MWTPs are of similar or higher quality as domestic wastewater from households. This expectation has placed a strain on the port-municipality interface in the handling of GW from ships, leading to the quest for sustainable solutions for ship-generated GW management. Some ports like the Trelleborg and Ystad ports in Southern Sweden, have installed Port-based Wastewater Treatment Plants (PWTPs) to pre-treat GW from ships before channeling it into the municipality's sewerage system. However, this treatment plants target mainly metals which MWTPs are not designed to remove (personal Communication with Gryaab) and organic matter to a certain extent. A recent study showed that, with regards to metals, there is no significant difference in Hazard Indexes (HI) between greywater, black water and mixed grey- and black water from ships, and domestic wastewater from land (Mujingni et al., 2024), therefore, all four sanitary wastewater types have the same potential to pollute the marine environment with metals. This shows that MWTPs could conveniently treat ship-originated sanitary wastewater in the same way as domestic wastewater.

The adoption of the revised Urban Wastewater Treatment Directive (EU) 2024/3019 adds new relevance to municipal wastewater treatment. This updated Directive explicitly requires monitoring and treatment of micropollutants, per- and polyfluoroalkyl substances (PFAS), and microplastics in wastewater treatment plants, alongside energy neutrality targets and extended producer responsibility (EC, 2024). Although the Directive does not directly cover ship greywater, it sets a benchmark for advanced treatment which may influence expectations for both onboard and port-based systems, since all ship-greywater delivered ashore via PRFs end in MWTPs. Another concern for MWTPs is sludge handling and disposal. MWTPs sludge is primarily used on farmland but must meet strict limitations concerning heavy metals (cadmium, copper, nickel, lead, zinc, mercury and chromium) and other pollutants such as dry matter, organic matter, pH, nitrogen, phosphorus, according to the EU sludge Directive (EC 86/278/EEC). Most Swedish ports outsource sludge handling to specialized companies.

As some stakeholders (especially ship owners and systems manufacturers) applaud the idea of PWTP, there are uncertainties regarding its necessity from a cost-benefit perspective. The inclusion of PWTP in the wastewater management chain means additional cost and it is not yet understood if PWTPs enhance the environmental benefit of the entire wastewater management chain (from "cradle" to "grave"). This strategy can only be advocated if stakeholders understand the net benefit of its inclusion, compared to other conventional wastewater management scenarios.

Nationally, Finland has moved first to close the regulatory gap by banning the discharge of ship wastewater in its territorial waters, phasing in prohibitions that already cover sewage and open-loop scrubber effluent (effective July 1, 2025) and will extend to greywater from January 1, 2030. This, according to the Baltic Sea Action Group (BSAG) signals a potential model for wider Baltic adoption, as ship wastewater is also on the political agenda in Sweden and Denmark where the ban of scrubber discharge water within their territorial waters is already effective from summer 2025. Environmental advocates referred to Finland's wastewater discharge ban as currently the most comprehensive in the Baltic Sea region and urge an extension of the ban to cover the entire Baltic Sea to maximize the impact of such measures (BSAG, 2024). Industry practice is ahead of regulation on many passenger vessels focusing on action by Cruise Lines International Association (CLIA) members who increasingly use advanced (tertiary) wastewater treatment systems that combine and treat black and greywater to standards exceeding baseline rules, and many ships discharge sewage to PRFs in Baltic ports equipped under the EU PRF regime. These measures reduce nutrient, and contaminant loads from direct discharge, but remain voluntary for greywater outside national bans, leaving significant residual discharges.

PRFs are essential infrastructure elements for sustainable maritime operations, as they provide designated points where ships can deposit waste, thereby mitigating the risk of illegal discharges at sea and supporting compliance with international maritime pollution standards (IMO, n.d.). EU-level rules such as the Revised PRF Directive require ports to develop waste reception and handling plans, implement cost-recovery mechanisms, and facilitate advanced notification to improve PRF efficiency and reduce administrative burdens (EU Parliament & Council, 2018). In the Baltic Sea, HELCOM has recognized the elevated sensitivity of the regional marine environment and issued technical guidance for better handling of wastewater in ports, highlighting the need for tailored solutions due to local infrastructure variation (HELCOM, 2019). By linking shipboard waste management with land-based municipal treatment systems, advanced PRFs offer a pathway for integrating circular economy principles, such as resource recovery and effluent reuse, into shipping practices (EMSA, n.d.). Summarily, PRFs play a dual role as both compliance tools for maritime regulation and as enablers of innovation, fostering sustainable shipping through infrastructure and policy integration.

1.6.4 Trelleborg's wastewater management system setup from "cradle to grave".

Trelleborg is a busy RoPax port in Sweden and one of the largest ferry hubs in the Baltic Sea region, handling about 30 daily ferry calls connecting Sweden with Germany, Poland, and Denmark (Port of Trelleborg, 2023). This high frequency of passenger ship traffic translates into substantial greywater generation, making Trelleborg a representative case study for understanding the scale of the problem and testing solutions. Furthermore, Trelleborg is a pioneer in port sustainability initiatives, having invested in a dedicated Port Wastewater Treatment Plant (PWTP) that pre-treats ship-generated wastewater before it enters the municipal sewerage system. The integration with the municipal wastewater treatment plant (MWTP) enables a combined ship-port-municipal approach that is rare in the Baltic and internationally. Thus, Trelleborg provides a unique opportunity to evaluate multiple scenarios of greywater management, from direct discharge at sea to advanced port-based and circular solutions, within one system boundary. Insights from this case study are expected to be transferable to other busy RoPax and cruise ports in the Baltic and beyond.

The ship-generated greywater management system in Trelleborg was selected for this study due to the availability of all system components across the modelled scenarios. Key features include well-structured RoPax vessel traffic to and from the port, stationery PRFs at the RoPax piers, a port-owned wastewater treatment plant, and a connection to the MWTP of Trelleborg municipality. Furthermore, the port's willingness to contribute to this study as well as prior collaboration in the initial Greywater Project also motivated its selection.

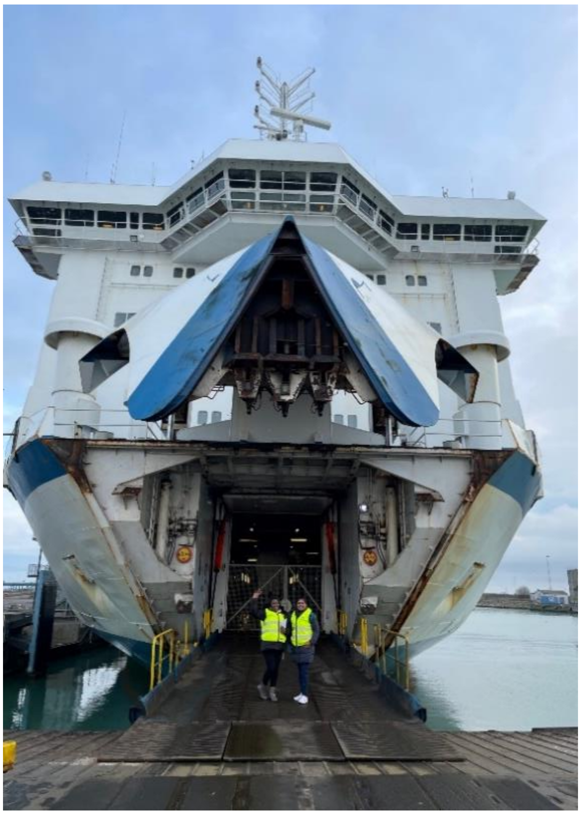
To quantify the volumes of greywater generated by RoPax ships in Trelleborg, three major shipping companies (TT-Line (9), Unity Line (4), and Stena Line (2)) operating a total of 15 RoPax vessels were identified (Trelleborgs Hamn AB, 2023). Annual passenger traffic through the port is ≈ 1,700,000 passengers (PAX) with RoPax vessels navigating an average of 364 days per year, corresponding to roughly 15 daily arrivals and 5,460 annual calls. This equates to an average of 311 passengers per call, or ≈ 4,670 passengers per day. Based on three-year port data (2021 - 2023), an annual average of 50,660 m^3/year of mixed greywater and blackwater, equivalent to 139 m^3/day was received from ships. Assuming an 80:20 ratio of GW to blackwater (Mujingni et al., 2024), the estimated GW volume is ≈111 m^3/day, 23.8L per passenger per day, and ≈ 40,528 m^3/year, equivalent to 7.41 m^3 per call received at the Trelleborg PRFs. This information has been summarized in Table 1.

Table 1: General information on RoPax and wastewater management activities at the Trelleborg port.

General information	Values
Total number of RoPax vessels calling at the port	15
Average annual passenger traffic through Trelleborg port	1,700,000 PAX/year
Annual duration of RoPax operations	364 days/year
Average daily arrivals	15
Average annual calls	5,460 calls/year
Average annual passengers per call	311 PAX/call
Average passengers per day	4670 PAX/day
Average annual volume of wastewater (mixed BW and GW) received (2021 – 2023)	50,660 m^3/year
Average daily wastewater volume received	139 m^3/day
Greywater fraction (based on GW-80:BW-20 ratio)	40,528 m^3/year
Average daily GW volume received	111.3 m^3/day

The Pollutant loads from ships delivering wastewater to Trelleborg port were calculated using average measured pollutant concentrations from 242 datasets obtained from 20 RoPax ships sampled during regular wastewater analyses at the port reception facility between 2021 and 2023, as well as the average annual volume of wastewater received at the port within the same period (≈ 50,660 m^3). Appendix 1, Table 2 presents the concentrations and loads of contaminants in mixed GW and BW received from RoPax ships at the Trelleborg port.

Regarding the pollutants PET and PP, the concentrations obtained from Mujingni et al. (2025) were converted to kg/m^3 based on literature values for densities, sizes and assumption of spherical shapes. Therefore, the count-based concentrations of PET and PP (119,000 MPs/m^3 and 33,000 MPs/m^3) were converted to mass-based obtained as 8.61×10^{-5} kg/m^3 PET and 1.56×10^{-5} kg/m^3 PP. Assumed densities were PET, 1.38 g/cm^3 = 1380 kg/m^3 (Ahtiainen et al., 2014; Dhaka et al., 2022) and PP, 0.90 g/cm^3 = 900 kg/m^3 (Stride et al., 2024), particle diameter was 100µm (0.0001 m). With these concentrations, 40,528 m^3/year of GW received by the Trelleborg port would result in average annual contaminant load as shown in Appendix 1, Table 1. Using data on concentrations of contaminants in mixed grey- and blackwater obtained from the Trelleborg port and considering the average annual volume of wastewater received by the port from 2021 to 2023 (50,660 m^3/year), the resulting loads are as shown in Appendix 1, Table 2.



The lack of international regulation for greywater makes it difficult to assess management scenarios. However, MARPOL Annex IV has specific provisions for sewage against which the discharge of GW could be assessed (Mujingni et al., 2024). In the Baltic Region, IMO MEPC 227 (64) of 2012 obliges passenger ships to discharge only effluent treated in Advanced Wastewater Treatment Plants to certain standards into the Baltic Sea. This standard only applies to greywater when it is mixed with black water, defined as sewage according to MARPOL Annex IV. Industry initiatives and environmental stewardship are the main motivating factors for controlled discharge of pure greywater. All RoPax ships calling at the port of Trelleborg discharge greywater to PRFs.

Figure 1*: A RoPax ship berth at the Trelleborg Port. The installation and use of PRFs is the obligation of the State in accordance with the EU Directive 2019/883, implemented in Sweden through SFS 1980:424 and SFS 1980:789 as well as the Swedish transport authority's regulations TSFS 2023:15.*

According to the PRF regulations, ports should provide reception facilities that are adequate to receive the types and quantities of waste delivered by ships, including sewage. The port also has the obligation to develop and communicate a Port Waste Reception and Handling Plan (PWRHP). At the port of Trelleborg, there are stationery PRFs at all quays receiving sanitary wastewater from RoPax ships.

Moreover, the port of Trelleborg has progressively modernized its wastewater management system to align with sustainability and regulatory objectives. Since June 2021, a complete ban on direct discharge of untreated sewage from passenger ships into the Baltic Sea Special Area under MARPOL Annex IV has been in effect (IMO MEPC.275(69), 2016), prompting all arriving vessels to offload their greywater and sewage at port facilities (Port of Trelleborg, 2025a; 2025b). Historically, such wastewater was directly transferred to Trelleborg MWTP. More recently, under the EU-supported "Green FIT 2025" initiative, Trelleborg has constructed an on-site port wastewater treatment plant (PWTP) designed for an initial daily capacity of approximately 400 m^3, equating to 146,000 m^3 annually, or about 58 Olympic swimming pools (Port of Trelleborg, 2025a; Marinfloc, 2025). Operational since late 2023, the Trelleborg PWTP conducts preliminary treatment, including removal of heavy metals such as copper and zinc.

***Figure 2**: Port reception facility for blackwater and greywater at the Trelleborg Port.*

According to Marinfloc, manufacturer of the plant, the PWTP was installed at the port to reduce the levels of heavy metals, grease, phosphorus and to some extent, BOD, COD in wastewater effluent from ships before pumping to the municipality, in conformity to the requirements of ABVA and P95 except for ammonium and nitrogen (Trelleborgs kommun, 2023). The main treatment process at the PWTP separates suspended materials, particulate-bound metals and phosphorus, through chemical precipitation and flocculation, as well as forced flotation using a flocculant known as Dialuminium Chloride Pentahydroxide and a coagulant (flocbooster). The treated effluent is then conveyed to the municipal sewage system for final purification (Port of Trelleborg, 2025c) while the dewatered sludge is delivered to a company to be used for soil enhancement in agriculture and landfilling.

Figure 3: Aerial view of the Trelleborg Port Wastewater Treatment Plant

Trelleborg PWTP received and treated an average of 50,660 m^3/year from 2021 to 2023. Of this volume about 40,528 m^3 (80%) is assumed to be greywater. Sampling results from the plant have shown some varied removal efficiencies for selected contaminants. The average results of influent and effluent samples collected, analyzed and reported by SGS Analytics Sweden AB on 23 April 2024, and 3 May 2024 (personal Communication with Trelleborg port) are shown in Appendix 1, Table 4.

The municipal wastewater treatment process at the Trelleborg MWTP combines mechanical, chemical, and biological methods to ensure effective and environmentally sound treatment. Incoming wastewater first passes through screens and sand traps for mechanical cleaning. Biological treatment uses the activated sludge method, where bacteria decompose organic matter (BOD) and convert nitrogen compounds into nitrogen gas via nitrification and denitrification, supported by ethanol as an external carbon source. Chemical treatment targets phosphorus removal through ferric chloride precipitation. A final polishing stage directs effluent through a series of ponds before discharge, with bypass options when necessary. Operational management focuses on optimizing efficiency, applying best available techniques appropriate to the plant's size and balancing environmental performance with cost-effectiveness.

Figure 4: Highlight of Wastewater treatment processes at the Trelleborg Municipal Wastewater Treatment Plant.

In addition to the requirements of the UWWD, the MWTPs treat wastewater according to industry recommendations in P95 (Svenskt Vatten, 2019) and decided municipal supplementary regulations to "Allmänna Bestämmelser för Vatten- och Avloppsanläggningar" (ABVA) (2009). Appendix 1, Table 5 shows that effluent from the PWTP meets the requirements of ABVA and P95, hence can be conveniently discharged into the Trelleborg MWTP sewerage system.

According to the Trelleborg MWTP Sustainability report of 2023, from 2021 – 2023 the plant processed on average 3,723,195 m^3 of wastewater. Of this volume, ≈ 50,660 m^3 (≈ 1.4%) was from Trelleborg port. An average energy of 63.3 kWh/person equivalent or 0.43 kWh/m^3 was consumed at the plant during this period. On average 2,231 m^3 sludge was produced with averagely 27.5% TS. BOD_7, Phosphorus and Nitrogen removal from 2021 – 2023 achieved average removal efficiencies of 99%, 95% and 80%, respectively, leading to attainment of levels lower than the permit levels. In 2023, a total of 4,177,879 m^3 of wastewater was processed by the plant, constituting a daily average of 11,446 m^3. 50% of this volume was make-up water (uncharged water from sources such as storm water in connection with rain, snowmelt and high groundwater levels). The mean concentrations of selected contaminants in the influent and effluent from the plant as well as their removal efficiencies in 2023 are as shown in Appendix 1, Table 6.

This integrated system, from vessel offloading at PRFs, through pre-treatment at PWTP, to full treatment at MWTP before discharging into the sea, establishes a comprehensive cradle-to-grave chain for ship-generated wastewater, reflecting innovative localized implementation of international maritime and environmental standards.

1.6.5 Rational for economic valuation of environmental improvements

The growing recognition of the ecological and socio-economic costs of marine pollution has spurred interest in economic valuation approaches. Shadow prices for pollutants such as nitrogen and phosphorus have been developed under EU water policy, HELCOM assessments, and published literature, providing monetary values for avoided eutrophication and ecosystem damage (HELCOM, 2018b; Gren et al., 2017). While valuation of microplastics and other emerging pollutants is more uncertain, proxy values have begun to emerge in the literature (Everaert et al., 2020). Economic instruments such as port fee differentiation, polluter-pays principles, and extended producer responsibility (EPR) (as introduced in the newly revised UWWTD) represent potential policy levers for shifting costs from society to polluters. Against this backdrop, a Cost-Benefit Analysis (CBA) applying shadow prices approach provides a systematic way to compare alternative GW management scenarios and align them with regional policy targets.

2 - Methodology

2.1 - Overview of CBA framework for ship greywater management

Cost-Benefit Analysis (CBA) has long been established as a fundamental economic tool for evaluating environmental management strategies, enabling policymakers and stakeholders to compare the monetary value of costs incurred with the benefits of avoided damage or improved environmental quality (Boardman et al., 2018). Within the maritime sector, CBA has been applied to interventions ranging from ballast water treatment to ship emission abatement, but its application to greywater management remains limited, with only a few studies addressing the socio-economic implications of wastewater discharges from ships (Olshamar & Baresel, 2019; Gren et al., 2017). The central strength of CBA lies in its ability to provide a net social benefit metric, usually expressed as net present value (NPV) or benefit-cost ratio (BCR), which allows for ranking of policy and technological alternatives on grounds of efficiency.

A key methodological challenge in applying CBA to ship greywater is the valuation of environmental externalities, since many of the pollutants, notably nutrients, organic matter, microplastics, metals, do not have direct market prices. This is typically addressed by employing shadow prices, which approximate the marginal social cost of pollutants based on damage costs, abatement costs, or willingness-to-pay studies (Hanley & Barbier, 2009). Recent empirical work has advanced the estimation of shadow prices for wastewater pollutants by applying distance-function and data envelopment analysis (DEA). For example, Antalova et al. (2000), estimated shadow prices for Slovak WWTPs as ≈ -€31.942/kg N, -€82.433/kg P, -€10.706/kg TSS, and -€2.277/kg COD, corresponding to an average environmental benefit of ≈ €4.9/m^3 of treated water. Similarly, Molinos-Senante, Hernandez-Sancho, and Sala-Garrido (2010, 2011) applied shadow-price based CBA to Spanish WWTPs, finding that environmental benefits from pollutant removal could make previously marginal reuse projects economically feasible. Such estimates illustrate both the feasibility of the method and the variability of shadow pricing approaches in wastewater contexts. In the Baltic Sea, shadow prices for nitrogen and phosphorus, COD, BOD and TSS have been estimated in several studies (Gren et al., 2017; HELCOM, 2018b), providing a robust basis for monetizing eutrophication impacts. In contrast, valuation of microplastics and emerging contaminants remains uncertain, though recent studies suggest approaches based on avoided clean-up costs, ecological damage functions, or substitution by analog pollutants (Everaert et al., 2020). However, unlike N and P, which have well-established shadow prices, microplastics lack empirical estimates from distance-function studies and must rely on proxy valuations.

Another methodological innovation is the integration of CBA with a distance function approach, which evaluates how much each management scenario reduces the “distance” to environmental targets such as the HELCOM Baltic Sea Action Plan or EU Marine Strategy Framework Directive objectives. This enhances conventional CBA by embedding results within a policy-relevant ecological context

(Kuosmanen & Kortelainen, 2005). Such approaches allow decision-makers not only to identify the scenario with the highest net benefits, but also to assess whether proposed measures are sufficient to close existing policy-environmental gaps. The distance-function approach has a strong theoretical foundation in Färe et al. (1993), who showed how output distance functions under weak disposability of undesirable outputs can be used to derive shadow prices for pollutants. By fitting a flexible functional form (e.g. translog), the distance function derives shadow prices by modeling the joint production of desirable outputs (treated effluent) and undesirable outputs (pollutants). Duality relationship allows recovery of shadow prices for pollutants once one observable market price, typically the price of treated water or wastewater tariffs, is used for normalization. This ensures that valuations are grounded in observed technology and production frontiers, rather than ad-hoc damage estimates. The literature highlights that shadow prices are plant- and technology-specific, sensitive to functional form, and must be interpreted as marginal abatement costs rather than universal external damage values (Färe et al., 1993).

These methods have been applied to wastewater management studies. For instance, Molinos-Senante and colleagues translated shadow prices into per m^3 environmental benefit estimates, which can be directly compared to per m^3 costs of treatment. Engineering-economic studies of wastewater treatment plants demonstrate wide variability in unit treatment costs depending on scale and technology, typically ranging from USD 1.10 – 1.46/m^3 in small to medium systems. Combining such cost evidence with pollutant-specific shadow prices provides a transparent way to calculate net benefits of ship greywater scenarios (e.g onboard treatment, port reception or reuse). These cost benchmarks are useful for parameterizing ship- and port-based scenarios. Costs include capital expenditure (CAPEX) for onboard systems or port infrastructure, operational expenditure (OPEX) for energy, maintenance, labour, chemicals and downstream costs of sludge handling and disposal. Environmental benefits are calculated as avoided pollution loads multiplied by corresponding shadow prices. When combined with sensitivity analysis, this framework provides a transparent and replicable method to balance private costs borne by ship operators and ports with social benefits accruing from cleaner marine environment.

We adopt both a societal and a stakeholder perspective in the CBA. The societal perspective includes all real resource costs (annualized CAPEX and OPEX), while excluding internal transfers such as port fees, which are redistributive rather than resource consuming. This allows us to assess the overall economic efficiency of each greywater management scenario.

In parallel, we calculate the financial positions of shipowners, ports, and municipalities under each scenario by including fee payments and revenues. This distributional analysis highlights who bears the financial burden or enjoys surplus revenues. While these transfers cancel out in the societal totals, they are important for understanding feasibility, equity and stakeholder incentives. Presenting both perspectives together thus enables us to compare the social desirability of scenarios with their financial attractiveness for individual actors.

In summary, the literature demonstrates that while CBA is well-established in environmental economics, its extension to ship greywater remains underdeveloped, offering scope for novel contributions. Integrating shadow pricing in this study addresses this gap, offering a robust way to link pollutant reductions to monetary benefits while embedding the analysis in the context of environmental targets such as HELCOM nutrient reductions. By applying this approach to Trelleborg's port wastewater setup, this study contributes both methodological innovation and policy-relevant evidence on sustainable greywater management in the Baltic Sea.

2.2 - System Boundary Definition: Cradle-to-Grave

In this study, the system boundary is defined on a "cradle-to-grave" basis with respect to both the greywater stream and the infrastructure required for its management. This dual framing ensures that all relevant processes, costs, and environmental effects are captured, from the production of treatment systems to their decommissioning. Adopting this comprehensive boundary is crucial in a cost-benefit analysis (CBA) to avoid underestimation of costs or overestimation of environmental benefits.

The cradle phase begins with the use of water and the generation of GW on board RoPax vessels calling at the Port of Trelleborg. Greywater originates mainly from accommodation spaces, laundries, showers, galleys, and other onboard activities, and its pollutant load is characterized by nitrogen (N), phosphorus (P), chemical oxygen demand (COD), suspended solids (SS), microplastics (MPs), metals (Zn, Cu, Mn), and other contaminants. At the same time, the "cradle" includes the capital expenditure (CAPEX) for the production, storage, transport, and installation of treatment equipment and infrastructure required in each scenario. For ship-based options this involves onboard greywater storage tanks, onboard Advanced Wastewater Treatment Plants (AWTPs), pumps, and piping, while for port-based scenarios it covers port reception facilities (PRFs), transfer pipelines, and the Port Wastewater Treatment Plant (PWTP). This phase thus links the physical creation of wastewater with the embedded costs of establishing the systems needed to manage it.

The transfer and operation phase comprises the collection, pumping, and conveyance of GW from ships to treatment facilities. Operation also encompasses chemicals, membranes, labour, crew time for monitoring, and routine maintenance. Pollutant removal efficiencies achieved during this phase directly determine the size of environmental benefits, which are monetized in the CBA using shadow prices.

The treatment phase represents the core of environmental performance. In onboard AWTPs, GW undergoes biological, chemical, and/or membrane-based purification before discharge. In land-based pathways, the PWTP provides pre-treatment, after which GW is either conveyed to the MWTP for full treatment, or reused at the port for non-potable applications such as flushing toilets, washing or dust suppression. Pollutant removal in each scenario is quantified in terms of kg/year of N, P, COD, SS, Zn, Cu, Mn, MPs (PET, PP), and monetized as environmental benefits through shadow prices. The treatment phase also determines variability in OPEX, since advanced processes often require more energy and consumables. The waste handling phase also captures the management of residuals generated by GW treatment, including sludge, screenings, and concentrate streams. These by-products must be dewatered, transported, and disposed of, typically at municipal facilities, or in some cases valorized. Sludge can be digested for biogas recovery or applied in agriculture for nutrient recycling, thereby generating secondary environmental benefits or avoided costs. This stage is also associated with disposal fees, transport costs, and potentially credits for recovered sources, all of which must be factored into the CBA. In this study, the sludge both at Trelleborg port and municipality is delivered to farmers to be used for soil enrichment. However, the sludge component was not included in the current CBA.

The disposal phase (the "grave") closes the system boundary by accounting for the final fate of both the treated effluent and the treatment infrastructure itself. For the effluent, this means either discharge into the Baltic Sea or reuse within the port. Discharge scenarios are assessed in terms of the avoided pollutant load relative to the baseline (direct untreated discharge at sea), monetized using pollutant-specific shadow prices. The reuse scenario adds additional benefits by substituting freshwater abstraction, contributing to resource efficiency goals. For infrastructure, disposal costs are often smaller than CAPEX or OPEX, they are included for completeness to ensure alignment with cradle-to-grave principle.

By structuring the analysis around this “cradle-to-grave” boundary, the study ensures that all scenarios are evaluated on a consistent basis, combining financial costs (CAPEX, OPEX) with monetized environmental benefits (pollutant reductions, resource substitution). This comprehensive approach makes visible the tradeoffs between low-cost but high-impact options and high-investment but high-benefit pathways. It also provides a transparent foundation for policy-relevant insights, showing how different management strategies contribute to regional objectives such as the HELCOM Baltic Sea Action Plan and the EU Marine Strategy Framework Directive.

2.3 - Defining Alternative Scenarios

Ten alternative scenarios have been modelled, categorized into two groups, each incorporating progressively enhanced treatment, circularity, and reuse elements: non-reuse (SC1 – SC5A) and reuse ($SC5B_1$ – $SC5B_5$) categories. The non-reuse scenarios SC1 and SC2 involve direct discharge into the sea at a minimum of 3nm from shore, with negligible transfer cost but potential environmental implications. SC1 discharges directly into the sea after generation while SC2 first treats GW in onboard AWTP before discharging into the sea. SC3 – SC5A discharge indirectly passing via facilities at the port and/or the municipality. All three scenarios discharge GW to PRFs but, while SC3 discharges directly from PRF to MWTP for treatment before discharging into the sea, S4 first pre-treats in PWTP before discharging to MWTP for treatment and sea discharge. This configuration improves treatment efficiency and reduces the pollutant load entering the municipal system. Moreover, S5A only treats at PWTP and discharges into the sea without conveying to MWTP.

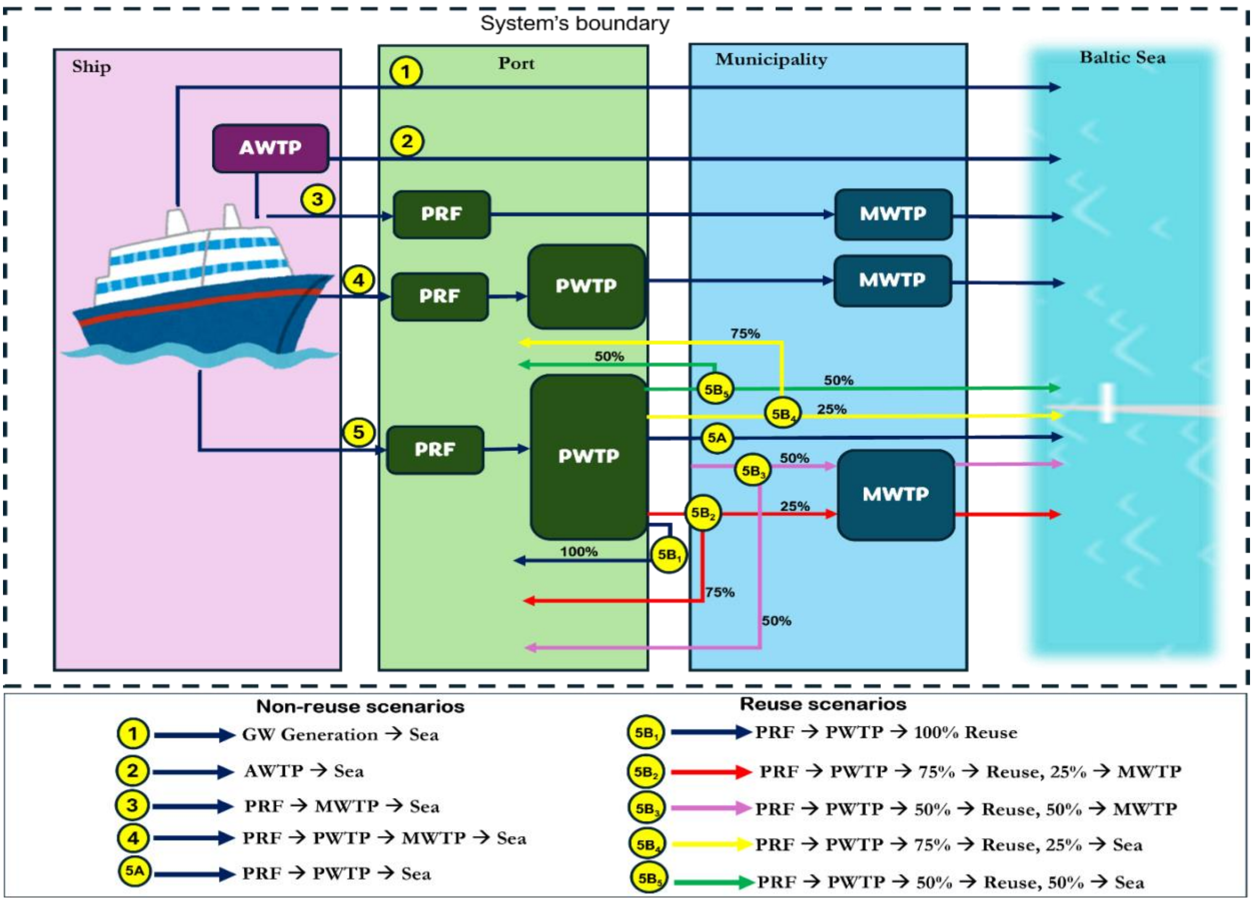


Figure 5: Schematic diagram showing ten modelled GW management scenarios at Trelleborg.

Reuse scenarios represent the most advanced circular configuration, involving port-based pre-treatment and reuse of treated effluent for non-potable purposes, such as toilet flushing at port facilities, or discharge into the sea when reuse is not operationally feasible. In reuse scenarios $SC5B_1$ – $SC5B_5$ GW is received in PRFs and conveyed to PWTP for treatment. The fate of the treated GW is modelled in five reuse scenarios, including both complete and partial scenarios. In $SC5B_1$, all the treated GW (100%) is reused at the port. In $SC5B_2$ and $SC5B_3$, 75% and 50% of GW treated respectively is reused at the port, while 25% and 50%, respectively, is channeled to MWTP. Moreover, in $SC5B_4$ and $SC5B_5$ partial reuse scenarios 75% and 50% of treated GW, respectively, is reused at the port, while 25% and 50%, respectively, is discharged to sea. These scenarios together form a credible and complete set of management options encompassing the full technological and operational spectrum, from current discharge practices to advanced hypothetical circular solutions. Evaluating these scenarios within the CBA framework allows for a systematic comparison of their relative costs, benefits, and long-term societal impacts, ultimately guiding the identification of the most cost-effective and sustainable pathway for GW management in the Baltic Region. The following schematic diagram presents all the scenarios. Figure 5 presents a schematic diagram of modelled scenarios.

2.4 - Cost Analysis of Greywater Management

The analysis of the cost of GW treatment from "cradle to grave" is done in two parts: the societal perspective and the stakeholder perspective. The basic information about wastewater activities at the port of Trelleborg is presented in Table 1. Information from the year 2021 to 2024 was considered based on data availability.

The societal perspective provides the foundational framework for the CBA, evaluating the efficiency of GW management options from the standpoint of overall societal welfare. It includes real resource costs: annualized capital expenditure (CAPEX) and operating expenditure (OPEX) for onboard systems, PRFs, PWTP, and MWTPs. CAPEX comprises the cost of manufacture and installation of various systems, while the operational expenditure (OPEX) comprises costs incurred during wastewater management. At treatment plants, CAPEX is usually expressed in per m3 of wastewater treated. The OPEX of greywater treatment is classified into five categories, namely: energy, staff, reagents, maintenance and waste management. Energy includes the cost related to the fixed part of the energy consumption, power term and the variable part, energy consumption for the installation. Staff cost includes wages, social security charges, taxes and social insurance for the workers. Reagents represent the cost of chemicals used for wastewater and sludge treatment. Maintenance includes cost of equipment and machinery maintenance and replacement. While waste management is the cost associated with the management of sludge and other wastes resulting from wastewater treatment (Molinos-Senante et al., 2010; Hernandez-Sancho et al., 2010).

Critically, the societal perspective excludes transfers such as port reception fees, subsidies, or VAT, because these are purely redistributive and do not reflect net resource consumption (OECD, 2006; OECD, 2015). These are calculated in the stakeholder CBA. By combining real resource costs and environmental damage, the societal perspective provides annualized net profit estimates per scenario, both as totals and per unit of greywater treated SEK/m3. These results indicate which scenarios are most efficient from a societal standpoint. Box 1 shows the cost chain model for calculating the societal cost for the ten modelled scenarios.

2.4.1 - Cost of Fresh water (C_{pot})

The cost of fresh water (C_{pot}) represents the economic value of each cubic meter of potable water supplied to ports by municipal drinking-water utilities. In this study, C_{pot} is a central parameter because GW reuse directly reduces the volume of potable water that ports need to purchase. Its accurate estimation is therefore essential for both the stakeholder cost analysis and the societal cost-benefit analysis. From the stakeholder perspective, C_{pot} corresponds to the tariff that ports pay for municipal water. This tariff generally reflects production and distribution costs and may also include wastewater-related charges linked to water consumption (OECD, 2009). Ports use freshwater for several operational purposes, including toilet flushing, cleaning, and service provision to ships (European Sea Ports Organisation (ESPO), 2020; IMO, 2018). When GW reuse replaces part of this demand, ports avoid some of these purchases and obtain direct financial savings (Grant et al., 2012; Lyu et al., 2016). For this reason, C_{pot} appears explicitly in stakeholder cost calculations as the avoided payment for water that would otherwise be bought from the municipality. In the societal analysis, C_{pot} has a broader meaning. Market tariffs do not fully capture the resource burden associated with potable-water production. Producing drinking water requires energy for pumping and treatment, chemicals for disinfection and filtration, labour, infrastructure, and distribution networks (Stokes and Horvath, 2009; Lundie et al., 2004; Friedrich et al., 2012). These processes also generate environmental impacts, such as emission, chemical use, and resource depletion. To reflect these wider costs, the CBA treats C_{pot} as the full societal cost of freshwater provision, not merely the price charged to customers (Boardman et al., 2018; Berbel and Expósito, 2020; EC, 2014). This ensures that the model internalizes the true resources savings that occur when GW reuse reduces freshwater demand (Molinos-Senante et al., 2010; Rogers, 2002; OECD, 2015).

C_{pot} is a critical input to the overall evaluation of GW management strategies. It reflects both the direct financial value of avoided potable-water purchases for ports and the broader societal value of reducing freshwater production. Including C_{pot} in the analysis ensures that the CBA captures one of the core advantages of GW reuse, its contribution to circular water management and resource efficiency in Baltic Sea port operations.

2.4.2 - Cost of Onboard Storage (C_{Stor})

Greywater generated on board passenger ships is temporarily retained in storage tanks prior to treatment or discharge, depending on the management option applied. The design and sizing of such tanks depend on several factors: vessel type and capacity, the number of passengers and crew, the duration of voyages between port calls, and whether greywater streams (e.g., from accommodation, galleys, and laundry) are collected in a combined system or separated into distinct streams. Separation typically requires either multiple smaller tanks or compartmentalized designs, while mixed greywater systems can rely on a single larger storage volume.

According to MARPOL Annex IV (IMO, 2003), ships engaged in international voyages are required to be equipped with holding tanks of sufficient capacity to retain sewage (and in many cases greywater, when not immediately discharged) when discharge is not permitted. Although MARPOL does not specify a fixed volume, capacity should be adequate for the maximum expected retention time, considering passenger numbers and crew. The HELCOM Guidelines for the Baltic Sea Region (HELCOM, 1990) further operationalize this by requiring tank capacity calculations based on daily wastewater generation rates, typically in the range of 100 – 150 L per person per day for passenger vessels. For RoPax ships, this implies that storage capacity must cover both the accommodation and catering services for passengers and crew, as well as contributions from vehicle deck washing and galley activities.

The cost of greywater storage onboard can be derived from the capital cost of tank construction and associated operating costs. Capital costs include the fabrication and installation of steel tanks, coating/Lining for corrosion protection, piping, pumps, and monitoring systems. Operating costs are comparatively small, mainly related to pumping energy and periodic maintenance (e.g. cleaning, inspection). For cost chain modeling, these can be annualized over the design life of the tank.

For a moderate-sized RoPax ship with around 1,500 persons on board, using a greywater generation rate of ≈ 200 L/person/day (within the 157 – 235 L/person/day range reported for passenger ships; (Kalnina et al., 2022), daily greywater production would be ≈ 300 m^3. If the required holding time were 1.5 days (36 hours) as with RoPax ships calling at the Trelleborg port, the required storage capacity would be ≈450 m^3. Using benchmark industrial installed tank costs of ≈US$100-300/$m^3$ (Thunder Said Energy, 2023) as a starting point and applying a marine-sector cost premium (for materials, certification, retrofitting), one might expect installed tank cost in the order of €500 - €900 per m^3, giving CAPEX in the range €225,000 - €405,000. When annualized over 15 – 20 years and adding moderate OPEX (pumping and maintenance), this yields a storage cost estimate of ≈ €12,000 - €30,000/year (on average ≈ €21,000/year). In the absence of cost data specific to greywater storage tanks on ships, this was the most appropriate cost estimate used in the cost analysis.

2.4.3 - Cost of Onboard Treatment (C_{AWTP})

Most AWTPs combine mechanical screening, biological treatment (often activated sludge or membrane bioreactor (MBR)), and tertiary polishing (filtration, disinfection, sometimes chemical or membrane steps) to produce effluent that meets stricter discharge standards than basic Type II Marine Sanitation Devices (MSDs). Vendors design AWTPs to fit constrained shipboard spaces and to handle combined black- and GW flows continuously or in batch mode. Sanitary wastewater management on board some passenger ships in the Alaska Region depends on how the AWTPs are designed. For instance, aboard Island Princess where Hamworthy AWTP is utilized, galley, food pulper, and laundry greywater streams are first collected in double-bottom holding tanks and held untreated before being discharged overboard 12 nautical miles from the shore, while accommodation GW is treated together with sewage before discharged (USEPA, 2006a). Moreover, Norwegian Star uses the Scanship AWTP which employs aerobic biological oxidation, followed by dissolved air flotation (DAF) and ultraviolet (UV) disinfection for wastewater treatment. In this system, wastewater from the galley, accommodations, laundry, and the collection, holding, and transport (CHT) subsystems is combined in a single holding tank, whereas food pulper wastewater is discharged untreated beyond 12 nm from shore (USEPA, 2006b). Aboard the Holland America Veendam, a Zenon AWTP is installed. This system employs aerobic biological oxidation, followed by ultrafiltration and ultraviolet (UV) disinfection. Wastewater originating from laundry, accommodations, food pulper, and galley subsystems is directed into two greywater storage tanks, while sewage is collected in four separate sewage tanks. The greywater and sewage streams are then combined in a common pipeline, pass through two parallel coarse screens, and enter a collection tank. From this tank, the wastewater is pumped into two aerated bioreactors and membrane chamber treatment trains, which operate in parallel (USEPA, 2006c). On the Holland America Oosterdam, the ROCHEM Bio-Filt® treatment system is in use. Like the Zenon system, it relies on aerobic biological oxidation, ultrafiltration, and UV disinfection to process highly concentrated wastewater from sewage, galley, and membrane concentrate generated by the ROCEHM greywater treatment system (USEPA, 2006d).

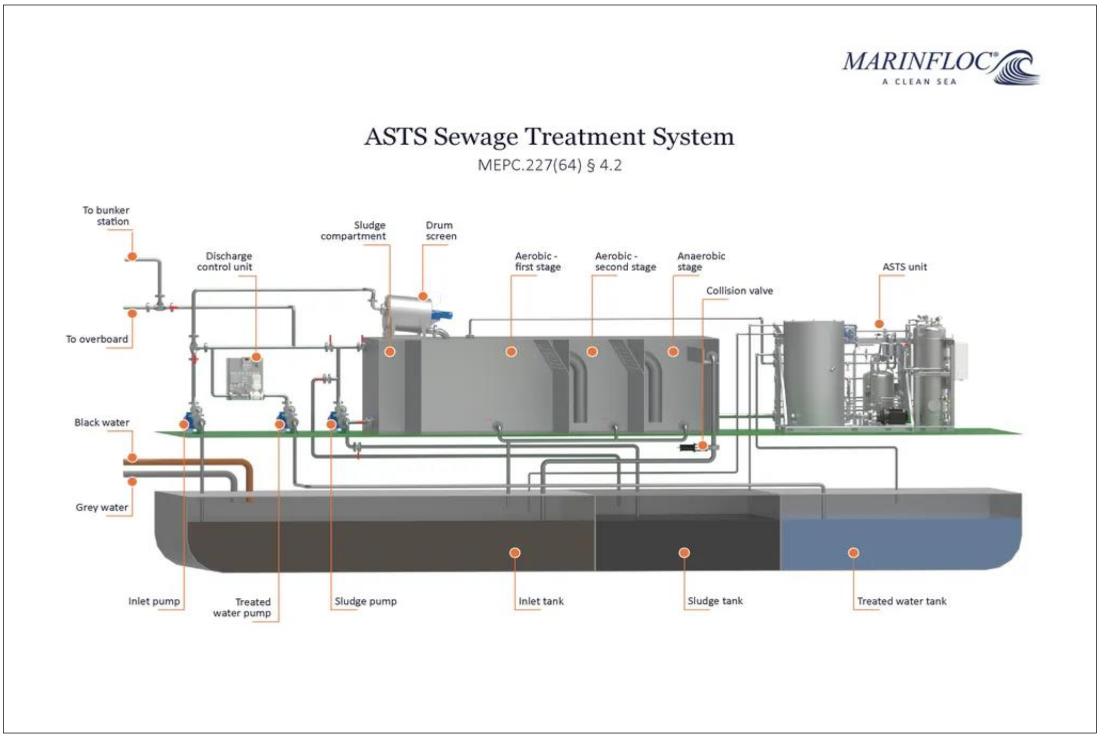


Figure 6: Marinfloc Advanced Sewage Treatment Pant

In the Marinfloc AWTP for example, the neptumatic system is divided into four major treatment states: 1) Mechanical, 2) Biological, 3) chemical flocculation and forced flotation, and 4) Filtration and sterilization. Blackwater and greywater are collected in a hull-integrated tank, after which treatment occurs in the Advanced Sewage Treatment Plant. The process begins with mechanical screening, which removes coarse solids and debris. Next, the bioreactor stage uses bacterial activity to break down organic matter, thereby reducing biochemical oxygen demand (BOD) and nitrogen. This is followed by chemical flocculation and forced flotation, where suspended solids and phosphorus are removed as sludge separates from the water. In the final stage, the effluent undergoes filtration and ultraviolet (UV) sterilization, effectively removing residual particles and neutralizing bacteria and viruses before discharge or reuse.

MARPOL Annex IV sets standards and operational rules for sewage from ships and requires that sewage systems be approved by the flag administration; it does not specifically regulate greywater as a separate stream. Nevertheless, many cruise lines operate AWTPs on board to meet stringent regional permits (e.g. Alaska) or company policies that go beyond Annex IV. Regional permits (e.g. Alaska Department of Environmental Conservation (ADEC) have driven AWTP adoption and influence on regulatory effluent limits (which vary by region), required hydraulic capacity (m^3/day), composition of incoming streams (relative share of greywater versus blackwater), ship space and power constraints, and redundancy/monitoring needs.

The most detailed cost analysis for AWTP we found in the literature was estimated in the ADEC/cruise-industry permitting and feasibility work (prepared for Alaska cruise ship permitting) which gives a survey/estimate range of wastewater treatment costs on passenger ships (ADEC, 2012). From this report, we calculated the average CAPEX and OPEX of 15 passenger ships and obtained \$5.60/m^3/year after index adjusting from June 2011 to July 2025. Costs were annualized using an annuity factor with n = 15 years and discount rate, r = 5% (Boardman et al., 2018). There exists variability in system types and differing accounting of costs. We adopted this cost as a primary literature anchor for AWTP unit costs, although the report stated that the shipowners did not clearly state all the cost components included in

CAPEX and OPEX. Vendor descriptions of shipboard AWTP systems (e.g. Marinfloc Neptumatic ASTS) confirm that AWTPs are compact, bespoke systems whose costs depend strongly on capacity, technology, certification requirements, and vendors typically price and quote on a project basis rather than publishing a single cost/m^3 figure. This vendors' information justifies that the Alaska survey range is plausible for shipboard AWTPs and that larger systems and stricter regional limits could push costs to the higher end.

2.4.4 - Cost of Port Reception (C_{PRF})

PRFs are shore-based installations where ships can deliver ship-generated waste, including sanitary wastewater (black water and greywater) for appropriate handling and treatment. Their establishment and operation are regulated under MARPOL Annex IV, which requires that ports provide adequate reception facilities for sewage to meet the needs of ships regularly calling at the port without causing undue delay. Although greywater is not explicitly regulated under Annex IV, many ports in the Baltic Sea region integrate it into sewage PRFs where collection systems and treatment pathways are available. In the EU, the operation of PRFs is further defined by Directive (EU) 2019/883 on port reception facilities for the delivery of waste from ships. The directive establishes a harmonized framework across EU ports to ensure that adequate facilities are in place and that delivery of ship-generated waste becomes the standard practice, thereby reducing the risk of illegal discharges into the marine environment.

A central feature of the PRF Directive is the cost recovery system, outlined in Article 8 and Annex 4. The directive requires the use of a "no special fee" system, meaning that most of the costs for PRF provision are recovered through a mandatory indirect fee paid by all ships, regardless of whether they use the facility. This approach ensures cost-sharing among ship operators while incentivizing the regular delivery of waste to shore. In addition to the indirect fee, ports may also apply direct fees, especially for waste streams that require special handling or for volumes exceeding what is considered a reasonable quantity. Through this combination of indirect and direct charges, ports can recover a significant portion of their capital and operational expenditures associated with wastewater PRFs.

In the case of greywater, the port handling chain generally involves pumping from onboard storage tanks to the PRF connection point and either channeling to the PWTP for pre-treatment, further to the MWTP for final treatment or directly channeling to the MWTP for treatment and final discharge into the sea. From a stakeholder perspective, the cost of PRF handling therefore includes the indirect fee paid by all ships, any direct fees levied on greywater delivery, and the operational expenses incurred by ports for pre-treatment, pumping, and transfer, which are partially recuperated through the fee system. From a societal perspective, however, PRFs form part of the shared waste management infrastructure, and the costs are redistributed across the maritime sector through the fee structure. In the context of the cost-benefit analysis, what is most relevant at the societal level is the balance between the environmental benefits achieved by diverting greywater from direct discharge and the overall system costs of its collection and management.

For this study, the cost of port reception and handling of greywater is expressed as a unit cost per cubic meter in SEK. This value already incorporates both capital and operational expenses in accordance with Annex 4 of the PRF directive and are therefore suitable for direct use in the cost chain model calculations of the alternative scenarios. Because the port of Trelleborg does not publish a fixed tariff for wastewater reception, comparative estimates were derived from other major Swedish passenger ports. Tariffs from the Ports of Stockholm (2025), Port of Gothenburg (2025), and Port of Ystad (2025) were applied to the study's operational profile of 5,460 RoPax calls per year and an annual greywater volume estimate of 40,477m^3. The Stockholm tariff (21.8 SEK/m^3) represents a purely volumetric model that scales linearly with service volume. The Gothenburg tariff combines a volumetric charge (45 SEK/m^3) with a per-call connection fee (880 – 2,680 SEK/call), while Ystad applies a flat 2,500SEK per call fee for discharges exceeding 5m^3. When these structures are applied to Trelleborg's high-frequency, low-volume RoPax operations (≈ 7.4 m^3 per call), the per-call tariffs yield disproportionately high effective costs (≈164 – 406 SEK/m^3 for Gothenburg and 337 SEK/m^3 for Ystad). Such flat or mixed-fee systems overstate the economic burden per cubic meter of wastewater handled, distorting the cost-benefit relationship between service cost and environmental gain. To maintain economic proportionality and ensure that the valuation of avoided marine discharges reflects realistic operational costs, the Stockholm volumetric tariff (≈ 22 SEK/m^3) was adopted. This provides a realistic benchmark for modeling purposes while acknowledging the actual costs may vary depending on operational factors and additional handling requirements.

2.4.5 - Cost of Port Treatment (C_{PWTP})

At the Trelleborg port, reception facilities are complemented by a port-based wastewater treatment plant (PWTP) which provides a critical pre-treatment step before the wastewater is transferred to MWTP. The rationale for operating such a PWTP is that most MWTP are optimized for domestic sewage and are not specifically designed to remove heavy metals or other contaminants that may be present in ship-generated wastewater. Pre-treatment at the port level therefore reduces the risk of exceeding influent quality thresholds at the MWTP and ensures compliance with local discharge and permit requirements. According to ABVA Trelleborg (2009) "the VA is not obliged to receive wastewater whose nature differs to a significant extent from that of domestic wastewater. To discharge such wastewater into the public water supply network, an agreement is usually required. …. Property owners are obliged to report to the VA any activities that may affect the composition of the wastewater". This implies that the MWTP might reject wastewater from ships if its quality is significantly different from domestic wastewater in terms of contaminant concentrations. The cost of wastewater treatment at PWTPs includes CAPEX and OPEX. This, as with MWTPs could be expressed in SEK/m^3.

The annual societal cost of the PWTP for greywater treatment from RoPax ships in Trelleborg was estimated using parametric unit-cost methodology. It includes the annualized capital expenditure (CAPEX) and the annual operating expenditure (OPEX). The societal cost is the sum of these two components. Capital and operating costs of the PWTP at Trelleborg were estimated using a parametric approach consistent with standard engineering economics (Peters et al., 2003; Turton et al., 2018). Unit capital costs ranges for physico-chemical treatment (chemical precipitation, dissolved air floatation) were drawn from US. EPA (2008), Metcalf and Eddy (2014) and WEF (2012), expressed per unit design capacity (€/m^3), with a 30% integration multiplier for civil works. OPEX estimates (€/m^3 treated) incorporate energy, chemicals, sludge disposal and labour, following ranges reported by U.S. EPA (2017) and WEF (2012). The Trelleborg PWTP process falls into the mid-range category (moderate CAPEX, moderate OPEX), therefore mid-band unit costs of €1, 000/m^3/day (CAPEX) and €1.5/m^3 (OPEX) were adopted for the base case, with low/high values tested in sensitivity analysis. Costs were annualized using an annuity factor with n=15 years and discount rate, r = 5% (Boardman et al., 2018). Volume data for greywater at Trelleborg PRFs were obtained from Trelleborg port via personal communication, with an 80:20 split for GW:BW taken from Mujingni et al. (2024). The levelized cost was obtained as ≈ 20.24 SEK/m^3.

2.4.6 - Cost of upgrading PWTPs to reuse standards (C_{reuse})

Reuse of greywater has been advocated for within the water and wastewater industry, encouraged by the EU urban wastewater treatment directive, and the Regulation (EU) 2020/741 of the European Parliament and of the Council of 25 May 2020 on minimum requirements for water reuse (EU, 2020). To estimate the cost of reuse of greywater treated at the PWTP in Trelleborg additional costs should be considered which could include: cost of additional treatment units (UF/activated carbon/disinfection), storage tanks, pumps, pipes, fittings, valving, instrumentation, controls; connection cost to the port buildings, as well as extra OPEX (energy, chemicals, labor, maintenance, sludge handling). In addition to treatment systems and operations and maintenance costs, the cost of treatment to meet these standards also includes building retrofitting costs. Like the Marinfloc onboard ASTS system, the effluent obtained after the flocculation process might need a post filtration and sterilization stage if discharging in sensitive areas like the Baltic Sea. The Marinfloc ASTS system sterilizes the effluent by means of free radicals that are produced by UV-light on a titanium oxide surface (Gombril & Eriksson, 2016). As the UV-light is involved the post filtration process would be achieved by two parallel sand filters. The addition of sand filters and UV would increase the CAPEX in the reuse scenario.

Regarding the cost of reuse of greywater treated at the PWTP in Trelleborg, incremental cost of reusing PWTP-treated GW at the Waterfront building at the Trelleborg port for non-potable uses (e.g., WC flushing, cleaning), we include additional CAPEX for (i) polishing treatment (e.g. UF for fine solids/pathogens, GAC for organics/odor, and disinfection, typically UV; some systems may apply UV-based advanced oxidation with photocatalysis such as TiO_2 in sensitive areas), (ii) a reuse day-tank and booster pumps, (iii) pipes/fittings/valving, instrumentation and controls (I & C), and (iv) building connection/retrofit (dual plumbing to non-potable circuits). We also add extra OPEX for energy, chemicals, cleaning, media replacement (e.g., GAC, lamp replacements, labour, maintenance, and any sludge/backflush handling from polishing. Swedish and European guidance not that membrane-based polishing with UV is a common train for safe non-potable reuse; post-filtration and sterilization are recommended for sensitive regions (e.g., Baltic Sea requirements for stringent effluent) and dual plumbing needs can drive retrofit costs in existing buildings. Accordingly, we model three per-m^3 proxy levels for the reuse add-on (C_{reuse}): Low - 2 SEK/m^3, Mid - 5 SEK/m^3, High - 10 -12 SEK/m^3. These reflect amortized CAPEX (10 year life, real 4% discount) plus routine OPEX and map to typical component ranges reported in European reuse/Life Cycle Costing (LCC) studies: Granular Activated Carbon (GAC) polishing often contributes ≈ 2–3 SEK/m^3 when media and handling are annualized; UF/UV energy and consumables are typically <1 – 2 SEK/m^3 at building scale, with balance from pumping, maintenance, Instrumentation and Control (I & C) and retrofit overheads. For SC5A (100% reuse), we used C_{reuse} = 5 SEK/m^3 as the reporting base, and show 2 / 5 / 10 – 12 SEK/m^3 in sensitivity to span minimal (existing pipe runs, small tank) through robust Utrafiltration (UF) + GAC + Ultraviolet disinfection (UV) + Advanced Oxidation Processes (AOP), larger tank, deeper dual-plumbing implementations (Gombril & Eriksson, 2016; Arden et al., 2024; Fredenham et al., 2020).

2.4.7 - Cost of Municipal Treatment (C_{MWTP})

After passing through the Port of Trelleborg's own wastewater treatment plant, the treated wastewater is conveyed to the MWTP for further treatment and final discharge. This connection is essential to ensure that the final effluent meets all environmental requirements and does not negatively impact on the local aquatic environment. Other sources of wastewater as defined in the UWTD are also channeled to and treated at the MWTP. To enable this transfer, permits are required from the relevant supervisory authority, which in Sweden is the County Administrative Board (Länsstyrelsen). The Board ensures that all wastewater discharges comply with environmental legislation and do not cause harm to the receiving environment. Furthermore, the treated wastewater must meet specific quality requirements before it can be discharged. These include: 1) Urban Wastewater Treatment Directive (UWWTD) which establishes minimum treatment standards for wastewater discharges from urban areas with population over 2000 pe; 2) P95 standards (Svenskt Vatten standards), which defines the maximum allowable concentrations of various pollutants in wastewater at the 95th percentile of measured values; and 3) ABVA (Allmänna Bestämmelser för Vatten och Avlopp) Trelleborg, which is the municipality's general regulations for water and wastewater, setting local discharge requirements.

Box 1: Formula for calculating Societal Cost of modelled scenarios

SC1: $C_{Pot} + C_{Stor}$

SC2: $C_{Pot} + C_{AWTP}$

SC3: $C_{Pot} + C_{Stor} + C_{PRF} + C_{MWTP}$

SC4: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP} + C_{MWTP}$

SC5A: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP}$

SC5B$_1$: $C_{Stor} + C_{PRF} + C_{PWTP} + C_{reuse}$

SC5B$_2$: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP} + f_{75} * C_{reuse} + (1 - f_{75}) * C_{MWTP}$

SC5B$_3$: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP} + f_{50} * C_{reuse} + (1 - f_{50}) * C_{MWTP}$

SC5B$_4$: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP} + f_{75} * C_{reuse}$

SC5B$_5$: $C_{Pot} + C_{Stor} + C_{PRF} + C_{PWTP} + f_{50} * C_{reuse}$

Where:

C_{Pot} = Cost of water purchase

C_{Stor} = Cost of onboard storage and maintenance of storage tanks

C_{AWTP} = Cost of onboard treatment in AWTP

C_{PRF} = Cost of Port reception and handling

C_{PWTP} = Cost of treatment at PWTP

C_{MWTP} = Cost of treatment at MWTP

C_{Reuse} = Cost of making treated greywater reusable. That is, upgrading the PWTP infrastructure

f = Reuse fraction

In Sweden, MWTPs are legally classified as environmental hazardous activities under Chapter 9 of the Environmental Code (SFS 1998:808). This means that MWTPs cannot operate without a valid environmental permit, which is issued either by the County Administrative Board (Länsstyrelsen) or for larger facilities, by the Land and Environment Court. The permit process requires MWTPs to demonstrate compliance with all the relevant requirements previously listed. Operators are obliged to apply precautionary principles, monitor and report their discharges, and ensure that effluents and sludge are managed in an environmentally sound way. These regulatory requirements form part of the cost structure of wastewater treatment, as compliance involves investment in monitoring systems, treatment processes, and reporting routines, in addition to the direct costs of conveyance and treatment.

To estimate the cost of treating greywater at the MWTP in Trelleborg, we used information from the municipality's water and wastewater tariff (31.73 SEK/m^3 where 50% is related to sewage so 15.87 SEK/m^3 and 50% to freshwater). There is also an additional fee for concentrations differing from municipal wastewater concentrations, applying these tariffs would, based on inlet concentrations in Table 3, result in ≈ 8.3 SEK/m^3. This cost is, however, reduced to ≈ 2.20 SEK/year by having a PWTP that reduces the concentrations of P, N and COD. Therefore, in scenarios having PWTP treatment the port would pay ≈ 18,87 SEK/m^3 for municipal treatment, and ≈ 24,17 SEK/m^3 where PWTP is absent.

2.4.8 - Savings from not paying wastewater discharge fee to MWTP (S_{pot})

When the port reuses greywater instead of sending it to the MWTP, it reduces the volume of water discharged to the MWTP. The Trelleborg municipality charges fees for wastewater discharge based on volume, strength (BOD/COD), or both (Trelleborgs kommun, 2025).

By diverting greywater for reuse, the port avoids part of these fees. This is a direct cash saving for the port operator. In this estimate, we include only those fees that are directly avoided. That is, 1) volume-based wastewater charges (SEK/m^3) applied to water entering the municipal sewer; 2) strength/quality-based surcharges (if the MWTP charges extra for high BOD, COD or nutrients) attributable to the reduced volume. We do not include fixed sewer fees or other municipal charges that do not vary with discharge volume.

Furthermore, a stakeholder perspective is adopted to illustrate the financial distribution among shipowners, ports, and municipalities. While transfers like port fees cancel out at the societal level, they greatly influence the budgetary incentives and feasibility for each actor. From this perspective:

1) Shipowners incur costs from onboard infrastructure and pay PRF fees at ports for handling greywater.

2) Ports collect reception fees from shipowners, bear PRF and PWTP costs, and remit connection and treatment fees to municipalities where applicable, and

3) Municipalities incur treatment costs and receive remittances from ports.

This breakdown clarifies whether high-efficiency scenarios are economically viable for each actor, and highlights areas for policy intervention (e.g. cost sharing, subsidies) if societal welfare gains are misaligned with private incentives (Kinell et al., 2012). A dual-perspective approach enhances transparency and supports the design of equitable, implementable policy solutions (International Institute for Environment and Development (IIED), 2013). Table 2 shows the formula for calculating stakeholder distributional cost of the modelled scenarios.

2.5 - Environmental Benefits of Ship-generated Greywater Management

Environmental benefit of greywater treatment was computed using both shadow prices derived via shadow price modelling and existing shadow prices of contaminants obtained from the literature. This is known as shadow price valuation methodology (SPV). For each scenario, the environmental benefits were obtained from a series of derived formulae shown in Box 2.

2.5.1 - Shadow Price Modelling

The wastewater treatment process at the MWTP is like a production process where there exist input components essential in carrying out wastewater treatment. It consists of energy, staff, chemicals, maintenance and waste handling. An output component consisting of a desirable output (treated effluent) and undesirable outputs which are the pollutants targeted for this study. They include P, N, COD, BOD, TSS, Pb, Cd, Zn, Cu, Cr, Ni, PET and PP. These undesirable outputs are the main pollutants driving the hazard potential of GW (Mujingni et al., 2024). They are considered undesirable because their discharge into the marine environment would cause adverse effects.

Several databases were examined among WATERBASE, Swedish pollutant release and transfer register and HELCOM for usable data for shadow price calculation, however none of the databases had the complete information required. Finally, the project decided to go along with the data found in Svenskt Vatten database, VASS, where MWTPs were sampled and data on their operation was obtained. In calculating the shadow price of contaminants from MWTPs, it was necessary to assess several MWTPs. The rule of thumb is that the number of MWTPs assessed should be at least thrice the number of contaminants examined (pers. Comm. with an expert). As such a sample of 517 MWTPs operating within the Baltic Sea, including the Trelleborg municipal wastewater treatment plant, was examined. All the plants have primary and secondary treatment processes where nitrogen and phosphorus are removed. However, only some of the plants operate the tertiary treatment step. The data collected was scrutinized, cleaned and preprocessed before shadow price modelling.

A major challenge was data handling. The original dataset contained 517 MWTPs, but included problematic entries such as implausible values (e.g., very small plants reporting costs an order of magnitude higher than larger plants), unclear cost reporting (where several categories were aggregated), and negative or zero values. This dataset was therefore manually cleaned by correcting clearly misreported negative cost entries and excluding plants where the total cost per PE was unreasonably high (suggesting reporting errors) or where treated effluent per PE was clearly misreported. The manual screening also addressed several potential outliers, hence reduced the dataset substantially, leaving 91 MWTPs. The sizes of the plants ranged from 2,132 population equivalents (p.e.) to 873200 p.e. and the volume of wastewater treated in the plants ranged from 0.024 to 106 Mm^3/year. The statistical results of the dataset is shown in Appendix 2 Table 4.

Despite this cleaning, 52 plants still had at least one zero or negative entry and 3 MWTPs had evident misreported values. Hence these plants were removed leaving 36 MWTPs usable in the strictest case. As a result of the problematic dataset, only 28 MWTP data-points remained in the strictest case after cleaning out zero-valued and filtering outliers according to the percentile or standard deviation methods. Outlier filtering was also applied based on percentile (or Standard Deviation) thresholds, where a reasonable threshold could be considered in the range 1^{st} to 5^{th} percentiles removed (or about 1.5 - 2 standard deviations). Particularly datapoints from the large plants (p.e. > 150,000) were flagged as outliers.

The overall framework for deriving shadow prices from distance functions follows Färe et al. (1993). The flexible translog functional form used to represent production technology is described by Christensen, Jorgenson and Lau (1973). Because the distance function includes log-transformed variables, zero (and negative) observations cannot be used directly. To retain observations (like a plant reporting no chemical cost) while ensuring strictly positive arguments for the log transformation, we applied an additive offset (pseudocount), $x^*=x+\varepsilon$. This type of shifted-log transformation is consistent with the shifted forms discussed in the Box-Cox transformation family (Box and Cox, 1964; Atkinson et al., 2021). We set ε proportional to each variables scale (a fraction of the mean of its positive, non-zero values) to minimize distortion while avoiding data loss. Variables with very small magnitudes were rescaled where needed to prevent the offset from dominating their variation. As alternatives to offset-based logging, methods such as the inverse hyperbolic sine transformation (Bellemare & Wichman, 2019) and the Yeo-Johnson transformations (Weisberg, 2001) are also commonly used to accommodate zeroes, however we retained the log-based specification for consistency with the chosen translog/distance-function formulation. Results were checked for robustness to the choice of ε using multiple fractions of the mean.

Thus, all 88 MWTPs, remaining after exclusion of clearly incomplete or misreported values, could be used in the subsequent stages. The dataset was screened for outliers using standardized deviations from the mean (z-scores), that is observations exceeding a selected multiple of the sample standard deviation. Such SD-based screening rules are common in applied outlier detection and are closely related to formal procedures such as Grubbs´ test under normality assumptions (Grubbs, 1950; Grubbs, 1969). We evaluated thresholds of ±1.5, ±2 and ±3 SD and found that ±2-3 SD still retained extreme observations that led to implausible and unstable optimization results. Therefore, ±1.5 SD was used as a stricter data-quality screening rule. After outlier removal, 82 MWTPs remained.

The modelling of shadow prices was carried out in a Python environment. We implemented optimization using Pyomo with the Interior Point OPTimizer (IPOPT) solver, which is suitable for non-linear programming problems. After each optimization run, programmatic checks ensured that all constraints derived from the methodological framework were satisfied.

Since the log transformation used in the distance function is undefined at zero (or for negative numbers), handling zero-values was not optional. Several correction strategies were considered:

- Exclusion of invalid points
- Small shift (+0.0001) to make log(x) defined
- Shift based on mean - using a fraction the mean of the positive and non-zero values
- Rescaling smaller pollutants where shifts would otherwise distort results,

as a final step of data processing was to filter out any remaining outliers that might skew the results in any direction. The process of identifying and removing outlier datapoints from the input and output variables was implemented as a function in the modelling section, where outliers found outside of a selected range of standard deviation or percentiles were highlighted and removed before optimization.

The method implemented by Hernández-Sancho et al. (2010) estimates several terms from the data according to the input-output space of the problem. In this framework, N represents the number of inputs and M represents the outputs (desired and undesired). The parameters estimated by the model are:

• The constant (offset)

• Linear terms related to inputs (N)

• Linear terms related to outputs (M)

• Quadratic input terms, relating the inputs internally

• Quadratic output terms, relating the outputs internally

• Cross input–output terms, relating the inputs and outputs

In addition, symmetry, homogeneity and sum constraints reduce the number of free parameters. With the selected inputs and outputs, the optimization estimated in total 78 parameters, of which 66 were free after constraints.

Finally, the ratio between the effective amount of information in the dataset and the number of free parameters provides a useful indication of whether the estimation problem is under-, well- or overdetermined. If the problem is effectively underdetermined, insufficient independent information relative to the number of free parameters, solutions may be non-unique or highly sensitive to "noise" and data perturbation. If the problem is only marginally determined, observations close to parameters, estimates can still be unstable particularly under measurement "noise" collinearity. In contrast, when the problem is sufficiently overdetermined and well-conditioned, the estimation is typically more stable and the implied shadow prices tend to be more robust.

2.5.2 - Environmental benefit of Pollutant Removal ($EB_{Pollution}$)

The Shadow Prices obtained, together with others obtained from the literature, the average volume of effluent released from the Trelleborg MWTP from 2021 to 2023, and the volume of pollutants eliminated during the same period were used to calculate the environmental benefit of the modelled scenarios, expressed as price per m^3 of treated effluent and per annum (Hernandez-Sancho, 2019). Box 2 shows the formula for calculating the environmental benefit of GW treatment, and the modelled scenarios.

The need to meet the growing demand for water resources, while preventing further degradation of ecosystems and natural processes results in issues that should be addressed from an integrated perspective. Therefore, implementation of this outcome in the field of water management to guarantee sustainability and quality of life in the present and future is relevant. Environmental benefits can be used to justify investments in technical improvements in MWTPs, specifically new technologies that achieve better quality of wastewater effluent such as those designed to remove metals which current MWTPs lack.

2.5.3 - Environmental benefit of Reuse (EB_{reuse})

In addition to pollution-removal benefits, the reuse scenarios generate a notable environmental benefit (EB_{reuse}) associated with freshwater substitution. The formula for calculating EB_{reuse} is shown in Box 3. This benefit captures the environmental value of avoiding the abstraction and supply of freshwater that would otherwise be required to meet shipboard demands. The magnitude of EB_{reuse} depends directly on the volume of GW that is reused and it involves only the reuse scenarios.

Box 2: Formula for calculating Environmental Benefit of Pollutant Removal ($EB_{Pollution}$).

Environmental benefit of each scenario is given by:

SC1:	$\sum (M_i \times 0 \times SP_i)$
SC2:	$\sum (M_i \times R_{i\_AWTP} \times SP_i)$
SC3:	$\sum (M_i \times R_{i\_MWTP} \times SP_i)$
SC4:	$\sum (1 - (1 - R_{i\_PWTP}) \times (1 - RE_{i\_MWTP})) \times M_i \times SP_i)$
SC5A:	$\sum (M_i \times R_{i\_PWTP} \times SP_i)$
$SC5B_1$:	$\sum (M_i \times 1 \times SP_i)$
$SC5B_2$:	$\sum (1 - (1 - R_{i\_PWTP}) \times (1 - RE_{i\_MWTP})) \times M_i \times SP_i \times (1-f_{75}))$
$SC5B_3$:	$\sum (1 - (1 - R_{i\_PWTP}) \times (1 - RE_{i\_MWTP})) \times M_i \times SP_i \times (1-f_{50}))$
$SC5B_4$:	$\sum (R_{i\_PWTP}) \times M_i \times SP_i \times (1-f_{75}))$
$SC5B_5$:	$\sum (R_{i\_PWTP}) \times M_i \times SP_i \times (1-f_{50}))$
Where,	
i	Pollutant type (e.g. BOD, COD, TN, TP, MPs)
M_i	Mass of pollutant i in the greywater generated on board (kg)
SP_i	Shadow price of pollutant i (SEK/kg)
R_{i_MWTP}	Removal efficiency for pollutant i in municipal wastewater treatment plant
R_{i_PWTP}	Removal efficiency for pollutant i in port wastewater treatment plant
f_{reused}	Reuse fraction

This component, often represented as the shadow price of freshwater (SP_{water}), captures the economic value of replacing potable freshwater with reclaimed greywater in the reuse scenarios. In the context of this study, the Port of Trelleborg uses the reclaimed water at the port for non-potable purposes such as toilet flushing, therefore the port no longer needs to purchase an equivalent volume of freshwater from the municipal drinking-water utility.

The avoided expenditure constitutes a direct financial saving for the port and is treated as a benefit in the CBA. From a broader perspective, the benefit extends further than simple tariff avoidance. Every cubic meter of potable water that is not produced and distributed also avoids upstream resource use: raw-water abstraction, chemical dosing, filtration, disinfection, energy consumption in pumping and pressure maintenance, as well as network-related losses. These avoided system-wide externalities form the conceptual basis for treating freshwater substitution as an environmental and resource benefit, even though only the port captures the monetary savings under current tariff structures.

2.6 - Net Societal Benefits of Ship-generated Greywater Management

The Net Profit of wastewater treatment at the MWTP is the difference between the total environmental benefit and the total societal cost. If the result of the computation is $NP>0$, then the system is economically viable, while $NP<0$ means the system is not economically viable. Comparatively, the best system refers to the one with the highest NP (Molinos-Senante et al., 2010).

Box 3: Formula for calculating Environmental Benefit of Reuse (EB_{Reuse})

Environmental benefit of Reuse is given by:

$$EB_{Reuse}\,(S_{pot,port}) = SP_{water} \times Q_{Reuse}$$
$$Q_{Reuse} = f_{reuse} * Q_{GW}$$

Where,

SP_{water}	Shadow price of freshwater, determined as the avoided potable-water tariff charged by the municipal supplier (SEK/m^3)
Q_{Reuse}	Annual volume of GW reused at the port (m^3/year).
Q_{GW}	Total volume of GW treated at the port (m^3/year)
f_{reuse}	Reused fraction

For each scenario, the environmental Benefit due to reuse of treated GW is added to the environmental benefit pollution removal (EB_{Reuse} + $EB_{Pollution}$) to obtain the total environmental benefit. For non-reuse scenarios EB_{reuse} is zero. The Net Societal Benefit is calculated using the formula in Box 4.

Box 4: Formula for calculating Net Societal Benefits

The ***Net Societal Benefit (NSB)*** **is given by*:***

$$\boldsymbol{NP = (EB_{Pollution\_i} + EB_{Reuse\_i}) - C_{Soc\_i}}$$

Where:

EB_{Reuse_i}	= Environmental Benefit of treated GW reuse in Scenario i
EB_{Reuse}	= 0 for non-reuse scenarios
$EB_{Pollution\_i}$	= Environmental Benefit of Pollutant removal of Scenario i
C_{Soc_i}	= Societal Cost of scenario i
i	= Scenarios $SC_1 - SC5B_5$

3 - Results

This section presents the results of the socio-economic and environmental evaluation of ten GW management scenarios for passenger ships operating in the Baltic Sea region. The analysis integrates (i) real-resource societal costs, (ii) financial stakeholder impacts; (iii) environmental benefits from pollution reduction and freshwater substitution; and (iv) the net societal welfare outcome (NSB). Results are interpreted in the context of existing scientific literature, empirical wastewater valuation studies, and Baltic Sea environmental policy objectives under HELCOM and EU Directives.

3.1 - Societal Cost Analysis

Societal cost reflects the total real-resource costs of each GW management scenario. Unlike stakeholder cost, which accounts for who pays, societal cost captures the true economic burden of potable water production, onboard storage, port reception, PWTP operation, and MWTP treatment. Because tariff transfers are excluded, societal cost is the most appropriate metric to evaluate economic efficiency.

Table 2: Societal cost of Greywater Management

SCs	Total (SEK/m^3)	Total (SEK/year)
SC1	26.98	1,093,243
SC2	68.48	2,775,155
SC3	67.05	2,717,200
SC4	87.28	3,537,486
SC5A	69.22	2,805,146
$SC5B_1$	74.22	3,007,786
$SC5B_2$	73.52	2,979,467
$SC5B_3$	66.28	2,686,297
$SC5B_4$	69.0	2,796,381
$SC5B_5$	63.78	2,584,977

SC1 (discharge to sea) unsurprisingly emerges as the scenario with the lowest societal cost. Because it avoids all treatment and handling beyond basic onboard storage, SC1's annual cost of roughly 1.09 million SEK represents a baseline of minimal resource use. This mirrors trends identified in Mediterranean wastewater studies such as Hernandez-Sancho et al. (2010) and Molinos-Senante et al. (2011), where no-treatment or primary-treatment baselines always produce the lowest resource costs.

Introducing on-board or port-based treatment significantly increases societal costs. SC2 and SC3, both single-stage treatment pathways (one at ship level, one at port/municipality), all between 2.7 and 2.8 million SEK/year. SC4 (PWTP + MWTP) demonstrates the highest burden at 3.54 million SEK/year, reflecting the cumulative resource intensity of two sequential treatment stages. Similar findings were reported in the Las Palmas study (Martinez-Lopez et al., 2020), where dual-node treatment configurations consistently produced the highest real-resource expenditures.

The reuse scenarios present a more nuanced cost structure. Full reuse ($SC5B_1$) generates a moderate societal cost (3.01 million SEK/year), moderately above single-treatment scenarios but well below SC4.

Partial-reuse scenarios routed partly to MWTP ($SC5B_2$ and $SC5B_3$) show some of the lowest costs among treatment-based pathways, particularly $SC5B_3$ (2.69 million SEK/yr), which benefits from PWTP polishing combines with the economies of scale inherent municipal treatment. Scenarios routing the non-reused fraction to sea instead of MWTP show slightly higher costs, reflecting the efficiency advantage of MWTP nutrient removal.

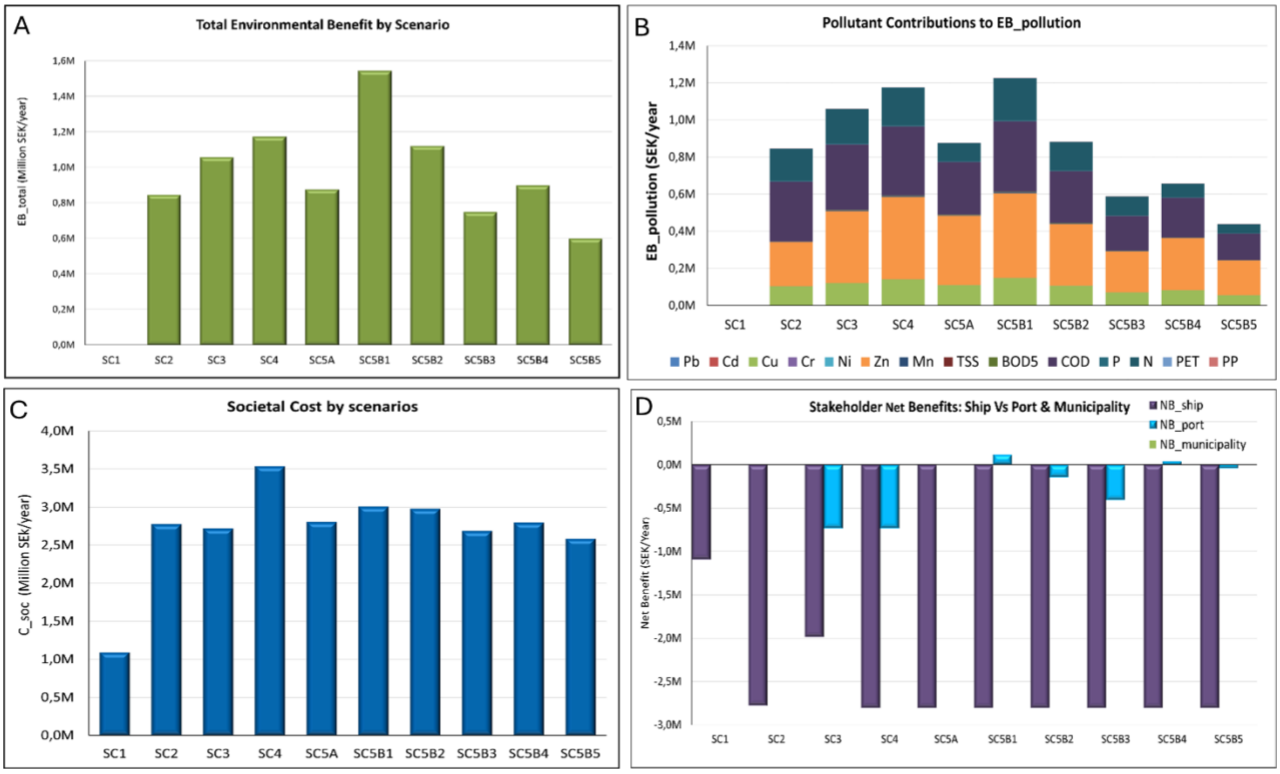


Figure 7: *A) Total Environmental Benefit by scenario, B) Pollutant contributions to Environmental Benefit of greywater treatment, C) Societal Cost by Scenarios, D) Stakeholder Net Benefit in the greywater management chain from "cradle to grave".*

Overall, societal costs show that, while untreated discharge is the least expensive, several reuse pathways, particularly $SC5B_3$, are resource-efficient and avoid the steep cost escalation associated with dual-treatment systems.

3.2 - Stakeholder Cost Analysis

While societal cost measures economic efficiency, stakeholder cost captures financial burdens for ship operators, ports and municipalities. These results reveal an extreme imbalance: ship operators consistently bear heavy financial losses, ports experience variable moderate gains or losses, and municipalities remain financially neutral.

Ship operators consistently incur the largest financial burden across all scenarios requiring treatment. Even the simplest onboard-treatment scenario (SC2) imposes roughly -2.77 million SEK per year. Complex configurations (SC3 – $SC5B_5$) impose costs of -2.81 million SEK annually. This outcome mirrors findings from Baltic Annex IV assessments (Wilewska-Bien, 2019; Peric, 2018), where ships consistently shoulder nearly all treatment-related costs due to tariff structures and limited cost-sharing.

Ports exhibit moderate financial variations depending on treatment destination and reuse. Port losses are most severe in SC3 and SC4 (-732,341 SEK/yr), where the port must pay MWTP fees exceeding ship tariffs. Small profits arise in $SC5B_1$ and $SC5B_4$ because reuse reduces water procurement needs. However, most reuse cases still result in net losses for the port. The Las Palmas analysis (Martinez-Lopez., 2020) similarly found that ports often under-recover costs when acting as wastewater handlers, even when providing significant environmental benefit.

Table 3: Stakeholder net costs and benefits of Greywater Management (SEK/year)

Modelled scenarios	NB_Ships	NB_Port	NB_Municipality
SC1	-1 093 243	0	0
SC2	-2 775 155	0	0
SC3	-1 984 859	-732 341	0
SC4	-2 805146	-732 341	0
SC5A	-2 805146	0	0
$SC5B_1$	-2 805146	+118 848	0
$SC5B_2$	-2 805146	-144 609	0
$SC5B_3$	-2 805146	-408 066	0
$SC5B_4$	-2 805146	+38 476	0
$SC5B_5$	-2 805146	-41 896	0

Contrarily, the municipality remains financially neutral across all scenarios, because the MWTP tariff is assumed to achieve full cost recovery, covering both CAPEX and OPEX costs scaled to treatment of GW from the port. As a result, municipal revenues exactly match total treatment expenditures, leaving the municipality with neither financial gain nor a loss under any GW management pathway. This modelling assumption reflects standard cost-recovery practices in Nordic wastewater utilities, where tariffs are designed to recover the full long-term marginal cost of service provision, including depreciation, asset renewal, and operating expenses. Under such arrangements, utilities do not generate profit from additional treatment loads but also do not subsidize them (Martinez-Lopez et al., 2020).

The overall picture is clear: the actor who controls the system financially (ships) does not benefit environmentally or economically from treatment or reuse, while the actors who benefit (the port, and society at large) do not share the cost burden. This misalignment is a primary reason why advanced GW treatment and reuse have not been widely adopted in the Baltic region.

3.3 - Environmental Benefits Analysis

3.1.1 - Environmental Benefit of Greywater treatment

The modelling of shadow prices of contaminants shows that the dataset (Appendix 3) proved highly dispersed. For most cost categories, the mean values were comparable to those reported in the literature, but standard deviations were often 2–3 times larger than the mean. This instability meant that the model sometimes placed disproportionate weight on highly inconstant variables, leading to skewed shadow prices. For example, TOC showed very high variance and dominated the optimization, while small pollutants could be overwhelmed by scaling effects.

The model estimated approximately 78 parameters, of which 66 were free after applying constraints. With 82 WTPs, the data-to-parameter ratio was about 1.05, leaving the system close to underdetermined. In practice, this caused sensitivity: shadow prices fluctuated strongly with small data modifications (filters, shifts, exclusions). As mentioned in the methods section, a noisy (but unique) solution is found, but for an estimate that considers the noise as equally good points for the estimate.

By removing the variable that dominates the results, TOC, the result became more stable. The number of parameters to estimate in the optimization are then 66 (55 free) leading to a data-to-parameter ratio of 1.24. This renders a more stable result, even though not fully as stable as desired and left TOC un-estimated.

To retrieve the reference price for water we started from the destination-specific reference prices for treated wastewater reported by Hernández-Sancho et al. (2010) for Spain: 0.7 €/m³ (river), 0.1 €/m³ (sea), 0.9 €/m³ (wetlands) and 1.5 €/m³ (reuse), all in 2009 euros (Table 3 in their paper).

To express these in 2025 price level, we update them using the euro-area Harmonized Index of Consumer Prices (HICP). The cumulative inflation factor is found to be 1.4 and the exchange rate of 11 SEK/€ is used to calculate river 10.9 SEK/m³ and sea 1.6 SEK/m³.

Hydrologically and ecologically, the Baltic Sea is a semi-enclosed, brackish and eutrophication-sensitive sea with limited water exchange, much more vulnerable than typical open-ocean receivers. This is reflected in international regulation: under the IMO's MARPOL Convention, the Baltic Sea is explicitly designated a "special area" requiring stricter pollution controls for oil, sewage and other discharges due to its oceanographic and ecological conditions.

We assume the Baltic Sea has an environmental value as a receiving water that lies between river and open sea and defined its reference price as a convex combination of the two taken as 58% river-like and 42% sea-like obtaining a reference price of 7 SEK/m³.

Four MWTPs were further removed from the dataset when optimizing the shadow prices for metals. The shadow prices of undesirable outputs are shown in Table 4.

Table 4: Shadow prices of contaminants from treatment of ship-generated greywater

Receptacle	Shadow prices for pollutants (SEK/kg)								
	P	N	COD	BOD	Zn	Cu	Mn	PET	PP
Sea	-9.0531	-328.7708	-10.5485	-0.1013	-42655	-29067	-2760	241,5	241,5

Sensitivity in shadow price calculations: As mentioned and discussed in both the methodology section, and the results, the ratio data to estimated parameters is close to one (1.05) meaning the system is just about determined. But this is with the inclusion of datapoints that may still include noise through semi-implausible values, misreporting and such. The shadow prices are unstable, and slight variations in the data before optimization make the prices blow-up or -down. In some runs, with harsher outlier filtering for instance, optimization failed to converge to a (reasonable) solution, due to either the constraints not being met, or the calculated derivates had the wrong sign.

Smaller shifts in the data provided vastly different results, often leading to the SP of organic compounds dominating the environmental benefit. The results for trace pollutants (metals here) were affected by the other variables' grandness and thus rendered unimportant for the full benefit. In an ideal case, 5-10 data points per parameter (approximately 400 - 800 observations for our desired input-output space) would be needed. Excluding outliers improved the results, however extremely harsh removals rendered the system underdetermined.

3.3.2 - Environmental Benefit of modelled scenarios

The environmental benefits of the scenarios arise from two components: (i) avoided pollution damage ($EB_{pollution}$) and (ii) the environmental value of freshwater substitution (EB_{reuse}). Their sum yields the total environmental benefits.

3.3.2.1 - Environmental Benefit of Greywater treatment (Pollutant removal) ($EB_{pollution}$)

Appendix 3, Tables 1 and 2, summarize the environmental benefits from pollution removal through GW treatment ($EB_{pollution}$) across all greywater management scenarios. The results reflect the monetary value of avoided environmental damage obtained by applying pollutant-specific shadow prices to annual pollutant loads and respective removal efficiencies. Total $EB_{pollution}$ values ranges from 0 SEK/year under untreated sea discharge (SC1) to ≈ 1.22 million SEK/year under the 100% reuse scenario ($SC5B_1$ – 100% Reuse).

Environmental benefits increase sharply when shifting from untreated sea discharge (SC1) to any form of treatment. Under SC2, when Greywater is treated by an onboard AWTP before being discharged to the sea, $EB_{pollution}$ rises to 843,583 SEK/yr. This increase is driven largely by significant reductions in COD, suspended solids and heavy metals such as Cu and Zn, pollutants associated with high shadow prices and thus substantial avoided damage when removed. The magnitude of $EB_{pollution}$ under SC2 is comparable to the benefits reported for secondary wastewater treatment in coastal environments in Spain and Portugal, where Hernandez-Sancho et al. (2010) found that modest nutrient removal combined with high removal of organics and metals yield considerable environmental value.

A further increase is observed under SC3, where GW is pumped ashore to the municipality treatment plant. Here, $EB_{pollution}$ increases to ≈1.05 million SEK/yr, reflecting the higher nutrient removal capacities of the MWTP (82% for N and 95% for P). This pattern is consistent with shadow price applications in coastal regions, where nutrient reductions account for a large share of total environmental benefits due to their substantial ecological effects. Bellver-Domingo and Hernandez-Sancho (2018) similarly reported that wastewater discharges into sensitive basins generate disproportionately high external costs, and therefore nutrient removal creates strong marginal benefits, an observation particularly relevant for the eutrophication-prone Baltic Sea.

Environmental benefits peak among non-reuse scenarios under SC4, reaching ≈1.17 million SEK/yr. This scenario combines port-side advanced pre-treatment with final polishing at the municipal plant, resulting in the highest overall pollutant removals, especially for microplastics, suspended solids, COD, and trace metals. This incremental improvement aligns with the literature on tertiary and quaternary wastewater treatment trains, where Molinos-Senante et al. (2011) and similar studies found that augmenting conventional treatment with additional filtration or membrane-based systems consistently produces 10 – 20% increase in monetized environmental benefits.

SC5A, which involves PWTP treatment followed by direct discharge to sea, results in a lower $EB_{pollution}$ value of ≈ 872,821 SEK/yr. Although PWTP removes a considerable share of heavy metals and microplastics, the lack of nutrient polishing by the MWTP reduces total environmental benefits. This confirms findings from the Mediterranean and Atlantic literature showing that marine discharges, even when pre-treated, yield lower environmental benefits than scenarios routed through MWTP (Machado & Imberger, 2012; Luthy et al., 2015).

Reuse scenarios ($SC5B_1$ – $SC5B_5$), display a different pattern, because environmental benefits depend on the fraction of GW that undergoes formal treatment. The 100% reuse scenario, $SC5B_1$, yields the highest $EB_{pollution}$ (1.22 million SEK/yr), as all GW passes through PWTP before being recirculated. Although reuse eliminates discharge, the system design ensures that all GW is treated and reused, thus maximizing pollution removal and associated benefits. The results correspond with reuse studies from La Palma (Spain), where full treatment-plus-reuse pathways were shown to produce the highest environmental benefits due to complete removal of organics, nutrients and trace contaminants to reuse (Martinez-Lopez et al., 2020).

Table 5: Total Environmental Benefit from Pollution Removal ($EB_{pollution}$)

Scenario	$EB_{pollution}$ (SEK/year)	EB_{Reuse} (SEK/year)	EB_{Total} (SEK/year)
SC1	0	0	0
SC2	845,878	0	845,878
SC3	1,059,862	0	1,059,862
SC4	1,175,638	0	1,175,638
SC5A	876,153	0	876,153
$SC5B_1$	1,226,279	321,488	1,547,767
$SC5B_2$	881,729	241,116	1,122,845
$SC5B_3$	587,819	160,744	748,563
$SC5B_4$	657,114	241,116	898,231
$SC5B_5$	438,076	160,744	598,820

Partial-reuse scenarios demonstrate predictable liner declines in $EP_{pollution}$. When only 75% of GW is reused and 25% sent to MWTP ($SC5B_2$), environmental benefits fall proportionally, reaching ≈882,000 SEK/yr. Similarly, 50% reuse and 50% sent to MWTP ($SC5B_3$) results in ≈ 588,000 SEK/yr. These values mirror results from Mediterranean reuse studies, where reduced treatment volumes correspond directly to proportional decreases in monetized environmental benefits. SC5A variants that discharge the non-reused fraction directly to sea (instead of MWTP) yield even lower benefits, as expected, because municipal tertiary polishing is absent. This again confirms the central role of nutrient removal in determining $EB_{pollution}$.

In all scenarios, heavy metals (particularly Cu and Zn), nitrogen and COD account for the largest share of total environmental benefit due to their high annual loads and extremely high shadow prices (-29,067 SEK/kg for Cu and -42,655 SEK/kg for Zn). Nutrients (N and P) also contribute significantly, especially in scenarios routed to the MWTP. These pollutant-specific patterns closely align with findings from shadow-price studies in Spain, Italy and Portugal, where metals, COD and nutrients consistently dominate monetized benefits. Overall, the environmental benefit results demonstrate that:

i) routing GW through high-efficiency treatment (PWTP + MWTP) maximizes environmental value;
ii) reuse scenarios achieve similar or higher environmental benefits when treatment remains comprehensive; and
iii) pollutants with high shadow prices (metals, COD, nutrients) drive most of the observed benefits.

These findings reinforce the broader literature, (Angelakis & Gikas, 2014; Chamaki et al., 2022) which highlight that advanced treatment and reuse in sensitive marine environments produce substantial societal environmental gains and are economically favourable when evaluated using shadow-price methods, especially in region like the Baltic Sea where nutrient sensitivity is high and marginal damage costs are substantial. Moreover, reuse becomes substantially more valuable when pollution abatement is combined with circular water management.

3.3.2.2 - Environmental Benefit of Reuse (EB_{Reuse})

In addition to pollution-removal benefits, the reuse scenarios generate a notable environmental benefit associated with freshwater substitution (EB_{reuse}). This benefit captures the environmental value of avoiding the abstraction and supply of freshwater that would otherwise be required to meet shipboard demands. The magnitude of EB_{reuse} depends directly on the volume of GW that is reused. As shown in Table 13, in the full-reuse scenario ($SC5B_1$), EB_{reuse} reaches 321,488 SEK/year, contributing substantially to the total environmental benefit of ≈ 1.54 million SEK/yr. This magnitude reflects the fact that all greywater is diverted to onboard reuse, thereby eliminating the need for an equivalent quantity of externally supplied potable water. Partial reuse scenarios exhibit proportional decreases in EB_{reuse}: 75% reuse generates 241,116 SEK/yr, while 50% reuse yields 160,744 SEK/yr. This proportional relationship indicates that EB_{reuse} scales linearly with the reuse fraction, consistent with the theoretical structure of the shadow-price method and with empirical observations from Mediterranean Island systems such as Las Palmas and Cyprus, where freshwater substitution is found to correlate strongly with reuse volume.

These EB_{reuse} values, though significantly smaller than the avoided potable-water production costs in earlier versions of the model, nevertheless represent a meaningful ecological benefit. They indicate that freshwater substitution alone can raise total environmental benefits by 30 – 50% compared with treatment-only scenarios, depending on the reuse fraction and discharge pathway. For instance, $SC5B_2$ 75% reused with discharge to MWTP yields a total environmental benefit of ≈ 1.12 million SEK/yr, substantially higher than the comparable non-reuse scenario SC3 (≈ 1.06 million SEK/yr). Likewise, $SC5B_4$ 75% reuse with discharge to sea yields ≈898,000 SEK/yr, higher than SC5A (≈ 876,000 SEK/yr). These improvements are driven solely by the reuse-related environmental benefit, as removal efficiencies of the underlying treatment trains remain constant. This pattern aligns with literature from Spain (Hernandez-Sancho et al., 2010; Molinos-Senante et al., 2011) and the Canary Islands, which consistently demonstrates that integrating reuse into wastewater systems produces environmental gains beyond pollutant removal alone, even in regions that do not experience acute water scarcity.

Overall, the EB_{reuse} component strengthens the environmental case for GW reuse by expanding the total benefits beyond pollution reduction. Although the absolute values are smaller than in earlier cost-based formulations of reuse benefit, the environmental shadow price still captures the ecological relevance of conserving freshwater resources, reducing strain on local water systems, and advancing circular-water practices. The results confirm that reuse not only mitigates pollutant emissions but also substantially enhances the environmental performance of maritime wastewater systems. Consequently, incorporating EB_{reuse} into the benefit framework provides a thorough and more accurate appraisal of the environmental value of reuse strategies in the Baltic Sea region.

3.4 - Net Societal Benefits of ship-generated Greywater Management

The calculated Net Societal Benefit (NSB) shows a striking divergence between scenarios without greywater reuse and those incorporating reuse. The five non-reuse scenarios (SC1 – SC5A) values, ranging from -1.09 million SEK/yr (SC1) to -2.37 million SEK/yr (SC4). These negative outcomes arise because the environmental benefits associated with pollution removal, although significant, up to ≈ 1.17 million SEK/yr in the most advanced treatment cases, are not large enough to compensate for the total societal costs of implementing and operating the GW treatment systems. All scenarios produce negative NSB under current shadow-price valuations. This is consistent with international literature showing that wastewater treatment alone does not generate positive net welfare unless water scarcity is high or reuse values are significant (Molinos-Senante, 2011; Berbel et al., 2023).

Table 6: Net Societal Benefits (SEK/yr)

Scenario	C_{Soc}	EB_{Total}	NSB
SC1	1,093,243	0	-1,093,243
SC2	2,775,155	845,878	-1,931,572
SC3	2,717,200	1,059,862	-1,661,980
SC4	3,537,486	1,175,638	-2,367,377
SC5A	2,805,146	876,153	-1,932,325
SC5B$_1$	3,007,786	1,547,767	-1,466,425
SC5B$_2$	2,979,467	1,122,845	-1,860,768
SC5B$_3$	2,686,297	748,563	-1,940,498
SC5B$_4$	2,796,381	898,231	-1,900,649
SC5B$_5$	2,584,977	598,820	-1,987,823

SC1, although ranked first (Fig. 8A), is the "least negative" only because it represents the baseline scenario with no capital or operational costs. However, this does not reflect environmental or resource-recovery performance and should not be interpreted as the best overall option. Removing SC1 from the ranking (Fig. 8B), the ranking becomes policy relevant. The results show clear dominance of reuse-oriented scenarios: SC5B$_1$ (100% reuse) achieves highest NSB, followed by SC5B$_2$, SCB$_4$, SCB$_3$ and SCB$_5$, with the proportion of water reused. The destination of the non-reused fraction has little influence on NSB. This demonstrates that reuse volume is the primary driver of societal benefit. The ordering among SC5B$_2$ – SC5B$_5$ shows that the destination of the non-reuse fraction has only a minor influence on NSB. This is because the freshwater-substitution benefit dominates the benefit structure, overwhelming differences in treatment or discharge routes for the residual fraction.

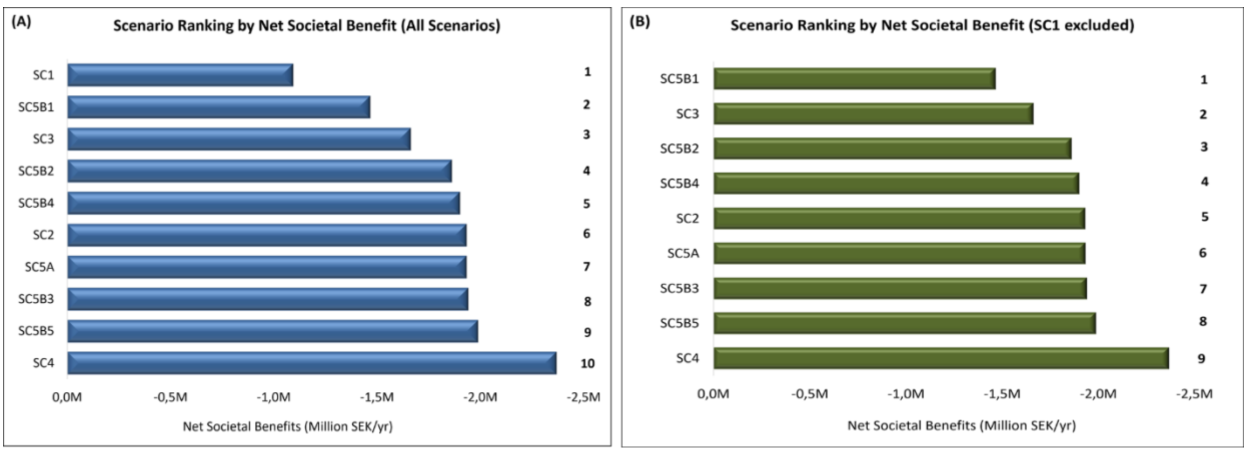


***Figure 8**: A) Scenario ranks by NSB (All scenarios). A) Scenario ranks by NSB (excluding SC1).*

SC3 performs better than the other non-reuse scenarios. This is because it involves only one treatment system and therefore has a lower capital and operating burden compared with SC2 and SC4. SC" ranks below SC3 because it introduces additional treatment complexity and costs without generating reuse-related benefits. The incremental environmental performance is therefore insufficient to compensate for the added financial burden. On the opposite extreme, SC4 performs worst. Its very low NSB reflects the dual burden of installing both PWTP and MWTP without sufficient offsetting benefits, leading to the most negative NSB, hence economically unattractive.

In summary, reuse substantially reduces societal welfare losses but does not fully offset treatment costs in the Baltic Sea context. Moreover, treatment alone does not deliver societal value unless it is linked to reuse, therefore adding more treatment stages without reuse rapidly degrades NSB. The results confirm that reuse is not merely beneficial, but essential for achieving positive or near-neutral societal outcomes.

It is worth noting that, in the current modelling, the non-reuse fraction in the partial-reuse scenarios ($SC5B_2$ – SCB_5) is treated as if it only passes through the "standard" PWTP and/or MWTP, with the same removal efficiencies that apply to untreated wastewater. In practice, the GW that is not reused would already have passed through the enhanced treatment train designed to produce reuse-quality water (e.g. additional filtration, adsorption, membranes etc…). This means that the pollutant concentration of the non-reused fraction is likely to be lower than assumed in the model. By updating the removal efficiencies to reflect this extra polishing step, the analysis effectively assumes that the discharged fraction is "dirtier" than it would be. Consequently, the environmental benefit from pollution removal ($EB_{pollution}$) in $SC5B_2$ – $SC5B_5$ is underestimated, because we are attributing a higher residual pollutant load to the sea or MWTP than the enhanced treatment line would produce. This simplification mainly biases the results against the partial-reuse options. In a more detailed mass-balance representation, all GW entering the reuse system would first experience a higher overall removal efficiency; only then would a share be reused and the remainder discharged either to the sea or to MWTP. $EB_{pollution}$ per cubic meter would therefore be higher than in the present model for all partial reuse variants, and the difference in $EB_{pollution}$ between "to sea" and "to MWTP" options might also be somewhat smaller. In other words, the current approach treats the reuse system almost exclusively as a source of water-saving benefits (EB_{reuse} and EB_{water}), while under-recognizing its contribution to improved effluent quality for the non-reused fraction. This is due to lack of data.

At the same time, it is important to note that in the current results the net societal benefits of the reuse scenarios are dominated by the avoided potable-water production term, which is several orders of magnitude larger than $EB_{pollution}$. This means that, although $EB_{pollution}$ for $SC5B_2$ – $SC5B_5$ is likely underestimated, the overall ranking of scenarios, particularly the conclusion that reuse scenarios are socially preferable to non-reuse scenarios, is unlikely to change. The limitation therefore concerns the magnitude and relative differences between partial-reuse options rather than the main qualitative conclusions of the study.

Future work should, therefore, replace the simplified, constant removal efficiencies with a more detailed mass-balance model of the treatment train, in which (i) the enhanced reuse treatment step is explicitly represented, (ii) separate effluent streams (reused water and discharged water) are characterized with their own pollutant concentrations, and (iii) scenario-specific removal efficiencies are calibrated using pilot or full-scale performance data. Such an approach would allow a more accurate estimation of $EB_{pollution}$ for partial-reuse scenarios and could be combined with uncertainty or sensitivity analysis around key performance parameters.

3.5 - Integrated Assessment of Scenario Performance and Ranking of scenarios

Taken together, the societal cost, stakeholder cost, environmental benefit, and NSB results paint a coherent picture of greywater management for Baltic passenger ships. The ten scenarios represent a spectrum from "do almost nothing" (SC1) to "dual advanced treatment plus reuse" (SC4 and $SC5B_1$), with multiple intermediate combinations that redistribute treatment effort across ship, port, and municipality.

At the societal level, SC1, direct discharge to sea, is the cheapest configuration, with a total real-resource cost of about 1.09 million SEK/yr. This reflects the absence of any treatment beyond minimal storage and the fact that potable-water production is the only significant cost driver. However, SC1 delivers no environmental benefit at all and is incompatible with the long-term environmental ambitions of the Baltic Sea Action Plan and MARPOL Annex IV norms. It represents a low-cost environmentally poor baseline.

Once treatment is introduced, societal costs rise substantially. All non-reuse treatment options (SC2 – SC5A) cost between about 2.7 and 3.5 million SEK/yr, with SC4 clearly the most resource-intensive configuration. Within this group, shifting treatment responsibility between shipboard units (SC2), port reception plus MWTP (SC3), or PWTP-only (SC5A) alters the distribution of costs but does not fundamentally change their magnitude.

Reuse changes the cost structure but not in a prohibitive way. The full-reuse scenario $SC5B_1$ carries a societal cost of 3.03 million SEK/yr, higher than some single-stage treatments but lower than the dual PWTP + MWTP configuration. Partial reuse with MWTP disposal ($SC5B_2$ and $SC5B_3$) is particularly interesting: these scenarios reduce total cost relative to SC4 and in the case of $SC5B_3$ even approach the lower bound of treatment-based configurations. This is because they take advantage of both reuse-capable PWTP polishing and cost-effective MWTP treatment. Scenarios where the non-reuse fraction is discharged to sea ($SC5B_4$, $SC5B_5$) lose some of this efficiency and sit closer to the mid-range of societal costs.

However, what looks efficient from a system perspective can be deeply unattractive for individual actors. The stakeholder cost analysis shows that ship operators incur the largest financial burden in all scenarios whether or not reuse is involved. Their net annual costs jump from -1.09 million SEK/yr in the direct-discharge case to roughly -2.8 million SEK/yr in all treatment and reuse scenarios. This burden arises from potable-water purchases, storage and port-reception and treatment charges. Critically, ships do not capture any of the environmental or reuse-related benefits monetized in the societal cost analysis. This asymmetry reproduces a pattern reported in Las Palmas and Baltic Annex IV evaluations, where ships are expected to pay for environmental improvements whose benefits accrue mainly to society and coastal waters, not to the vessels themselves.

The port's financial position is mixed and fragile. Under non-reuse configurations, the port either breaks even (SC1, SC2, SC5A) or experiences moderate losses when MWTP fees are involved (SC3, SC4). Reuse scenarios improve the picture slightly: $SC5B_1$ and $SC5B_4$ yield modest net gains for the port (on the order of tens of thousands of SEK), but other reuse configurations still produce moderate losses. This reflects the reality that operating reuse-capable PWTP systems is expensive, and current tariff structures do not enable ports to fully recover these costs from ship fees or water savings. In effect, ports sit on an uncomfortable middle position: they are expected to enable circular water management, but the business case for investing in advanced treatment remains weak unless external funding or new tariff models are introduced.

Municipalities, by contrast, are deliberately neutral in this model, MWTP tariffs are assumed to reflect full cost, so municipal utilities neither profit from nor subsidize additional greywater loads from ports. This mirrors standard Nordic and EU cost-recovery principles, but it also means that municipalities are not economic drivers of change in this system, they are passive service providers.

Total Environmental Benefits strongly favour treatment and particularly reuse. Pollution-removal benefits alone ($EB_{pollution}$) range from nearly zero in SC1 to around 1.17 million in SC4, with SC3 not far behind. These values are dominated by the removal of COD, nitrogen, and zinc, reflecting both the relatively high mass loadings of these pollutants and their corresponding high shadow prices. This pattern is consistent with empirical studies in Spain and Portugal, which have repeatedly highlighted COD and N, as dominant contributors to avoided environmental damage in wastewater systems (Hernandez-Sancho et al., 2010).

Reuse adds a second layer of environmental benefit, EB_{Reuse}, by assigning value to freshwater substitution. In the Baltic context, where water scarcity is moderate and the freshwater shadow price is not extreme, EB_{Reuse} is not enormous in absolute terms; however, it still significantly increases total environmental benefit. For example, $SC5B_1$'s EB_{Total} of 1.54 million SEK/yr is considerably larger than any non-reuse option. Partial reuse scenarios produce proportional benefits, with the 75% and 50% reuse cases delivering roughly three-quarters and half the EB_{Reuse}, respectively. The destination of the non-reused fraction matters somewhat for pollution removal (MWTP being superior to sea discharge), but far less than the reused fraction itself.

These environmental benefits, however, are not large enough to fully offset treatment costs in any scenario under the current shadow price framework. When environmental benefits are netted against societal costs, all simply because it has negligible cost and zero benefit, while SC4 has the most negative NSB due to its very high-cost relative to its environmental benefits. Among all treatment and reuse configurations, $SC5B_1$ (100% reuse) stands out as the socially "least bad" scenario: its NSB is around -1.47 million SEK/yr, significantly better than SC2, SC3, SC4, or SC5A. Partial reuse scenarios fall in between, with NSB values that deteriorate as the reuse fraction shrinks.

The integrated picture is therefore one of multiple misalignments. From a narrow cost perspective, SC1 is the most attractive but environmentally unacceptable. From a societal welfare perspective, $SC5B_1$ and $SC5B_2$ are clearly superior to SC2 – SC5A, even though NSB remains negative. From a stakeholder perspective, ships strongly prefer SC1 and have no financial reason to support any treatment or reuse pathway. Ports are at best marginal winners in certain reuse configurations and often face net losses in others, Municipalities are indifferent. From an environmental perspective, scenarios that integrate treatment and reuse, particularly with MWTP as a polishing step, are clearly most desirable, especially when Baltic eutrophication and microplastic accumulation are considered.

These tensions between societal efficiency private incentives, and environmental outcomes reflect a broader pattern described in the international literature: wastewater reuse and advanced treatment rarely emerge spontaneously in the absence of strong policy direction, economic instruments, or externally financed infrastructure. In water-scarce regions, high potable-water values sometimes yield positive NSB and create a natural economic drive towards reuse. In the Baltic Sea region, where water is not scarce, but the marine environment is highly vulnerable, the impetus must come from regulatory obligations, environmental targets, and deliberate cost-sharing mechanisms, rather than from direct profitability.

3.6 - Metal to Phosphorus Ratios and their effect on Sludge Quality

The data presented in Tables 6 and 7 illustrate metal-to-phosphorus (Me/P) ratios for greywater entering the Trelleborg port wastewater treatment plant (TWTP) and the municipal wastewater treatment plant (MWTP). These ratios provide an effective indicator of sludge quality in relation to potential agricultural use, since phosphorus is the main nutrient of agronomic value while metals represent the limiting contaminants for land application. Expressing metal concentrations as Me/P (g metal per kg P) normalizes the data to the fertilizing component of the sludge and facilitates direct comparison with established quality criteria such as those of the Swedish Renare Vatten med Avloppsslam som Kretslopp (Revaq) certification system.

At the $MWTP_{Trell}$, Me/P ratios for zinc (26–30 g/kg P), copper (5.6–5.7 g/kg P), lead (0.09 g/kg P), cadmium (0.006–0.007 g/kg P), nickel (0.34–0.65 g/kg P) and chromium (0.25–0.27 g/kg P) indicate that most metals are efficiently retained in the sludge relative to phosphorus. The corresponding ratio for cadmium, approximately 6–7 mg Cd per kg P, aligns well with the Revaq long-term target of maintaining Cd/P below 7 mg/kg P, which represents the average cadmium content of mineral phosphorus fertilizers in Sweden. The relatively low Cd/P at the TWTP suggests that the port-generated greywater does not introduce a disproportionate cadmium load to the sludge and could therefore constitute a "clean P" source in terms of heavy metal contamination.

In contrast, the MWTP data show higher Me/P ratios for several elements, particularly cadmium (0.016 g/kg P, equivalent to 16 mg Cd/kg P) and copper (11 g/kg P). These levels exceed the Revaq benchmark and imply that, if phosphorus in the municipal sludge were applied to arable land, it would contribute more cadmium per unit of plant nutrient than mineral fertilizers. Elevated Cd/P and Cu/P ratios typically reflect diffuse and industrial urban inputs, corrosion of household plumbing, and other anthropogenic sources within the municipal catchment. The nickel ratio (0.14 g/kg P) is somewhat lower than at TWTP, while chromium and lead are comparable. Overall, the port-derived stream is characterized by lower metal contamination per unit of phosphorus than the mixed municipal influent.

From a sludge management perspective, these findings have several implications. First, sludge or biosolids derived from the TWTP would be more suitable for agricultural recycling under current Swedish quality objectives. If blended with municipal sludge, the TWTP material could help reduce the overall Cd/P of the combined product to within the Revaq target range. Second, the MWTP results highlight the continued need for upstream measures to reduce metal inputs, especially cadmium and copper, through industrial pretreatment, substitution of corrosion-prone materials, and public awareness programs. Such actions are essential to ensure long-term compliance with both the national limits for metals in sewage sludge (SNFS 1994:2) and the stricter voluntary criteria for Revaq-certified plants.

Although the Me/P ratios provide a useful normalization, compliance with legal thresholds expressed as total metal concentrations (mg/kg dry solids) and annual loading rates per hectare remains obligatory. Nevertheless, the normalization to phosphorus clarifies the relative quality of different sludge streams in a nutrient recycling context. The comparatively favorable Cd/P ratio of the TWTP sludge demonstrates that port-received wastewater, when properly managed, can contribute to phosphorus of acceptable quality for land application and may even improve the overall metal-to-phosphorus balance of municipal sludge destined for agricultural use.

Table 7: Comparison of Metal to phosphorus ratios in sewage sludge

Contaminant	Unit	Inlet TWTP	Sludge	Outlet TWTP	MWTP
Zn /P	mg/g	26	30	17	21
Cu /P	mg/g	5.6	5.7	5.3	11
Pb /P	mg/g	0.09	0.09	0.09	0.29
Cd /P	mg/g	0.006	0.007	0.005	0.016
Ni /P	mg/g	0.43	0.34	0.65	0.14
Cr /P	mg/g	0.25	0.27	0.17	0.23

The Me/P ratios observed at the Trelleborg port wastewater treatment plant (TWTP) before and after treatment suggest that pretreatment may not substantially improve the overall metal quality of the sludge from a phosphorus recycling perspective. Since the Me/P ratio expresses the concentration of metals relative to phosphorus, a reduction in both elements during treatment can yield little or no change in the ratio. In several cases, the Me/P values after treatment were similar to or even higher than those of the influent, indicating that phosphorus removal occurred to a similar or greater extent than metal removal. Consequently, while pretreatment at TWTP effectively lowers the total load of both P and metals entering the municipal system, it does not necessarily enhance the relative quality of the sludge (i.e., metals per unit of P).

From a sludge-handling standpoint, this means that the net benefit for the municipality—measured as improved sludge quality for agricultural use, is limited if pretreatment primarily reduces phosphorus together with metals. The municipality would receive a smaller phosphorus input but with roughly the same or slightly poorer metal-to-phosphorus balance. The primary advantage of TWTP pretreatment would therefore lie in reducing total loading and hydraulic stress on the municipal plant, rather than in improving Me/P-based sludge quality. Unless pretreatment selectively removes metals more efficiently than phosphorus, its contribution to achieving lower Cd/P or Cu/P ratios in the final biosolids will remain marginal.

3.7 - Sensitivity Analysis

3.7.1 - Sensitivity of scenario ranking to pollutant concentration

Using empirically derived concentration data for ship-generated greywater from a recent characterization study in the Baltic Sea (Mujingni et al., 2024) the cost–benefit analysis (CBA) shows that the ranking of management options is highly sensitive to the underlying water quality assumptions. When the average measured concentrations of metals, nutrients and organic matter are used as input to the damage-cost functions, the scenario with direct discharge of greywater to the sea emerges as the most beneficial option in Net Present Value terms. This result is driven by the combination of (i) relatively low mean concentrations in the greywater, implying modest marginal environmental damage per cubic meter discharged, and (ii) the relatively high investment and operational costs associated with on-board treatment or shore-based upgrading to enable reuse. Under these average conditions, the additional environmental benefits of treating or reusing greywater do not outweigh the added costs of the more advanced management options.

However, when the analysis is repeated using the maximum concentrations observed in the same greywater data set, representing a conservative high-contamination case, the CBA results change markedly. Under this parameterization, scenarios involving reuse of treated greywater (for example, for non-potable purposes on board or after discharge to shore-based systems) generate the highest net societal benefits. In this case, the substantially higher pollutant loads per unit volume translate into significantly larger avoided damages to the marine environment when discharges are reduced, while the treatment and reuse costs remain unchanged. As a result, the net benefit of reuse scenarios surpasses that of direct discharge, and the ranking of options is effectively changed.

Taken together, these findings highlight that optimal greywater management is highly sensitive to assumptions about water quality and associated environmental damages. They underline the importance of (i) representing the full variability of contaminant concentrations in economic assessments, and (ii) complementing point estimates with sensitivity analyses or scenario ranges. From a policy perspective, the results suggest that for ships or routes where greywater concentrations are systematically closer to the upper end of the observed range, regulation or incentives favoring treatment and reuse are likely to be welfare-enhancing, whereas for traffic segments with consistently low concentrations, stricter requirements may yield only limited additional net benefits.

3.7.2 - Sensitivity of NSB to Freshwater Shadow Price (SP_{water})

Fresh water substitution is the dominant benefit component in all reuse scenarios, because each cubic meter of reused greywater avoids the production, distribution and purchase of an equivalent volume of potable water. To examine how strongly the reuse results depend on this valuation, the shadow price of freshwater, SP_{water}, was varied multiplicatively around the base case. Five values were tested: 0.0x, 0.5x, 1.0x, 1.5x and 2.0 x SP_{water}, while holding all other parameters (treatment costs, pollution-removal benefits, volumes) constant. For each value, the reuse benefit term EB_{reuse} and the resulting NSB were recalculated for SCB_1 – SCB_5.

Table 18 presents the resulting NSB values across the range of SP_{water} multipliers. Under the assumption that freshwater has no economic value (0.0x SP_{water}), all reuse scenarios perform poorly, with NSB values between -2.10 and -2.38 million SEK/year. In this case, the additional treatment cost of producing reuse-quality greywater is not compensated by pollution-removal benefits alone. At the base valuation (1.0x SP_{water}), $SC5B_1$ (100% reuse) performs best, with NSB ≈ -1.47 million SEK/year, substantially better than all non-reuse scenarios and then the other reuse variants ($SC5B_2$ – $SC5B_5$). Doubling SP_{water} (2.0x) reduces the welfare loss for $SC5B_1$ to ≈ -1.14 million SEK/year, representing a significant improvement relative to the base case.

Partial-reuse scenarios also improve as SP_{water} rises, but the improvement is less pronounced because they substitute smaller volumes of potable water. Their NSB trajectories remain roughly parallel to that of $SC5B_1$ and do not overtake it at any tested freshwater value. This pattern confirms that the economic attractiveness of reuse is highly sensitive to the assumed value of freshwater and that high reuse fractions consistently produce better outcomes than partial-reuse designs.

Table 8: Sensitivity of NSB (SEK/year) to SP_{water} for Reuse Scenarios.

SP_{water} multipliers	$SC5B_1$	$SC5B_2$	$SC5B_3$	$SC5B_4$	$SC5B_5$
0,0	-1790000	-2100000	-2100000	-2140000	-2150000
0,5	-1630000	-1980000	-2020000	-2020000	-2070000
1,0	-1470000	-1860000	-1940000	-1900000	-1990000
1,5	-1310000	-1740000	-1860000	-1780000	-1910000
2,0	-1140000	-1620000	-1780000	-1660000	-1830000

Although all reuse scenarios remain negative in NSB under current Swedish conditions, the sensitivity analysis demonstrates that higher freshwater values, such as in regions with water scarcity or higher energy and environmental costs of drinking-water production, would improve the net societal performance of reuse and may render reuse scenarios net-positive in other contexts. Across all tested conditions, $SC5B_1$ remains the best-performing reuse scenario, and all reuse configurations outperform non-reuse treatment options in relative terms. A line plot of these values (Figure 9A), with SP_{water} on the x-axis and NSB on the y-axis, reveals a clear nearly linear improvement in NSB as water value increases. $SC5B_1$ always forms the upper envelope of the scenario bundle, demonstrating the strong economic advantage of maximizing reuse.

3.7.3 - Sensitivity of NSB to Reuse Treatment Cost (C_{Reuse} ±20%)

A second sensitivity test examined uncertainty in the cost of producing reuse-quality greywater (C_{reuse}) at the PWTP. This cost reflects the incremental expense of enhanced treatment steps, such as filtration, adsorption, or disinfection, required to meet reuse standards.

Table 9: NSB Sensitivity to the cost of producing reuse-quality greywater ±20% (SEK/year)

Scenarios	NSB_{Low} (-20%)	NSB_{base}	NSB_{high} (+20%)
$SC5B_1$	-1425897	-1 466 425	-1506953
$SC5B_2$	-1825904	-1 860 768	-1895632
$SC5B_3$	-1964268	-1 940 498	-1916728
$SC5B_4$	-1902402	-1 900 649	-1898897
$SC5B_5$	-2031856	-1 987 823	-1943789

Because real-world treatment costs vary with energy prices, chemical dosing rates, membrane replacement schedules, and maintenance needs, C_{reuse} was altered by ±20% around its base value. NSB was recalculated for $SC5B_1$ – $SC5B_5$ under three cases: (1) C_{reuse} -20% (optimistic), (2) C_{reuse} (base), (3) C_{reuse} +20% (pessimistic)

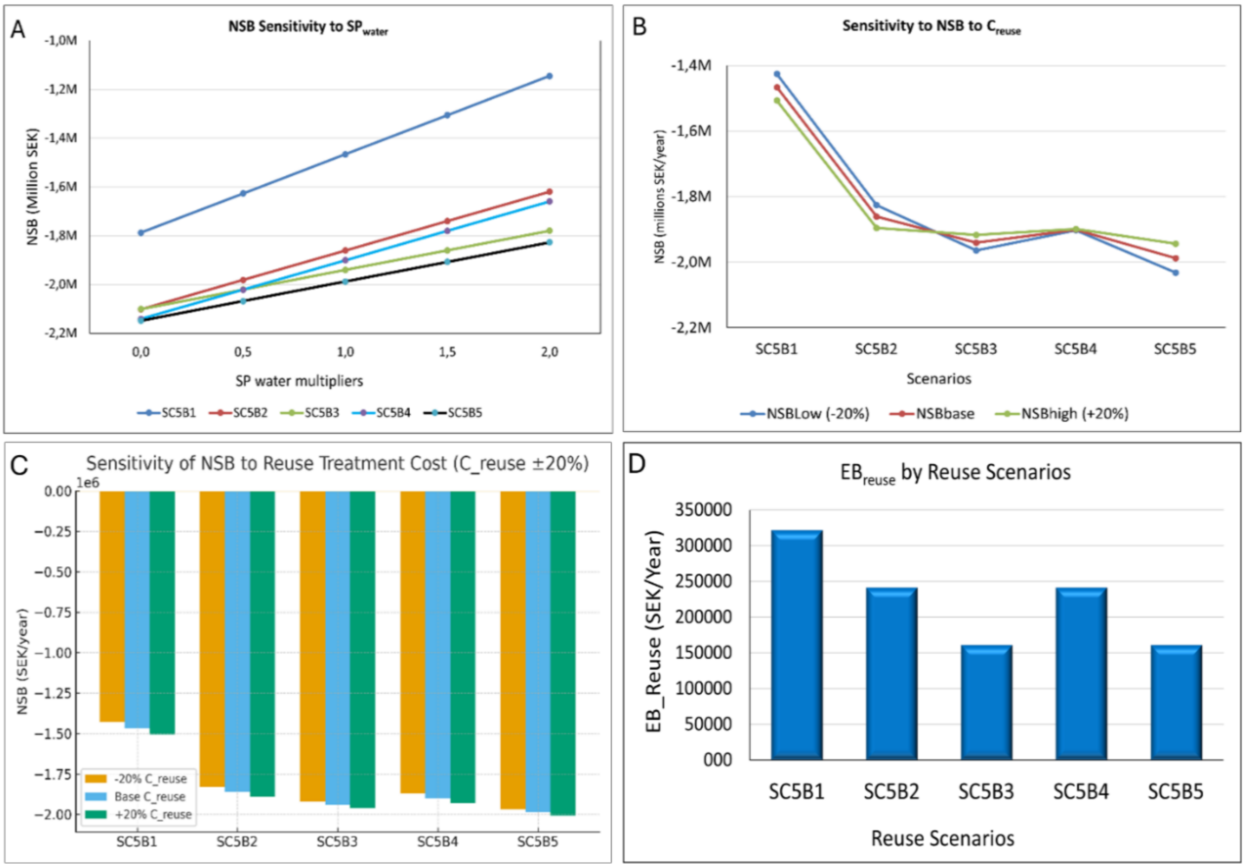


Figure 9: A) *Line plot of NSB Sensitivity to the shadow price of freshwater*, B) *Line plot of NSB Sensitivity to Reuse treatment cost, C) Sensitivity of NSB to reuse treatment cost, D) Environmental Benefit of reusing treated greywater*

In all reuse scenarios, the annual reuse-treatment cost is relatively small compared with total societal cost (typically 0.10 – 0.20 million SEK/year). Consequently a ±20% change in C_{reuse} modifies NSB by only ±20,000 to ±40,000 SEK/year, an adjustment that is negligible relative to the overall magnitude of NSB (-1.14 to -2.15 million SEK/year). As a result, the ranking of scenarios remains unaffected: $SC5B_1$ consistently performs best, followed by $SC5B_2$ and SCB_4, with $SC5B_5$ remaining the lowest-performing reuse options (Figures 9B and 9C). These results indicate that reasonable uncertainty in treatment costs has only a modest impact on economic outcomes, especially when compared to the very large influence of SP_{water}. The overall conclusions of the CBA are therefore robust to variations in C_{reuse}: the economic case for greywater reuse depends overwhelmingly on the value assigned to freshwater substitution, not the precise cost of reuse treatment. We can therefore conclude as follows:

Greywater treatment alone is not enough: Even the most advanced non-reuse treatment configuration (SC4) generates negative NSB: environmental benefits are simply too small, at current shadow prices, to cover the most of dual-stage treatment. This mirrors Mediterranean and Atlantic shadow-price studies, which report that tertiary and quaternary wastewater treatment seldom produce positive net benefits without reuse or energy recovery.

Reuse significantly improves societal performance: All reuse scenarios ($SC5B_1$ – $SC5B_5$) improve NSB relative to non-reuse treatment options, and 100% reuse ($SC5B_1$) performs best overall, with the smallest net welfare loss. This is consistent with international evidence that freshwater substitution is the dominant economic driver in reuse projects, even when water scarcity is moderate.

Partial reuse with MWTP polishing offers favourable trade-offs: $SC5B_2$ and $SC5B_3$ combine reuse with municipal polishing and achieve relatively low societal costs while reuse is not technically or operationally feasible.

Stakeholder incentives are severely misaligned: Ships face substantial cost increases under every treatment and reuse scenario, while ports see small and sometimes negative financial outcomes. Municipalities are indifferent. This misalignment ensures that, absent regulatory pressure or financial support, the actors responsible for implementation are unlikely to choose the socially and environmentally preferred scenarios.

Baltic Sea policy goals require economic instruments: The HELCOM BSAP and EU water policies demand reductions in nutrient and pollutant pressures, and greywater management can contribute to this. However, achieving the desired environmental outcomes will require more than technical standards; it will require targeted economic mechanisms to shift stakeholder choices.

4 - Policy and Industry implications

The findings of this study reveal deep structural barriers to the adoption of sustainable GW management solutions in the Baltic Sea shipping sector. Although the treatment and reuse scenarios deliver important environmental benefits and improve net societal outcomes, they impose disproportionate financial costs on ship operators and, in many cases, offer limited or negative financial returns for ports. At the same time, the regulatory environment, most notably MARPOL Annex IV's exclusion of greywater, provides no compliance incentive for vessels to treat or deliver GW to ports. These factors collectively hinder voluntary progress and highlight the need for coordinated policy action of regional environmental objectives.

4.1 - How results inform HELCOM BSAP and EU MSFD goals

The HELCOM Baltic Sea Action Plan (BSAP) and the EU Marine Strategy Framework Directive (MSFD) mandate reductions in nutrient inputs, hazardous substance, and marine litter. Greywater from large passenger vessels contributes organic matter, nutrients (N and P), metals (Cu, Zn, Ni), surfactants, pharmaceuticals, and microplastics, pollutants directly linked to eutrophication, toxicity, and ecological degradation highlighted in both BSAP and MSFD assessment.

The environmental benefits analysis in this study demonstrates that treatment and especially reuse scenarios provide measurable reductions in these pollutant loads. Reuse scenarios generate the highest $EB_{Pollution}$ values and eliminate large volumes of direct discharge, thereby supporting BSAP objectives on nutrient reduction and the MSFD's descriptors for eutrophication (D5), contaminants (D8), and marine litter (D10).

However, the societal and stakeholder cost analyses show that the scenarios most aligned with HELCOM and MSFD goal, advanced treatment and reuse, are precisely those that impose significant financial burdens on ships, with no mechanism under current regulation to redistribute costs in alignment with societal or environmental benefits. This creates tangible implementation barrier. Without policy intervention, the environmental objectives of HELCOM BSAP and MSFD cannot be met through voluntary adoption of advanced GW management systems.

4.2 - Policy instruments: voluntary adoption, subsidies, regulation, port incentives

The study shows that environmental stewardship alone is insufficient to change industry behaviour at scale. Three types of policy instruments emerge as essential for overcoming the misalignment between private incentives and societal/environmental goals:

Voluntary adoption mechanisms: Voluntary schemes, such as green-port certification, eco-labels for passenger vessels, or operator-led sustainability strategies, may encourage limited uptake among environmentally proactive shipowners. However, the results show that these measures will not be enough on their own, because the financial burdens on ships are too large to justify participation without cost relief. Voluntary tools may complement but cannot replace economic or regulatory measures.

Subsides and co-financing: Subsidies, public co-financing, or EU funding (e.g., CEF Transport, LIFE, Interreg, Horizon Europe) can significantly reduce the economic barriers to advanced port-based treatment and reuse. The societal analysis demonstrates that reuse scenarios produce net benefits for the region despite negative NSB values; thus, public support is justified on efficiency grounds when widely distributed environmental benefits are at stake. Subsidies can also reduce port losses under partial-reuse and dual-treatment scenarios.

Regulatory measures: Given that GW is unregulated under MARPOL Annex IV, the Baltic Region faces a governance gap. Regulatory measures, either region-specific (e.g. HELCOM agreements akin to sewage requirements for passenger ships) or EU-wide amendments, may be necessary to mandate a minimum level of GW treatment or port delivery. Regulation is the only mechanism capable of shifting all ships away from SC1 to treatment-based scenarios. This aligns with conclusions from Mediterranean and Atlantic wastewater-policy evaluations, where regulation was decisive for adoption of tertiary treatment and reuse.

Port incentives and tariff reform: Differentiated port fees are an effective tool for internalizing environmental benefits. Reduced fees for GW delivery, rebates for ships participating in reuse systems, and penalties for direct discharge (where permitted) can shift decisions. Ports may also recover a portion of treatment costs through sale or internal reuse of reclaimed water. However, tariff reform requires coordination between ports, municipalities, and shipping lines to avoid competitive distortions.

Overall, a combination of voluntary, economic, and regulatory instruments will be needed to bring ship and port incentives into alignment with societal and environmental goals.

4.3 - Wider applicability to other sea regions

Although this study focuses on the Baltic Sea, a semi-enclosed, sensitive marine environment with unique eutrophication pressures, the insights carry relevance for other regional seas. The Mediterranean, Adriatic, Canary Islands, and parts of the Caribbean have already demonstrated that conventional treatment alone rarely yields positive net societal benefits; instead, reuse and circular-water strategies are the key drivers of economic viability. Likewise, misalignment of incentives between ships and ports has been repeatedly documented in the Atlantic and Mediterranean basins, mirroring the patterns observed here.

Sea regions experiencing water scarcity such as the Mediterranean, Middle East, Australia, U.S. West Coast, would likely experience even higher environmental and economic justification of reuse, because freshwater substitution has much greater value. The value of freshwater substitution is significantly higher in those contexts, meaning reuse scenarios would likely yield positive net society. In these regions, as in the Baltic, advanced treatment without reuse seldom achieves positive net societal benefits, reinforcing the conclusion that circular water strategies are crucial for economic viability. In those contexts, the NSB of reuse scenarios may shift from negative to strongly positive. Conversely, regions without environmental governance structures like HELCOM may find it more difficult to implement coordinated policies or shared infrastructure investment.

Thus, the Baltic Sea offers an important case study for integrated policy design, highlighting how environmental protection, maritime governance, and circular water systems intersect. The framework and findings developed here can inform regulatory debates in other passenger-shipping regions worldwide, seeking to upgrade maritime wastewater governance and support the development of harmonized approaches to greywater management in international shipping.

5 - Conclusions and Recommendations

This study has delivered the first comprehensive, multi-scenario cost-benefit analysis (CBA) of greywater management for Baltic Sea passenger ships, using the Port of Trelleborg as a representative high-traffic RoPax case. By integrating societal costs, stakeholder financial outcomes, and monetized environmental benefits based on pollutant shadow prices, the analysis quantifies the full economic and ecological consequences of the distinct greywater pathways, ranging from direct sea discharge to advanced treatment and full reuse. The study directly addresses a long-standing gap in maritime wastewater governance, where greywater, despite its significant pollutant load, remains unregulated under MARPOL Annex IV and receives far less policy attention than sewage or other ship-generated waste.

The results reveal that non-reuse scenarios (SC1 – SC5A) consistently produce negative net societal benefits (NSB), even when advanced port and municipal treatment are applied. SC1, involving direct discharge, has the lowest societal cost but provides no environmental benefit and is misaligned with the objectives of the HELCOM BSAP and the EU MSFD. SC4, combining port-level PWTP polishing with MWTP treatment, achieves high pollutant removal but at the highest cost, yielding the most negative NSB of all scenarios. Intermediate scenarios such as SC3 (PRF - MWTP) and SC5A (PRF - PWTP - sea) reduce pollutant loads to varying degrees, but their environmental benefits remain insufficient to offset their real resource costs. These findings confirm earlier Mediterranean and Atlantic CBA studies, notably Hernandez-Sancho et al. (2010), Molinos-Senante et al. (2011), and Martinez-Lopez et al. (2020), which have shown that advanced wastewater treatment alone rarely generates positive welfare outcomes unless coupled with additional value streams.

A transformative shift emerges with reuse scenarios ($SC5B_1$ - $SC5B_5$). Although environmental benefits from pollutant removal contribute meaningfully, it is freshwater substitution that fundamentally reshapes societal outcomes. Reuse provides substantial additional environmental benefits (EB_{reuse}), and although total NSB figures remain negative under Swedish price assumptions, reuse scenarios, particularly $SC5B_1$ (100% reuse), perform significantly better than all non-reuse alternatives. These results mirror international experience from Spain, Cyprus, Israel, and Australia, where the avoided cost of potable-water production is the key driver of economic viability of reuse projects, even in contexts without extreme water scarcity. Partial reuse scenarios with MWTP polishing ($SC5B_2$ and $SC5B_3$) demonstrate cost-efficient hybrid configurations that balance real resource costs, environmental performance, and circular water benefits, offering pragmatic transitional solutions for ports that cannot accommodate full reuse volumes.

When the CBA is reparametrized using measured concentrations in the higher range in ship-generated greywater instead of the averages, the relative performance of the scenarios shifts substantially and positive values for NSB are archived in some cases. Under this high-contamination assumption, the marginal damage of direct discharge (SC1) increases sharply, and the additional avoided damage achieved through treatment and reuse become much more pronounced. As a result, reuse configurations (SC5B1 – SC5B5) move from being "less negative" options under Swedish price assumptions to becoming the socially preferred choices, with markedly higher NSB than all non-reuse alternatives, including SC1. In other words, the ranking of options is highly sensitive to the assumed pollutant load. When greywater quality is relatively benign, the high real-resource costs of treatment and reuse dominate, whereas under upper-bound concentration conditions, the environmental and freshwater-substitution benefits associated with reuse are sufficiently large to outweigh the cost disadvantage and push reuse scenarios to the top of the welfare hierarchy.

However, the stakeholder analysis highlights a profound misalignment of incentives. Ship operators incur substantial financial losses under both non-reuse and reuse scenarios, driven primarily by MWTP fees and PWTP O&M costs. Municipalities operating under cost-recovery tariffs remain financially neutral. Consequently, the actors who bear the highest direct costs (ships and ports) capture none of the environmental or societal benefits of reuse. This mirrors a pattern observed in the Las Palmas MARPOL Annex IV study and Baltic port assessments: without regulatory drivers or economic incentives, voluntary adoption of advanced GW management is highly unlikely.

From a policy perspective, the findings demonstrate that Baltic ports occupy a pivotal but constrained role in enabling sustainable circular water systems. As semi-enclosed waters heavily affected by eutrophication, hazardous substances, and micropollutants, the Baltic Sea stands to benefit significantly from improved GW management. Yet the governance vacuum around GW, stemming directly from its omission from MARPOL Annex IV, creates major implementation barriers. The environmental targets of HELCOM BSAP and EU MSFD cannot be met solely through voluntary measures or ship-based initiative; rather, a coordinated policy framework is required that aligns the incentives of ships, ports, and municipalities with societal environmental goals.

Shadow-price modelling was feasible but sensitive to data quality, reflecting variance and limited sample size in the underlying datasets used to estimate pollution damage costs. Although the results yield realistic and policy-consistent environmental valuations, future work should focus on improving data resolution, refining removal-efficiency assumptions for reuse trains, and incorporating co-benefits such as microplastic removal, pharmaceutical reduction, and avoided eutrophication damage.

Ultimately, this study demonstrates that a transition from linear "discharge-based" GW management to circular "treat-and-reuse" systems is both an environmental imperative and a strategic opportunity for the Baltic Sea region. Treatment alone is insufficient to generate positive welfare, but reuse-capable systems significantly improve societal outcomes and contribute to BSAP/MSFD pollution-reduction targets. Realizing these benefits will require a coordinated suite of financial, regulatory, and voluntary measures. Economic instruments, such as differentiated port fees, cost-sharing models, subsidies, and EU co-financing, must complement regional governance reforms, including the possible introduction of a Baltic Sea greywater requirement under HELCOM or amendments to MARPOL Annex IV through IMO processes.

To support implementation, the region should adopt a coordinated progression in which voluntary adoption (e.g., EcoPort incentives, sustainability branding, reduced PRF fees) evolves into harmonized regional standards and eventually formal regulatory requirements. Shared monitoring of effluent quality, pollutant loads, and economic performance across ports and utilities will be essential for refining long-term targets and achieving environmental objectives. Leveraging EU programs such as LIFE (EU programme for the Environment and Climate Action), Interreg Baltic Sea Region, Horizon Europe's

"Mission Ocean", and national environmental funds can reduce financial risks and accelerate infrastructure deployment.

In conclusion, the Baltic Sea's transition toward sustainable greywater management requires aligning policy incentives, and technical innovation across maritime and municipal actors. By embracing circular water principles, advanced treatment, and reuse, the region can significantly reduce pollutant loads, conserve freshwater, and advance toward Good Environmental Status. A coordinated, multi-actor strategy, built on robust CBA evidence, economic instruments, and targeted regulations, will enable ports and ships to collectively contribute to a cleaner, more resilient Baltic Sea ecosystem while supporting Europe's broader climate and resource-efficiency ambitions.

5.1 - Limitation of the study

While this study provides a detailed CBA of greywater management scenarios, several limitations should be acknowledged to contextualize the results and guide future research.

Data Availability and Quality: The analysis relies heavily on operational data from Trelleborg, including the volume of greywater available for reuse. Variability in daily operations, seasonal passenger loads, and maintenance schedules could lead to fluctuations in greywater generation that are not fully captured in the annualized estimates. Additionally, some system parameters, such as pump efficiency and energy consumption, are based on literature values rather than direct measurements on the vessels, which introduces uncertainty in cost estimates. For **shadow price calculations**, the dataset was not fully reliable, with high risk of misreported costs and inconsistent reporting structures. The method is highly sensitive to data dispersion: standard deviations often exceeded means, unlike the healthier dataset in the reference study. Strong dominance of certain pollutants (TOC, COD, BOD) distorted the balance of shadow prices. Moreover, categorization by plant size (pe) would make sense (and the data tells this story in which biggest plants were deemed the outliers) but would require more datapoints per group to be meaningful.

CAPEX and OPEX Estimations: Capital expenditure estimates used unit cost multipliers from existing literature due to lack of such data from the Trelleborg GW management system, and an integration factor to account for installation and contingencies. These values may not fully reflect the specific costs and constraints of retrofitting a passenger ship in the Baltic Sea context as well as the Trelleborg PWTP. Similarly, operational costs, including energy, maintenance, and chemical dosing, are derived from general literature ranges. Actual costs could differ significantly due to local energy prices, labour costs, and system-specific efficiency.

Simplified Annualization: The CAPEX annualization assumes a fixed system lifetime (10 – 15 years) and a uniform discount rate of 5%. System degradation, maintenance interventions, and economic conditions may affect the effective lifetime and discount rate, potentially altering the annualized cost. This approach also assumes constant operating days and greywater generation, ignoring possible downtime, seasonal variations, or operational disruptions.

Exclusion of Indirect Environmental and Social Impacts: The societal reuse cost calculation incorporates savings from potable water use and avoided municipal wastewater treatment costs but does not fully account for other environmental or social factors, such as reduced eutrophication, microplastic retention, or public perception benefits. As such, the computed societal cost may underestimate the broader ecological and social benefits of greywater reuse.

System Performance Uncertainty: The analysis assumes that the greywater reuse system achieves consistent treatment efficiency and reliability. In practice, factors such as fouling, variability in influent water quality, or human error could reduce system performance and increase operational costs, which are not explicitly modeled in this study.

Future Cost Trends: The study does not account for potential future changes in water pricing, energy costs, or regulatory incentives, which could influence the economic attractiveness of greywater reuse over the system's lifetime. Sensitivity analyses could partially address this, but inherent uncertainty remains.

Overall, while the study provides a rigorous framework for evaluating costs and benefits across multiple scenarios, these limitations highlight the need for onboard-, port- and municipality-specific measurements, long-term operational monitoring, and context-specific adjustments to improve accuracy and applicability.

5.2 - Recommendations for ship operators, regulators, and technology providers

The implementation of sustainable GW management in the Baltic Sea region requires coordinated action among ship operators, port authorities, technology providers, and regulators. The CBA results demonstrate that advanced port reception and treatment systems can deliver both economic and environmental gains, but their realization depends on clear responsibilities, financing mechanisms, and supportive regulatory frameworks.

For ship operators, the findings emphasize the value of delivering GW to PRFs rather than discharging at sea. When supported by differentiated port fees or recognition under environmental indexing schemes, participation in port-based treatment systems can yield long-term economic and reputational benefits. Operational data with environmental performance indicators should be used to measure performance and recognition made. Engaging in early adoption can also help shape practical regulatory standards and position companies as leaders in sustainable maritime operations.

For port authorities, the establishment of cost-recovery mechanisms is essential. Ports can recover investments in reception, treatment, and reuse infrastructure through transparent service fees paid by shipowners. These can be structured as part of existing waste management fees, environmental service tariffs, or performance-based "green port" incentives. A standardized pricing framework, coordinated at regional level, would enhance predictability for investors and fairness among users.

For regulators and policymakers, the priority should be to integrate GW management into existing port state control, HELCOM, and MSFD monitoring regimes. This includes harmonizing definitions of "treated" and "reused" water an ensuring that water reuse aligns with non-potable quality standards applicable to port facilities. Regulatory evolution should be complemented by flexible instruments such as eco-differentiated fees, tax incentives, or innovation grants that encourage voluntary compliance and technology adoption before stricter discharge limits come into force.

To support implementation, dedicated funding channels should be mobilized. The EU LIFE Programme, Horizon Europe Mission "Restore Our Ocean and Waters", Interreg Baltic Sea Region, and Connecting Europe Facility (CEF) – Transport which offer suitable instruments for demonstration projects, infrastructure upgrades, and cross-border coordination, are some examples. National environmental and maritime agencies can complement these through targeted green transition funds or public-private partnership frameworks that de-risk capital investments.

At the regional level, establishing a Baltic Circular Port Facility Fund could pool financing for wastewater reception, treatment, and reuse installations across passenger ship ports. Such a fund would allow economies of scale, enabling smaller ports to benefit from shared procurement and knowledge transfer.

Aligning funding with HELCOM BSAP targets and MSFD Programme of Measures cycles would ensure that environmental and financial accountability progress together, linking measurable pollutant reduction with transparent investment returns.

For technology providers, innovation should focus on modular, energy-efficient, and easily maintainable treatment systems adapted for both shipboard and port-based deployment. Partnerships between technology developers and port authorities can facilitate system standardization, interoperability, and long-term maintenance agreements. Integrating digital monitoring tools into treatment processes will also be crucial for reporting and verification under future regulatory frameworks.

In summary, realizing the full societal and environmental value of sustainable GW management requires an integrated approach: operational commitment by ship operators, economic foresight by ports, regulatory clarity by authorities, and technological innovation by solution providers. Strategic funding coordination at EU and national levels can turn the Baltic region into a model of circular and low-impact water governance, demonstrating that economic efficiency and ecological restoration are not competing objectives but mutually reinforcing pillars of maritime sustainability.

5.3 - Research and policy gaps for future work

Several knowledge gaps remain. There is limited longitudinal data on greywater pollutant variability across stream types, ship types, seasons, and operating conditions, which constrains model calibration and impact assessment accuracy. The data reported in Mujingni et al. (2024) is currently the most comprehensive within the Baltic Region. Future studies should expand monitoring coverage and integrate real-time sensing technologies to improve emission inventories.

On the policy side, further work is needed to define reuse water quality standards for non-potable port applications and to harmonize these across Baltic Countries. This could be included as an amendment to the Water Reuse Directive (EU Regulation 2020/741) which establishes minimum requirements for the reuse of treated wastewater in the EU, aiming to enhance water security and sustainability in agriculture. Economic research should explore cost-recovery models that equitably distribute investment burdens among stakeholders, including mechanisms for performance-based financing. Moreover, the need for the regulation of greywater discharges cannot be overemphasized; hence it should be included in MARPOL Annex IV.

Integrating environmental valuation into port management systems, through tools such as shadow pricing or pollution crediting, could enhance the visibility of ecosystem service benefits and incentivize long-term adoption. Continued interdisciplinary collaboration between economists, engineers, and policymakers will be essential for translating technical feasibility into actionable governance.

5.4 - Outlook

The outcome of this study points to a clear strategic opportunity for the Baltic Sea region to lead Europe in circular water management within the maritime sector. The comparative cost-benefit and environmental analyses of the ten greywater management scenarios show that achieving significant pollution reduction and water reuse is both technically feasible and socio-economically justified, if coordination mechanisms, financing instruments, and incentive structures are properly aligned. The results demonstrate that port-based greywater management and reuse can generate tangible economic and environmental benefits across the maritime waste management chain. However, these benefits represent only the first step toward a fully circular maritime water and wastewater system.

The next frontier in implementing circular maritime waste management could point at the integration of greywater, blackwater, and food waste co-treatment, a co-management approach that explores the source-segregation of these waste streams and their combined valorization in a similar model as the RecoLab model currently implemented onshore in Helsingborg, Sweden. This approach could potentially enhance resource recovery, minimize energy use, and further substantially reduce the nutrient footprint of passenger shipping in the Baltic Sea. When adapted for shipboard applications, such systems could separate greywater from blackwater, allowing each fraction to be treated optimally: greywater for water reuse and blackwater and food waste for biogas, nutrient recovery as struvite and ammonium sulphate fertilizers. This "Three-Pipes-Out" or "Four-Pipes-Out" approach could possibly turn ships and ports into circular resource nodes, directly supporting EU and HELCOM BSAP and MSFD.

To advance this next phase, dedicated pilot projects are needed to assess the technical feasibility, regulatory acceptance, and circular value chains of onboard biowaste co-management. Such projects could be co-funded under a variety of funding programmes. Partnerships among ship operators, ports, technology developers, and research institutions will be key to transforming current wastewater liabilities into resource recovery opportunities.

Ultimately, by extending the insights from the ten-scenario greywater management analysis to include integrated biowaste co-management, the Baltic Sea region can lead the global transition toward closed loop maritime sanitation systems. This aligns with long-term vision of zero-discharge, resource-positive shipping sector, where waste streams are not merely treated, but repurposed as clean water, energy and nutrients for a sustainable blue economy.

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7 - APPENDICES

7.1 - APPENDIX 1

Contaminants in Wastewater Streams.

***Table 1**: Concentration and loads of contaminants in greywater from ships calling at the Trelleborg port (Q_{GW}: ≈ 40,528 m^3/year).*

Greywater			
Contaminants	**Unit**	**Mean concentration**[1]	**Contaminant loads into the $PWTP_{Trell}$ via ship-generated greywater (Kg/year)**[2]
Pb	µg/L	3.03	0.123
Cd	µg/L	0.12	0.0049
Cu	µg/L	126	5.10
Mn	µg/L	52.14	2.11
Cr	µg/L	3.98	0.161
Ni	µg/L	10.14	0.410
Zn	µg/L	264	10.69
TSS	mg/L	132	5343
BOD5	mg/L	431	17446
COD-Cr	mg/L	888	35944
Phosphorus	mg/L	12.5	506
Nitrogen	mg/L	17.1	692
PET	MPs/L	119000	140860
PP	MPs/L	33000	25501

[1] – Based on Mujingni et al. (2024)

[2] - Loads are based on estimated average annual volume of GW received at the Trelleborg port and conveyed to the $PWTP_{Trell}$.

***Table 2**: Concentration and loads of contaminants in sanitary wastewater (mixed BW and GW) from ships calling at the Trelleborg port (≈ 50,660 m^3/year).*

Mixed black- and greywater			
Contaminants	**Unit**	**Mean concentration**	**Contaminant loads flowing into the $PWTP_{Trell}$ via mixed grey and blackwater (kg/year)**
Pb	µg/L	4.90	0.248
Cd	µg/L	0.30	0.0152
Cu	µg/L	146	7.396
Cr	µg/L	8.25	0.418
Ni	µg/L	10.33	0.522
Zn	µg/L	1255	63.58
TSS	mg/L	524	26546
BOD_5	mg/L	359	18187
COD-Cr	mg/L	1219	61755
Phosphorus	mg/L	30.9	1565
Nitrogen	mg/L	243	12310

***Table 3**: Concentration and loads of contaminants in GW from a ship (S3: Mujingni et al., 2024) at the Trelleborg port (Q_{GW}: 1133 m^3/yr).*

	$S3_{A/L}$ Trelleborg Port	$S3_{A/L}$ Trelleborg Port	$S3_{A/L}$ by Mujingni et al.	$S3_G$ Trelleborg Port	$S3_G$ Trelleborg Port	$S3_G$ by Mujingni et al.	MEAN GW Pollutant conc.	Unit	MEAN (Kg/m^3)	LOADS (Kg/Year)
Sample date	2022-09-16	2023-02-10	2023	2022-09-16	2023-02-10	2023				
	Mixed A/L	Mixed A/L	Mixed A/L	Galley	Galley	Galley	MIXED GW			
Pb	1,2	1,5	8,48	0,69	0,79	1,44	2,35	µg/L	0,00000235	0,00266
Cd	0,034	0,063	0,025	0,015	0,093	0,025	0,04	µg/L	0,00000004	0,00005
Cu	110	220	111	93	280	43,9	143	µg/L	0,00014298	0,162
Cr	1,1	1,9	0,45	1,4	1,7	1,06	1,27	µg/L	0,00000127	0,0014
Ni	3,2	4,1	9,7	4,8	3,7	5,61	5,19	µg/L	0,00000519	0,006
Zn	120	280	124	150	290	116	180	µg/L	0,00018	0,204
TSS	16	93	20	220	290	190	138	mg/L	0,13817	157
BOD_5	28,7	104	68,1	600	1044	757	434	mg/L	0,43363	491
COD-Cr	79	380	125	880	2100	1180	791	mg/L	0,79067	896
P_{TOT}	1,5	7,8	2,96	20	28	23,4	14	mg/L	0,01394	15,80
N_{TOT}	9,4	38	10,2	28	29	26,5	24	mg/L	0,02352	26,64

***Table 4**: Concentration of selected contaminants in wastewater from RoPax ships at Trelleborg Port (Data collected between 2021 and 2023).*

	Unit	SH1	SH2	SH3	SH4	SH5	SH6	SH7	SH8	SH9	SH10	SH11	SH12
Pb	µg/L	2.90	4.76	2.19	14.15	4.08	5.09	5.32	6.01	1.82	3.74	2.73	8.65
Cd	µg/L	0.273	0.27	0.21	0.60	0.35	0.36	0.28	0.32	0.21	0.22	0.21	0.46
Cu	µg/L	165	125	116	644	90.5	126	143	117	71.1	119	82.3	143
Cr	µg/L	5.36	6.24	5.35	53	4.96	5.62	7.04	5.60	2.58	5.14	4.21	5.74
Ni	µg/L	7.27	7.35	8.01	37.4	10.4	8.38	8.06	8.26	6.18	8.69	6.63	11.8
Zn	µg/L	758	1,073	1,140	7,190	802	1,324	1,353	894	439	752	631	1,628
TSS	mg/L	564	362	409	990	651	634	510	469	412	362	345	1,163
Fat	mg/L	138	117	148	156	66.1	111	121	47.9	39.2	137	98.3	30.5
BOD_5	mg/L	2.90	4.76	2.19	14.2	4.08	5.09	5.32	6.01	1.82	3.74	2.73	8.65
COD-Cr	mg/L	1,255	954	1,137	3,036	1,130	1,223	1,106	1,090	802	1,117	1,050	1586
Phosphorus	mg/L	28.41	18.18	26.79	78.1	32.9	44.1	24.32	31.6	31.9	20.7	17.2	39.8
Nitrogen	mg/L	202	134	231	518	195	150	151	384	288	209	187	315

Contaminants	Unit	SH13	SH14	SH15	SH16	SH17	SH18	SH19	SH20	MEAN	SD	Annual Loads (kg/year)
Pb	µg/L	7.21	1.79	1.90	1.60	5.72	2.15	1.20	17	4.90	7.67	0,00025
Cd	µg/L	0.36	0.15	0.14	0.055	0.25	0.18	0.14	0.64	0.30	0.27	0,000015
Cu	µg/L	144	102	67	80	103	63.0	56	490	146	388	0,00740
Cr	µg/L	12.3	4.42	4.40	2.35	5.79	3.30	3.70	19	8.25	36.4	0,00042
Ni	µg/L	20.2	6.32	5.30	3.25	11.0	5.17	7.30	16	10.33	23.5	0,00052
Zn	µg/L	620	478	540	250	550	334	410	1,100	1,255	5,565	0,0635
TSS	mg/L	442	286	250	116	342	214	230	1,000	524	529	26,55
Fat	mg/L	80.9	103	92	69	197	93.6	93	190	99.7	107	
BOD_5	mg/L	7.21	1.79	146	1.6	5.72	2.15	322	539	359	341	18,23
COD-Cr	mg/L	1383	977	780	455	945	783	910	2,200	1,219	1,625	61,77
Phosphorus	mg/L	36.6	20,0	16	9.35	25.8	16.0	13	62	30.9	45.5	1,57
Nitrogen	mg/L	445	88	92	44	282	140	160	670	243	216	12

Table 5*: Contaminants in wastewater from ships into the $PWTP_{Trell}$, and effluent from the $PWTP_{Trell}$ into $MWTP_{Trell}$, including treatment efficiencies at the plant (average of 2 data sets).*

Contaminant	Influent concentration	Effluent concentration	Removal efficiency	Loads into $MWTP_{Trell}$ (kg/year)	ABVA	P95
N_{tot}	156 mg/L	88 mg/L	43%	4458	-	-
P_{tot}	21 mg/L	5.85 mg/L	72%	296	-	-
TSS	245 mg/L	69 mg/L	72%	3496	-	-
BOD_7	420 mg/L	42 mg/L	90%	2128	-	-
COD-Cr	885 mg/L	220 mg/L	75%	11145	-	-
Fat_{tot}	84 mg/L	8.2 mg/L	90%	415	-	-
Zn	550 µg/L	99 µg/L	82%	5.01	200	200
Cu	118 µg/L	31 µg/L	74%	1.57	200	200
Pb	1.85 µg/L	0.515 µg/L	72%	0.0261	50	10
Cd	0.135 µg/L	0.031 µg/L	77%	0.00157	0.21	0.10
Ni	8.95 µg/L	3.8 µg/L	58%	0.1925	50	10
Cr	5.15 µg/L	0.995 µg/L	81%	0.0504	50	10

Table 6*: Contaminants flowing in and out of the Trelleborg MWTP, including their treatment efficiencies at the plant in 2023 (Trelleborg MWTP Sustainability report of 2023).*

Contaminant	Influent concentration	Effluent concentration	Removal efficiency	Loads from $MWTP_{Trell}$ (kg/year)	UWTD requirements
TSS	216 mg/L	6.4 mg/L	97%	26738	35 mg/L
BOD_7	165 mg/L	2.2 mg/L	99%	9191	25 mg/L
COD-Cr	365 mg/L	24 mg/L	93%	100269	125 mg/L
N_{tot}	42 mg/L	7.6 mg/L	82%	31751	8 mg/L*
P_{tot}	4.6 mg/L	0.22 mg/L	95%	919	0.7 mg/L **
Zn	107 µg/L	16 µg/L	85%	66.9	-
Cu	58 µg/L	11 µg/L	81%	46	-
Pb	1.6 µg/L	0.31 µg/L	81%	1.30	-
Cd	0.085 µg/L	0.015 µg/L	82%	0.063	-
Ni	3.4 µg/L	2.8 µg/L	18%	11.7	-
Cr	1.4 µg/L	0.40 µg/L	71%	1.67	-
* Based on the population equivalent (p.e.) served by the Trelleborg MWTP					

***Table** 7: Formula for calculating Stakeholder costs of modelled scenarios.*

Modelled Scenarios	**SHIP (SH)**	**PORT (PT)**	**MUNICIPALITY (MPAL)**
SC1	$-(C_{Pot} + C_{Stor})$	0	0
SC2	$-(C_{Pot} + C_{\_AWTP})$	0	0
SC3	$-(C_{Pot} + C_{\_Stor} + F_{PRF})$	$F_{PRF} - (C_{PRF} + T_{MWTP})$	$T_{MWTP} - C_{MWTP}$
SC4	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$F_{PRF} + F_{PWTP} - (C_{PRF} + C_{PWTP} + T_{MWTP})$	$T_{MWTP} - C_{MWTP}$
SC5A	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$F_{PRF} + F_{PWTP} - (C_{PRF} + C_{PWTP})$	0
SC5B1	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$(F_{PRF} + F_{PWTP}) - (C_{PRF} + C_{PWTP} + C_{Reuse}) + S_{\_pot\_port\_100\%}$	0
SC5B2	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$(F_{PRF} + F_{PWTP}) - (C_{PRF} + C_{PWTP} + C_{Reuse} + (1 - f_{75}) * T_{MWTP}) + S_{pot\_port\_75\%}$	$(1 - f_{75}) * T_{MWTP} - (1 - f_{75}) * C_{MWTP}$
SC5B3	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$(F_{PRF} + F_{PWTP}) - (C_{PRF} + C_{PWTP} + C_{Reuse} + (1 - f_{50}) * T_{MWTP}) + S_{pot\_port\_50\%}$	$(1 - f_{75}) * T_{MWTP} - (1 - f_{75}) * C_{MWTP}$
SC5B4	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$(F_{PRF} + F_{PWTP}) - (C_{PRF} + C_{PWTP} + C_{Reuse}) + S_{pot\_port\_75\%})$	0
SC5B5	$-(C_{Pot} + C_{Stor} + F_{PRF} + F_{PWTP})$	$(F_{PRF} + F_{PWTP}) - (C_{PRF} + C_{PWTP} + C_{Reuse}) + S_{pot\_port\_50\%})$	0

$^{*}S_{pot\_port100\%)}$	$= SP_{water} * (f_{100} * Q_{reuse})$.	SP_{water}	is the economic value or avoided cost per cubic meter of potable water replaced by reused, treated GW at the Port (SEK/m3)
T_{MWTP}	Municipal Wastewater Tariff	F_{PWTP}	Fee for PWTP use
C_{Reuse}	Cost of infrastructure upgrade of PWTP to reuse standard	C_{PWTP}	Cost of wastewater treatment at PWTP
C_{Pot}	Cost of freshwater production	C_{Stor}	Cost of GW storage on board
F_{PRF}	Fee for PRF use	f	Reuse fraction

7.2 - APPENDIX 2

Municipal Wastewater Treatment Plant (MWTP) used for Shadow Price Calculations.

***Table 1**: OPEX (SEK/year) and volume of treated effluent (m^3/year) of MWTPs analyzed for shadow price calculations.*

No.	MWTP CODES	PE	Cost per PE	Treated effluent per PE	N per PE	N per treated effluent	Total input cost (positives)	Energy	Staff	Reagents	Maintenance	Waste Management	Treated effluent
1	MWTP1	797485	316,91	106	3,29	0,031	252731370,1	9599008	1,42E+08	26646047	49996009	24735034	84744960
2	MWTP2	541285	339,95	84	3,96	0,0468	198629618,8	-7310000	1,12E+08	4861010	50394987	24041985	45728915
3	MWTP3	369900	238,09	111	3,27	0,0294	88070637,69	14212002	31091649	2911002	19927993	19927993	41185079
4	MWTP4	343892	405,97	99	3,31	0,0334	139608700,4	11334990	66478691	16780004	31190007	13825009	34033531
5	MWTP5	155609	373,69	74	5,75	0,0779	58150019,69	11500005	22200011	4049994	10200004	10200004	11485371
6	MWTP6	49117	231,89	76	0,29	0,0037	11389770,6	65001	5276772	880000	3269999	1897999	3746266
7	MWTP7	46428	422,96	64	4,65	0,0722	19637016,14	2751002	9998007	2300003	2594002	1994002	2990866
8	MWTP8	45120	496,79	111	1,51	0,0136	22415137,73	2558002	15625142	0	2665996	1565998	4995584
9	MWTP9	44103	161,66	78	5,39	0,0693	7129807,67	0	0	1652203	2738802	2738802	3428329
10	MWTP10	36195	137,04	130	4,15	0,032	5559997,199	-299999	0	2259998	1500000	1500000	4700058
11	MWTP11	36000	581,78	110	3,71	0,0336	20944094,4	1679000	12000092	1370999	3948001	1946002	3974986
12	MWTP12	31880	140,12	78	3,24	0,0413	4467000,096	1385999	0	3081001	0	0	2501243
13	MWTP13	31687	174,93	66	3,82	0,0575	5543000,573	1951000	589999	0	1796000	1206001	2105038
14	MWTP14	30000	667,03	174	1,18	0,0068	20010999	3051000	9690000	2802999	3012000	1455000	5217836
15	MWTP15	27235	835,17	212	1,85	0,0087	22745968,43	1494199	7893269	2046001	6012500	5299999	5768883
16	MWTP16	25207	1171,53	111	3,45	0,031	29530700,8	700701	24522600	621400	3164800	521200	2805650
17	MWTP17	24774	67,09	85	2,37	0,0279	1662000,951	1662001	0	0	0	0	2104093
18	MWTP18	23737	488,19	50	1,17	0,0235	11588147,04	2800000	5428146	559999	1900002	900000	1176051
19	MWTP19	23168	183,01	15	6,95	0,4569	4239999,991	1611000	515001	0	1873000	240999	352206
20	MWTP20	20560	465,12	131	3,41	0,026	9562854,864	1770000	3582856	0	2659999	1550000	2702155
21	MWTP21	18298	226,47	114	0,43	0,0037	4144001,124	0	2136001	0	1004000	1004000	2082478
22	MWTP22	18000	461,83	134	-1,32	-0,0098	8312999,4	1468001	2930000	376999	2588999	949000	2418093

23	MWTP23	16800	719,23	-43	2,66	-0,0619	12083000,16	1030000	6896000	423000	2334000	1399999	-721224
24	MWTP24	16746	1063,2	111	2,32	0,021	17804400,79	0	10817400	0	6987001	0	1856700
25	MWTP25	15600	483,72	132	1,81	0,0137	7545999,24	156000	5399999	650001	670000	670000	2062911
26	MWTP26	15050	426,52	136	0,94	0,0069	6418943,822	943201	2977743	0	1632600	865400	2051247
27	MWTP27	14541	205,31	115	-0,82	-0,0072	2985451,971	1191354	0	0	897049	897049	1665450
28	MWTP28	14418	155,91	165	1,4	0,0085	2248000,525	921000	309000	630001	382000	5999	2378047
29	MWTP29	14006	249,82	122	1,1	0,009	3498999,929	704000	644000	0	1569000	582000	1704368
30	MWTP30	11930	271,75	114	4,1	0,0359	3242000,167	840000	714000	250000	1076000	362000	1364585
31	MWTP31	11554	266,44	154	1,26	0,0081	3078499,28	526000	460000	290000	1177500	625000	1782006
32	MWTP32	11545	431,27	101	0,64	0,0063	4979000,605	1584001	1700000	375000	1210000	110000	1166642
33	MWTP33	11434	270,51	40	4,79	0,1193	3092999,02	443000	295000	0	1324999	1030000	459435
34	MWTP34	11250	1053,62	124	0,99	0,008	11852872,01	1076999	5732874	612999	3065000	1365000	1391553
35	MWTP35	11057	489,28	77	3,36	0,0438	5409998,814	1093999	1992000	226000	1494000	604000	849124
36	MWTP36	11032	587,2	2	3,53	1,7435	6478001,432	1074001	3814000	0	1060001	530000	22342
37	MWTP37	10899	526,85	77	1,24	0,0161	5742161,407	714000	1072161	0	1978000	1978000	836097
38	MWTP38	10300	442,04	15	3,41	0,2209	4552999,64	665000	2538000	0	889999	460000	158906
39	MWTP39	5579	272,27	54	1,7	0,0313	1518999,909	583000	0	248000	344000	344000	302569
40	MWTP40	5204	250,06	170	0,49	0,0029	1301321,087	535049	0	0	383136	383136	885934
41	MWTP41	4689	146,29	64	-0,14	-0,0021	686000,1746	456000	0	230000	0	0	298165
42	MWTP42	4165	646,95	107	-0,12	-0,0012	2694529,257	131000	2010529	0	393000	160000	443664
43	MWTP43	4158	131,55	175	1,12	0,0064	546999,8688	355000	0	0	96000	96000	728084
44	MWTP44	3785	310,7	27	2,85	0,107	1176000,257	334000	0	39000	594000	209000	100786
45	MWTP45	3400	794,69	93	3,47	0,0374	2701933,08	687000	1061933	0	953000	0	314993
46	MWTP46	3396	909,31	111	2,27	0,0205	3087999,78	149000	2115000	105000	719000	0	376259
47	MWTP47	3248	129	-32	0,52	-0,0162	419000,12	98000	0	34000	241000	46000	-104484
48	MWTP48	3124	955,51	113	1,34	0,0118	2985000,119	195000	1315000	0	1026000	449000	354451
49	MWTP49	3050	111,05	130	4,48	0,0343	338700,06	64700	0	105600	85300	83100	397654
50	MWTP50	3012	329,35	55	2,09	0,0378	992000,0916	196000	86000	0	430000	280000	166172
51	MWTP51	2800	1948,21	154	2,75	0,0179	5455000,32	259000	3115000	181000	1450000	450000	430211
52	MWTP52	2671	219,46	99	0,91	0,0092	586166,4418	304140	0	0	141013	141013	264124
53	MWTP53	2651	606,56	143	0,37	0,0026	1607999,839	299000	1228000	0	81000	0	377914
54	MWTP54	2594	968,02	121	1,55	0,0128	2510567,132	453000	571567	0	743000	743000	315035
55	MWTP55	2565	1000,26	231	0,11	0,0005	2565669,978	189000	1576670	0	560000	240000	593000

56	MWTP56	2522	369,7	72	1,61	0,0224	932395,2534	452105	0	0	240145	240145	181478
57	MWTP57	2510	1049,4	10	1,37	0,1438	2634000,024	205000	2110000	0	319000	0	23907
58	MWTP58	2500	1004,2	204	-0,32	-0,0016	2510500	98000	1927500	85000	400000	0	511152
59	MWTP59	2491	435,65	180	2,3	0,0128	1085334,098	221000	482334	0	191000	191000	447274
60	MWTP60	2309	364,61	76	0,57	0,0076	842000,0349	242000	93000	132000	240000	135000	174623
61	MWTP61	2254	570,86	185	1,58	0,0086	1287000,042	217000	88000	128000	479000	375000	416050
62	MWTP62	2200	1175	89	-0,67	-0,0075	2585000	283000	1123000	167000	689000	323000	195058
63	MWTP63	2132	305,65	117	0,34	0,0029	651524,6058	170000	163524	0	159000	159000	249999

Table 2*: Contaminant loads from the studied MWTPs (kg/year)*

No.	MWTP CODES	P	N	BOD	COD	TOC	Zn	Cu	Mn
1	MWTP1	395074	2626437	2694702	13019660	44214243	10518	7143	6508
2	MWTP2	319845	2141486	2556814	13209086	30667747	8455	6616	4417
3	MWTP3	173890	1210017	1521991	5974976	2389990	8682	5788	3019
4	MWTP4	145707	1136804	1167101	4238985	16037300	9384	4835	2806
5	MWTP5	72406	894768	687003	4630002	1852001	7207	4155	1628
6	MWTP6	-2102	14003	152999	497000	2063397	12056	8129	401
7	MWTP7	29621	216000	253700	1127758	2962002	12614	5418	379
8	MWTP8	24640	68014	205896	605808	2510239	45892	8767	368
9	MWTP9	20521	237555	162730	909200	1968080	7004	4155	360
10	MWTP10	0	150253	177851	1002540	3046602	9661	6951	295
11	MWTP11	18353	133524	155038	656989	1711397	9098	6392	294
12	MWTP12	16970	103406	138742	622447	1961733	8626	5669	260
13	MWTP13	15257	121000	147598	559000	1413000	13477	6382	259
14	MWTP14	11361	35547	66597	329223	853812	14469	8009	245
15	MWTP15	14410	50301	91499	312152	1486500	9377	4540	222
16	MWTP16	13171	86934	107099	499652	199861	9371	5645	206
17	MWTP17	10970	58690	71453	443133	1187807	10586	8013	202
18	MWTP18	4192	27658	33631	113930	443147	8375	4927	194

19	MWTP19	4914	160912	22230	812963	1503222	3317	1015	189
20	MWTP20	12554	70200	91280	433999	1043601	10068	5034	168
21	MWTP21	11617	7789	72689	278486	1243785	10297	5279	149
22	MWTP22	-3665	-23776	50130	165040	492616	0	0	147
23	MWTP23	6918	44678	66468	270342	816394	6863	4875	137
24	MWTP24	3428	38923	51862	175384	283686	14227	10695	137
25	MWTP25	8243	28211	44262	228122	888408	22398	5991	127
26	MWTP26	5019	14159	48428	99085	288829	11817	6377	123
27	MWTP27	7845	-11931	32983	289613	734548	14195	5320	119
28	MWTP28	8830	20249	54268	232876	640725	8687	2907	118
29	MWTP29	5772	15388	39353	72393	388307	7843	2778	114
30	MWTP30	6192	48960	48282	196287	329887	6614	6526	97
31	MWTP31	5677	14500	31981	194544	576700	24625	10812	94
32	MWTP32	3700	7400	22600	125356	341800	10698	6531	94
33	MWTP33	6629	54793	44600	155000	561000	4750	4400	93
34	MWTP34	2400	11144	17201	43639	305500	81343	36772	92
35	MWTP35	5609	37159	38979	160420	402368	6483	4552	90
36	MWTP36	6328	38953	41745	192057	605354	8694	4257	90
37	MWTP37	5983	13481	44612	133801	497019	677	498	89
38	MWTP38	4803	35097	36700	158630	405010	9862	5375	84
39	MWTP39	2890	9460	20782	88767	298412	6720	1919	46
40	MWTP40	3834	2536	18413	85128	228882	25873	8624	42
41	MWTP41	1995	-640	9499	78900	209376	14460	4536	38
42	MWTP42	1160	-520	1580	22479	80200	0	0	34
43	MWTP43	1766	4656	11457	28903	126642	6430	1738	34
44	MWTP44	1329	10789	11203	33514	98140	24539	25448	31
45	MWTP45	2126	11787	13546	117711	297678	9431	7964	28
46	MWTP46	1735	7700	15100	69700	187480	0	0	28
47	MWTP47	368	1690	2761	7846	24549	13377	11848	27
48	MWTP48	1390	4200	13000	48000	153500	7784	2558	25
49	MWTP49	1879	13650	14470	64778	25911	10241	10394	25
50	MWTP50	1363	6285	11341	26597	136370	15656	4175	25
51	MWTP51	1413	7709	13486	41676	147529	10841	7943	23

52	MWTP52	1594	2435	6021	35688	105914	16882	4855	22
53	MWTP53	996	972	-2640	22850	91016	0	0	22
54	MWTP54	1367	4030	7916	37366	153609	9889	4215	21
55	MWTP55	1029	292	4292	18929	81224	18846	9231	21
56	MWTP56	1332	4064	9314	-64542	77018	141791	38060	21
57	MWTP57	1674	3437	5859	44545	142211	0	0	20
58	MWTP58	509	-803	2038	9566	73839	0	0	20
59	MWTP59	2719	5739	10682	13431	249352	0	0	20
60	MWTP60	747	1324	5883	19951	65924	15993	3736	19
61	MWTP61	1046	3571	7153	39189	102471	27227	7359	18
62	MWTP62	-80	-1471	4972	25921	80298	0	0	18
63	MWTP63	817	717	6181	14940	84633	0	0	17

***Table 3**: Average OPEX of 91 MWTPs operating within the Baltic catchment area and discharging effluents in the Baltic Sea*

Input variables & treated effluent (After initial, manual cleaning, 91 WTP) SEK/YEAR							
OPEX Parameters	**Total input cost (kr)**	**Energy (Kr)**	**Staff (Kr)**	**Reagents (Kr)**	**Maintenance (Kr)**	**Waste Management (Kr)**	**Treated effluent (m^3/year)**
Mean	22 881 406	1 961 680	10 055 170	1 841 105	5 799 579	3 223 873	6 371 819
SD	55 412 592	3 652 070	24 281 929	6 109 845	18 024 411	9 526 707	17 571 144
Min	338 700	0.00	0.00	0.00	0.00	0.00	22 342
Max	378 016 663	25 904 962	141 755 271	47 982 875	154 910 046	75 248 010	105 785 000
Total	2 082 207 910	178 512 854	915 020 483	167 540 511	527 761 652	293 372 409	579 835 558

Table 4: Average load of contaminants from 91 MWTPs used for model optimization

Output variables (After initial, manual cleaning, 91 WTP) KG/YEAR								
Pollutant	**P**	**N**	**BOD**	**COD**	**TOC**	**Zn**	**Cu**	**Mn**
Mean	33 914	238 081	514 381	1 645 418	2 275 041	14 966	7 506	515
SD	97 440	737 088	2 975 520	6 740 486	6 286 788	23 812	9 962	1 341
Min	0	292	1 557	7 639	2 919	0	0	17
Max	615 001	5 288 972	28 066 653	59 172 815	44 214 243	161 960	69 990	7 126
Total	3 086 201	21 665 362	46 808 676	149 732 995	207 028 724	1 361 900	683 014	46 883

Table 5: Average OPEX of contaminants from 82 MWTPs used for model optimization

Used for model optimization (82 WTP) SEK/YEAR							
OPEX Parameter	**Total input cost (kr)**	**Energy (Kr)**	**Staff (Kr)**	**Reagents (Kr)**	**Maintenance (Kr)**	**Waste Management (Kr)**	**Treated effluent (m^3)**
Mean	8 575 837	1 087 055	3 660 503	563 471	2 165 719	1 099 088	2 106 821
SD	10 697 2907	1 243 503	5 476 945	1 160 697	2 957 440	1 887 347	2 969 987
Min	338 7007	0.00	0.00	0.00	0.00	0.00	22 342
Max	56 331 0007	7 024 002	27 283 996	5 141 002	14 635 003	12 347 002	14 807 299
Total	703 218 603	89 138 543	300 161 268	46 204 606	177 588 940	90 125 246	172 759 307

Table 6: Average cost of contaminant removal from 82 and 78 MWTPs used for model optimization.

	Used for model optimization (82 WTP) KG/YEAR					Used for model optimization (78 WTP) Kg/year		
Pollutant	**P**	**N**	**BOD**	**COD**	**TOC**	**Zn**	**Cu**	**Mn**
Mean	10 603	56 960	77 377	354 529	961 656	11 224	5 716	176
SD	16 942	103 632	117 231	539 128	1 370 836	9 706	4 382	227
Min	0	292	1 580	7 846	24 549	0	0	17.39
Max	81 462	641 856	634 593	2 518 994	7 190 000	60 881	25 448	1 177
Total	869 433	4 670 733	6 344 948	29 071 404	78 855 828	875 459	445 836	13 700

***Table** 7: Average shadow prices from other studies compared with current study*

	Study 1	Study 2	Study 3	Study 4	Study 5
Ref. water price		0.1	0.991		
P	-9.0531	-7.533	-82.433	/	1.805
N	-328.7708	-4.612	-31.942	/	4.99
COD	-10.5485	-0.010	-2.277	/	/
BOD	-0.1013	-0.005	/	/	/
TSS	/	-0.011	-10.706	/	/
Zn	-42655	/	/	226	/
Cu	-29067	/	/	2,377	3.74
Mn	-2760	/	/	/	/

Study 1: Current study (€/m^3)
Study 2: Hernandez-Sancho et al. (2010) (€/m^3)
Study 3: Antalova et al. (2000) (€/m^3)
Study 4: Shadow price handbook CE Delft 2000 (€$_{2008}$/kg emission)
Study 5: CE Delft handbook 2024 (€$_{2021}$/kg)

7.3 - APPENDIX 3

Environmental Benefit ($EB_{Pollution}$) Models

Table 1: Environmental Benefit of wastewater treatment.

Enbart rening av metallerna skulle ge:

Environmental Benefit Summary (Table 4)

	Pollutant	Pollutant removal (kg/year)	Environmental value pollution (kr/year)	Environmental value pollution (kr/m^3)	%
0	Zn	1,353,436	57,731,185,381	99.667	73.3
1	Cu	678,686	19,727,668,714	34.058	25.1
2	Mn	46,852	1,290,322,078	2.228	1.6
3	Total	2,078,974	78,749,176,173	135.953	100.0

Environmental Benefit Summary (Table 4)

	Pollutant	Pollutant removal (kg/year)	Environmental value pollution (kr/year)	Environmental value pollution (kr/m^3)	%
0	P	857,214	68,837,937,535	404.954	12.8
1	N	4,612,868	464,199,886,257	2730.758	86.1
2	Zn	875,459	1,552,397,887	9.132	0.3
3	Cu	445,836	4,380,641,573	25.770	0.8
4	Mn	13,700	51,341,375	0.302	0.0
5	Total	6,805,077	539,022,204,628	3170.916	100.0

Och det ger Env.Benefit:

Environmental Benefit Summary (Table 4)

	Pollutant	Pollutant removal (kg/year)	Environmental value pollution (kr/year)	Environmental value pollution (kr/m^3)	%
0	P	869,433	7,871,025	0.046	0.4
1	N	4,670,733	1,535,600,592	8.889	83.0
2	BOD	6,344,948	642,702	0.004	0.0
3	COD	29,071,405	306,658,322	1.775	16.6
4	Total	40,956,518	1,850,772,641	10.713	100.0

***Table 2**: Environmental Benefit of GW Treatment for the modelled scenarios*

Pollutant	M_i (kg/yr)	SP_i (SEK/kg)	RE_{AWTP}	RE_{PWTP}	RE_{MWTP}	SC1	SC2	SC3	SC4	SC5A	$SC5B_1$	$SC5B_2$	$SC5B_3$	$SC5B_4$	$SC5B_5$
Pb	0,12	84,99	55%	72%	81%	0	5,6	8,3	9,7	7,3	10,2	7,2	4,8	5,5	3,7
Cd	0,005	198,37	78%	77%	82%	0	0,8	0,8	1,0	0,8	1,0	0,7	0,5	0,6	0,4
Cu	5,11	29067	70%	74%	81%	0	103972,7	120311,2	141194,9	109914,0	148532,4	105896,2	70597,4	82435,5	54957,0
Cr	0,16	0,66	84%	81%	71%	0	0,1	0,1	0,1	0,1	0,1	0,1	0,0	0,1	0,0
Ni	0,41	102,77	81%	58%	18%	0	34,1	7,6	27,6	24,4	42,1	20,7	13,8	18,3	12,2
Zn	10,7	42655	52%	82%	85%	0	237332,4	387947,2	444085,5	374255,0	456408,5	333064,1	222042,7	280691,2	187127,5
Mn	2,11	2760	30%	50%	70%	0	603,48	622,74	636,6072	462,24	642	477,4554	318,3036	346,68	231,12
TSS	5350	0,011	94%	72%	97%	0	603,5	622,7	636,6	462,2	642,0	477,5	318,3	346,7	231,1
BOD_5	17468	0,1013	80%	90%	99%	0	1415,6	1751,8	1767,7	1592,6	1769,5	1325,8	883,9	1194,4	796,3
COD	35989	10,5484	85%	75%	93%	0	322682,4	353052,5	372982,9	284719,8	379626,4	279737,2	186491,5	213539,8	142359,9
P_{TOT}	507	9,0531	90%	72%	95%	0	4130,9	4360,4	4525,7	3304,7	4589,9	3394,2	2262,8	2478,6	1652,4
N_{TOT}	693	328,7708	76%	43%	82%	0	173157,0	186827,3	204462,0	97970,4	227838,2	153346,5	102231,0	73477,8	48985,2
PET	3,49	241,5	80%	99,45%	90%	0	674,3	758,6	842,4	838,2	842,8	631,8	421,2	628,6	419,1
PP	0,63	241,5	80%	99,45%	90%	0	121,7	136,9	152,1	151,3	152,1	114,0	76,0	113,5	75,7
TOTAL EB $_{pollution}$						0	845,878	1,059,862	1,175,638	876,153	1,226,279	881,729	587,819	657,114	438,076

7.4 - APPENDIX 4

Schematic diagram of “Cradle-to-Grave” greywater treatment processes in Trelleborg