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Managing coastal eutrophication

Land-sea and hydroclimatic linkages with focus on the Baltic coastal system

Guillaume Vigouroux

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Abstract

Eutrophication endangers coastal ecosystems all over the world and is most often associated with an increase in anthropogenic nutrient loads to coastal waters, which fuel the growth of algae and create a variety of environmental problems. This is also the case for the Baltic Sea where coastal waters may be affected by various land, coast-sea, and hydroclimatic drivers and feedbacks, over different scales, including the eutrophic open sea. This thesis aims at improving our understanding of how these drivers affect coastal eutrophication and its management opportunities across the various coupled scales of the Baltic land-coast-sea system. To achieve this aim, the interactions between land-catchment, coastal, and open sea processes, and their influences on coastal eutrophication have been investigated through water quality modelling with applications to specific Baltic coastal waters. Hydroclimatic influences on the propagation of change-impacts through the land-coast-sea continuum to coastal eutrophication have also been investigated via the water quality modelling and additional analysis of actual water quality trends over the last 30 years along the Swedish coast. Moreover, coastal eutrophication research on the Baltic Sea system has been investigated through scientific literature analysis with focus on how the reported research has accounted for and linked components in the land-coast-sea system, and the aim to identify possible research gaps.

Results show that impacts of water quality improvements in the open sea propagate to a large share of the coastal waters, especially for phosphorus and phytoplankton, while impacts of reducing nutrient loads from land are more localised and more pronounced for nitrogen than for phosphorus. Therefore, reducing coastal nitrogen, phosphorus and phytoplankton concentrations requires both regional measures for open sea improvements and local land-catchment measures for reduction of nutrient loads to the specific coast. Moreover, data analysis shows that trends in coastal Summer chlorophyll a (Chl-a) are well correlated with those in open sea Summer Chl-a and in riverine nitrogen loads. Regarding hydroclimatic drivers, warmer and wetter conditions are found to complicate remediation of coastal eutrophication in comparison to drier and colder conditions. In addition, trends in coastal Summer Chl-a are well correlated with those in sea-ice conditions. These results highlight the various land-based, coastal, open sea, and hydroclimatic drivers and conditions that mix, interact in and influence the coastal waters. The various driver, management, and ecosystem components involved are overall included in Baltic coastal eutrophication research. However, specific coastal management measures, and feedbacks between drivers and impacts of coastal eutrophication are under-investigated, and the social and ecological components of the whole land-coast-sea system are not well-connected in the research.

Furthermore, long-lived legacy sources on land, as well as at sea, have not been much accounted for in coastal eutrophication research so far. This calls for further research on recovery time scales and specific remediation measures that can be effective against such sources, like mussel farming and wetlands. Finally, coastal eutrophication management needs to account for the influences on local coastal conditions from a melting pot of multi-scale drivers and biogeochemical as well as ecological impacts and feedbacks.

Keywords: *coastal eutrophication, land-coast-sea continuum, management, hydroclimatic change, research gaps, eutrophication modelling, temoporal trends, scoping review, Baltic Sea.*

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Eutrophication endangers coastal ecosystems all over the world and is most often associated with an increase in anthropogenic nutrient loads to coastal waters, which fuel the growth of algae and create a variety of environmental problems. This is also the case for the Baltic Sea where coastal waters may be affected by various land, coast-sea, and hydroclimatic drivers and feedbacks, over different scales, including the eutrophic open sea. This thesis aims at improving our understanding of how these drivers affect coastal eutrophication and its management opportunities across the various coupled scales of the Baltic land-coast-sea system. To achieve this aim, the interactions between land-catchment, coastal, and open sea processes, and their influences on coastal eutrophication have been investigated through water quality modelling with applications to specific Baltic coastal waters. Hydroclimatic influences on the propagation of change-impacts through the land-coast-sea continuum to coastal eutrophication have also been investigated via the water quality modelling and additional analysis of actual water quality trends over the last 30 years along the Swedish coast. Moreover, coastal eutrophication research on the Baltic Sea system has been investigated through scientific literature analysis with focus on how the reported research has accounted for and linked components in the land-coast-sea system, and the aim to identify possible research gaps.

Results show that impacts of water quality improvements in the open sea propagate to a large share of the coastal waters, especially for phosphorus and phytoplankton, while impacts of reducing nutrient loads from land are more localised and more pronounced for nitrogen than for phosphorus. Therefore, reducing coastal nitrogen, phosphorus and phytoplankton concentrations requires both regional measures for open sea improvements and local land-catchment measures for reduction of nutrient loads to the specific coast. Moreover, data analysis shows that trends in coastal Summer chlorophyll a (Chl-a) are well correlated with those in open sea Summer Chl-a and in riverine nitrogen loads. Regarding hydroclimatic drivers, warmer and wetter conditions are found to complicate remediation of coastal eutrophication in comparison to drier and colder conditions. In addition, trends in coastal Summer Chl-a are well correlated with those in sea-ice conditions. These results highlight the various land-based, coastal, open sea, and hydroclimatic drivers and conditions that mix, interact in and influence the coastal waters. The various driver, management, and ecosystem components involved are overall included in Baltic coastal eutrophication research. However, specific coastal management measures, and feedbacks between drivers and impacts of coastal eutrophication are under-investigated, and the social and ecological components of the whole land-coast-sea system are not well-connected in the research.

Furthermore, long-lived legacy sources on land, as well as at sea, have not been much accounted for in coastal eutrophication research so far. This calls for further research on recovery time scales and specific remediation measures that can be effective against such sources, like mussel farming and wetlands. Finally, coastal eutrophication management needs to account for the influences on local coastal conditions from a melting pot of multi-scale drivers and biogeochemical as well as ecological impacts and feedbacks.

Sammanfattning

Övergödning hotar kustekosystem över hela världen och är oftast förknippad med en ökning av näringsämnen från mänskliga aktiviteter till kustvattnet, som ger upphov till tillväxt av alger och en rad relaterade kustmiljöproblem, som syrebrist och förändringar i näringskedjan. Detta gäller även för Östersjön, ett halvslutet hav i norra Europa, där olika drivkrafter och återkopplingar på olika skalor, från land, kust, hav och klimatet, kan påverka kustvattnet. Syftet med denna avhandling är att öka vår förståelse av hur de olika drivkrafterna påverkar kustområdenas övergödning, samt hur drivkrafterna kan hanteras och reduceras för att förbättra vattenkvalitet och ekologisk status i Östersjöns sammanhängande land-kust-hav-system. För att uppnå detta undersöks i avhandlingen samspelet mellan processer på land, i själva kusten och i det öppna havet, samt deras kombinerade påverkan på kustområdenas övergödning. Undersökningen omfattar modellering av kustvattenkvalitet med tillämpning på specifika kustområden i Östersjön. Den omfattar också studier av hur klimat- och hydrologiska förändringar sprids och leder till relaterade förändringar i vattenkvalitet och övergödning genom det sammanhängande land-kust-hav-systemet. Förändringsspridningen har undersökts genom både modellering av ett specifikt kustområde och databaserad analys av faktiska förändringstrender i vattenkvalitet längs hela den svenska kusten under de senaste 30 åren. För att identifiera möjliga luckor i forskning och kunskap om övergödning av kustvatten har också den relaterade vetenskapliga litteraturen analyserats med fokus på Östersjöns kustområden och på hur olika komponenter i det komplexa sammanhängande land-kust-hav-systemet hanteras och kopplas ihop i forskningen.

Resultaten visar att förbättringar av vattenkvaliteten i det öppna havet har betydande effekter på kustvattnets kvalitet, i synnerhet när det gäller fosfor och växtplankton. Minskad näringsbelastning från land ger mer lokala effekter, som även är mer uttalade för kväve än för fosfor. För att minska koncentrationerna av kväve, fosfor och växtplankton i kustvatten krävs därför både storskaliga regionala åtgärder för att förbättra vattenkvaliteten och minska övergödningen i det öppna havet och mer småskaliga åtgärder lokalt i det landområde som dränerar in till varje kust och i själva kusten för att minska näringsbelastningen till det specifika kustvattnet. Dessutom visar dataanalysen av faktiska trender att förändringar av klorofyll a i kustvatten under sommartid är väl korrelerade med motsvarande förändringar i klorofyll a i det öppna havet samt i kvävebelastningen från vattendragen på land. För klimat och hydrologi som drivkrafter för kustförändringar visar resultaten att varmare och fuktigare förhållanden försvårar reducering av övergödning och uppnående av relaterade miljömål för kustområden jämfört med torrare och kallare förhållanden. Vidare är trenderna i klorofyll a vid kusten under sommartid väl korrelerade med trenderna i havsisförhållanden. Sammantaget visar resultaten i denna avhandling att flera olika drivkrafter på land och i själva kusten såväl som i det öppna havet, samt i klimat och hydrologi, blandas, samspelar och påverkar varje kustområdes vattenkvalitet och ekosystemstatus. De olika inblandade drivkrafterna och deras påverkan på kustekosystemen samt hantering i och för kustmiljöförvaltning tas upp i forskningen om övergödning i Östersjöns kustområden. Specifika åtgärder för kustmiljöförvaltning, samt återkopplingar mellan ekosystemeffekter och drivkrafter i kustområdenas övergödning är dock relativt svagt utforskade. De mänskliga och ekologiska komponenterna i det sammanhängande land-kust-havs-systemet är inte heller väl sammankopplade i forskningen.

Vidare har inte forskningen om Östersjökusternas övergödning i någon större utsträckning beaktat kvardröjande ärvda källor av näringsämnen på land och i havet, som visat sig väsentligt bidra till näringsbelastningen vid kusterna. Ytterligare forskning behövs om tidsskalor för återhämtning vid olika åtgärder mot kusternas övergödning, inklusive sådana som musselodling och våtmarker, som kan vara effektiva mot ärvda källor. Slutligen behöver kustmiljöförvaltningen och relaterad forskning också ta hänsyn till hela blandningen av effekter och återkopplingar från biogeokemi, ekologi, hydrologi och klimat på olika skalor som påverkar lokala kustförhållanden. Det innebär att för de lokala kustförhållandena också ta hänsyn till påverkan från de storskaliga regionala förhållandena i öppna havet och inte bara till de lokala förhållandena vid kusten samt på land och i de inlandsvatten som dränerar in till varje kustområde. Vidare behöver hänsyn också tas till de möjliga synergier som kan finnas och de avvägningar som kan behöva göras mellan åtgärder för minskad kustövergödning i olika delar av Östersjösystemet och på olika skalor upp till hela regionala systemet.

Dissertation content

This doctoral compilation dissertation consists of a summarising text and the four articles listed below (I–IV). The papers are referred to as Papers I to IV in the summary text, and are appended to the end of the thesis and reprinted with permission from the respective copyright holders:

I Vigouroux, G., Destouni, G., Jönsson, A. and Cvetkovic, V., 2019. A scalable dynamic characterisation approach for water quality management in semi-enclosed seas and archipelagos. *Marine Pollution Bulletin*, 139, pp.311-327. doi: 10.1016/j.marpolbul.2018.12.021

Supplementary Material for Paper I, DOI: 10.13140/RG.2.2.26669.82403

II Vigouroux, G., Chen, Y., Jönsson, A., Cvetkovic, V. and Destouni, G., 2020. Simulation of nutrient management and hydroclimatic effects on coastal water quality and ecological status – The Baltic Himmerfjärden Bay case. *Ocean & Coastal Management*, 198, p.105360. doi: 10.1016/j.ocecoaman.2020.105360

Supplementary Material for Paper II, DOI: 10.13140/RG.2.2.33380.71047

III Vigouroux, G., Kari, E., Beltrán-Abaunza, J.M., Uotila, P., Yuan, D. and Destouni, G., 2021. Trend correlations for coastal eutrophication and its main local and whole-sea drivers – Application to the Baltic Sea. *Science of the Total Environment*, 779, p.146367. doi: 10.1016/j.scitotenv.2021.146367

Supplementary Material for Paper III, DOI: 10.13140/RG.2.2.16603.49449

IV Vigouroux, G. and Destouni, G., 2021. Gap identification in coastal eutrophication research – The Baltic system case. *[Manuscript]*

Supplementary Material for Paper IV

Author contributions

The contributions from listed authors are divided as follows for each article.

I GV led the writing and compiled the datasets. GV led the implementation of the water quality model and the water quality simulations, with help from AJ. The study and related methods were designed by all co-authors. The writing was assisted by all co-authors.

II GV led the writing. GV led the Himmerfjärden hydrodynamic model set up, data processing and simulations with help from YC and AJ. GV led the Himmerfjärden Bay water quality model set up, data processing and simulations with help from AJ. GV led the result analysis with help from GD and YC. YC led the Baltic Sea hydrodynamic model set up and simulations. The study and related methods were designed by GV, GD and VC. The writing was assisted by all co-authors.

III GV led the writing, compiled the datasets, did the data analysis and led the data curation with help from JMBA. The study and related methods were designed by GV, GD and EK. The writing was assisted by all co-authors.

IV GV led the writing and results analysis, assisted by GD, and reviewed the literature. The study and related methods were designed by GV and GD.

Contents

Abbreviations

r^2	Pearson coefficient of determination
BSAP	Baltic Sea Action Plan
C	carbon
CB	Coastal Basin
Chl-a	Chlorophyll a
CRU TS	Climate Research Unit Time Series
DIN	dissolved inorganic nitrogen: ammonium, nitrite and nitrate
DIP	dissolved inorganic phosphorus: phosphate
EcS	Ecological Status
EU	European Union
FMI	Finnish Meteorological Institute
FVCOM	Finite Volume Community Ocean Model
GCS	Good Component Status
GEcS	Good Ecological Status
GEMSS®	Generalized Environmental Modelling System for Surface waters
GEnS	Good Environmental Status
HELCOM	Helsinki Commission
HYPE	Hydrological Predictions for the Environment
MAI	Maximum Allowable Inputs
MB	Marine Basin
MBI	Major Baltic Inflow
MSFD	Marine Strategy Framework Directive
MVM	Soil, Water, Environment database
N	nitrogen
ND	Nitrate Directive
nm	nautical mile
P	phosphorus
PLC	Pollution Load Compilation
SLU	Swedish University of Agricultural Sciences
SM	Supplementary Material
SMHI	Swedish Meteorological and Hydrological Institute
Summer Chl-a	surface concentration of Chl-a during the summer period
Summer TN	surface concentration of TN during the summer period
Summer TP	surface concentration of TP during the summer period
TN	total nitrogen
TP	total phosphorus
UWWTD	Urban Wastewater Treatment Directive
WFD	Water Framework Directive
Winter DIN	surface concentration of DIN during the winter period
Winter DIP	surface concentration of DIP during the winter period
Winter TN	surface concentration of TN during the winter period
Winter TN:TP	ratio between Winter TN and Winter TP
Winter TP	surface concentration of TP during the winter period
WOS	Web of Science™
WWTP	wastewater treatment plant

1 Introduction

1.1 Marine and coastal eutrophication in the world

Eutrophication is a major issue worldwide, which affects more than 900 coastal and marine areas (Figure 1.1, Panel A top right; Diaz et al., 2019) as well as freshwater rivers and lakes (Smith, 2003; Smith and Schindler, 2009). Eutrophication is generally associated with an increase in nutrient concentrations, especially phosphorus (P) and nitrogen (N), in freshwater and marine systems that leads to enhanced primary production by algae and plants. In contrast to many other contaminants, nutrients are essential elements for life in any ecosystem, and increased nutrient concentrations can be associated with higher fish biomass (Nixon and Buckley, 2002). However, in too high quantities, nutrients can also negatively impact aquatic systems by changing species distribution and habitats (e.g., decline in salt marshes; Deegan et al., 2012), and depleting bottom oxygen concentrations, leading to hypoxia that can kill benthic fauna and fish and create so-called dead zones (Diaz and Rosenberg, 2008). Eutrophication is strongly associated with human developments that increase nutrient loads to freshwater and marine systems, and affects primarily industrialized catchments and water systems in the vicinity of population centres (Rabalais et al., 2010; Diaz and Rosenberg, 2008). Thereby, this issue has been widely reported in Europe and North America, and is now also expanding in Asia and Africa (Rabalais et al., 2010; Lee et al., 2016).

Eutrophication has been defined by Nixon (1995) as "an increase in the rate of supply of organic matter to an ecosystem". While this definition stresses that eutrophication is a process rather than a state (Ferreira et al., 2011), a narrower definition is generally used for management and legal purposes (Ferreira et al., 2011) to emphasize the role of anthropogenic nutrient loads and their undesirable ecosystem impacts (e.g., in OSPAR, 2003; European Commission, 1991b). In this thesis, the term eutrophication refers more generally to the biogeochemical and ecological changes in a water system resulting both directly and indirectly from increased anthropogenic nutrient loading.

In marine systems, eutrophication is often exacerbated in the coastal zones, which are the interfaces through which large amounts of energy and matter are transferred between land and sea (Elliff and Kikuchi, 2015). The coastal zones also provide a variety of provisioning (e.g., food provision), supporting (e.g., waste processing), and cultural (e.g., recreation and tourism) ecosystem services (Elliff and Kikuchi, 2015), and are inhabited by a large share of the human population (Neumann et al., 2015). Each coastal zone receives nutrient loads from its whole contributing land-catchment area, in addition to direct nutrient discharges from coastal cities and industries. Coastal areas may also be impacted by other direct and indirect human pressures, such as pollution, overfishing, and climate change (Malone and Newton, 2020, and references therein). The combined pressures may make coastal zones particularly vulnerable to eutrophication, which also to various degrees interacts with the open sea eutrophication, especially in enclosed and semi-enclosed seas such as in the Gulf of Mexico and the Baltic Sea (Diaz et al., 2019).

1.2 Eutrophication development and management in the Baltic Sea

In the Baltic Sea, a semi-enclosed brackish sea situated in northern Europe, the first symptoms of coastal eutrophication appeared in the 1950s in the proximity of large cities due to their nutrient discharges that promoted phytoplankton blooms (Elmgren, 2001). Eutrophication progressed during the next decade owing to the industrialisation of agricultural practices and further increase of nutrient loads (Voss et al., 2011). In addition, due to a slow water exchange with the North Sea, nutrient concentrations and eutrophication increased dramatically in the open Baltic Sea during the 1970s (Elmgren, 2001).

Eutrophication has led to substantial ecosystem impacts in the coastal and marine waters of the Baltic Sea. During the last three decades, naturally occurring hypoxic zones have increased from around 1 % of the Baltic Sea area in 1993 to 19 % in 2016 (Jokinen et al., 2018; Conley et al., 2009b), to form the largest anthropogenically driven hypoxic zone worldwide (Diaz and Rosenberg, 2008). Potentially toxic cyanobacteria blooms, which can settle in coastal areas and reduce their recreational values, have also increased (Kahru and Elmgren, 2014). Valuable macroalgae covers have been reduced dramatically due to decreasing water clarity and increasing cover of filamentous algae (e.g., 75–80 % reduction in eelgrass cover since 1900; Malone and Newton, 2020; HELCOM, 2009). Eutrophication in synergy with other pressures has also led to Baltic-wide ecosystem and food web changes. A collapse of the cod population, which is a highly valued key predator in the Baltic food web, is one of the most notable impacts. This has shifted the food web from a cod to a pelagic fish dominance due to overfishing in combination with eutrophication-driven deterioration of spawning habitats and decadal climatic changes (Moellmann et al., 2009; Alheit et al., 2005).

In reaction to the growing environmental impacts of eutrophication, the Convention on the Marine Environment of the Baltic Sea Area (HELCOM, 1974) was signed by the coastal states in 1974 (Finland, Sweden, Denmark, Poland, Germany, Latvia, Lithuania, Estonia, Russia and the European Community; updated in 1992) to combat eutrophication and protect the marine environment, and the Helsinki Commission (HELCOM) was created. HELCOM recognizes the need for collaboration between the coastal countries and is the main commission for protecting the marine environment (Kern, 2011). However, despite load reduction targets set in 1988 (Boesch, 2019), eutrophication and pollution have remained problematic, leading to the creation of the HELCOM Baltic Sea Action Plan (BSAP) in 2007 (HELCOM, 2007). The BSAP aims to achieve Good Status in the Baltic Sea in terms of eutrophication, biodiversity, hazardous substances and marine activities (HELCOM, 2007) by using an ecosystem approach that integrates land, water, and "living resources" management (Backer et al., 2010). To reach the eutrophication objectives of clear water, natural level of algae blooms, natural distribution and occurrence of plants and animals, and natural oxygen levels (HELCOM, 2007) the BSAP has set Maximum Allowable Inputs (MAI) of nutrients for each country using water quality modelling for the Baltic Sea and its land-catchment (Boesch, 2019).

Following and adding to the HELCOM conventions, several directives have also been put in place by the European Union (EU) that require EU member states to reduce human pressures to and eutrophication of coastal and open sea European waters (Ferreira et al., 2011). The Urban Wastewater Treatment Directive (UWWTD, European Commission, 1991b) and the Nitrate Directive (ND, European Commission, 1991a) have both come into force in 1991 to counteract the negative effects of urban wastewater discharges and promote good agricultural practices that protect surface and ground waters. The Water Framework Directive (WFD), put in place in 2000, is a more comprehensive river basin regulation that requires management plans for EU member states to achieve Good Ecological Status (GEcS) in their inland and coastal waters (the latter within 1 nm (nautical mile) from shore), initially by 2015, and then for every 6-year management cycle (European Commission, 2000; Borja et al., 2010). The WFD uses a "deconstructing structural approach" (Borja et al., 2010) that separates the ecosystem into five main biological elements, as well as hydro-morphological and physico-chemical conditions. The Marine Strategy Framework Directive (MSFD), put in place in 2008 (European Commission, 2008), requires achievement of Good Environmental Status (GEnS) for the European seas from the coastline until 200 nm, initially by 2020, and for every 6-year management cycle thereafter. In contrast to the WFD, the MSFD uses a "holistic functional approach" that aims to represent the functions of the whole ecosystem via a set of 11 process-based descriptors, including human-induced eutrophication (Borja et al., 2010).

The management programmes and frameworks put in place by HELCOM and the EU have promoted science-based management (Wulff et al., 2007) and made the Baltic Sea one of the best monitored sea in the world (Reusch et al., 2018). Management has also successfully reversed trends in nutrient loads, which increased strongly between 1950 and 1980, peaked in 1990, decreased until 2005 by 24 % and 50 % of their peak value for N and P, respectively, and have since then plateaued (Gustafsson et al., 2012; Reusch et al., 2018). The decreases can be mostly attributed to improved wastewater treatment, leading to 50 % and 70 % decline in point source loads of N and P, respectively, between 1985 and 1995, following the UWWTD implementation (Savchuk et al., 2012). Despite these substantial nutrient load reductions after the 1990s, a further 16 % and 38 % reductions are still needed for N and P, respectively, to meet the BSAP targets (Jetoo, 2019). Moreover, agricultural loads, accounting for two thirds of the diffuse sources, have only marginally decreased since the 1980s (Reusch et al., 2018). Together with the current plateau in nutrient loads, this implies that further reduction is needed and will come at higher cost (Reusch et al., 2018), while eutrophication is still an issue in the Baltic Sea and its coastal waters (Fleming-Lehtinen et al., 2015; HELCOM, 2018a).

1.3 Problem description

The likely higher costs associated with further nutrient reductions stress the need for cost-efficient measures to reduce coastal eutrophication. Cost efficiency may not only be influenced by land-based processes, but also by complex interactions between the local land-catchment, local coastal water, and regional open sea conditions. For example, open coastal waters may interact to relatively high degree with the open sea and with other coastal waters, in addition to also being affected by land-based nutrient loads (Almroth-Rosell et al., 2016). Moreover, the various management frameworks focus on different scales, with the BSAP explicitly considering the land-catchment, coast and open sea waters, the MSFD considering the transition between the coastline and the open ocean (200 nm), and the WFD considering the land-catchment and the coastal waters (1 nm). On the coastal scale, the BSAP and MSFD spatially overlap with the WFD. The overlap can be substantial in archipelagoes (Borja et al., 2010) and create mismatching goals and conflicting results due to the different approaches in these two action and regulation programmes. The spatial scale in focus may also be at odds with the physical processes, as for example riverine nutrient plumes may affect open sea waters more than coastal waters, which are too turbid to sustain high primary production (Ferreira et al., 2011). Understanding the propagation of management effects through the whole land-coast-sea continuum is therefore necessary for identifying measures that can comply with the different and overlapping management frameworks.

While nutrient loads have been reduced since the 1990s, most open sea waters have still not reached the HELCOM BSAP goals for eutrophication (Fleming-Lehtinen et al., 2015; HELCOM, 2018a). For coastal waters, the situation is more mixed with recent improvements for example in the Stockholm Archipelago (Walve and Rolff, 2021), while most of the coastal waters still do not reach GEcS (HELCOM, 2018a), and eutrophication indicators even are worsening in other coastal waters (c.f. Paper III, Figure 2). Indeed, changes in drivers may impact differently on the land, coastal and open sea parts of a linked system. For example, the Baltic Sea experiences rates of climate change above the world average (Reusch et al., 2018; The BACC II Author Team, 2015), which may counteract achievement of nutrient reduction targets by increasing precipitation, and river discharges and their nutrient loads (Bring et al., 2015). Climate-driven changes in water temperature may also impact coastal and marine eutrophication directly, by increasing algal growth rate and nutrient recycling, as well as indirectly through stratification and hypoxia (Glibert et al., 2014; Meier et al., 2012). Moreover, by modifying the flow between coastal and open sea waters, hydroclimatic changes may also impact the propagation of management effects throughout the coastal system and increase its vulnerability to nutrient loads (Chen et al., 2019b). Thereby, implementation of effective management strategies and measures requires knowledge on the main land-based, hydroclimatic and hydrospheric drivers of coastal eutrophication and their interactions and change-impact propagation through the coupled components and scales of the Baltic system.

In addition to direct change drivers, associated ecosystem changes may also maintain and further accentuate eutrophication. For example, hypoxia enhances the P release from sediments, which in turn promotes N-fixing cyanobacteria, increasing N concentrations, primary production and hypoxia in the sea, and leading to a self-reinforcing vicious circle (Vahtera et al., 2007; Funkey et al., 2014). Moreover, ecosystem changes, such as loss of macroalgae, and anthropogenic pressures, such as overfishing, can also act in synergy with directly managed (e.g., nutrient loads) and indirectly managed (e.g., climate change) pressures to reinforce eutrophication, by e.g., changing the food web and reducing zooplankton grazing on phytoplankton (Malone and Newton, 2020; Eriksson et al., 2009). Understanding and quantifying these complex interlinkages of coastal eutrophication dynamics with ecological conditions is therefore necessary for managing the ecosystem impacts of eutrophication.

1.4 Aim and objectives

Figure 1.1 presents the Baltic Sea study system (Panel A) and the components and concepts forming the main aim of the thesis (Panel B). The thesis aim is to further our understanding of how the combined hydroclimatic, hydrospheric, and land-based drivers and feedbacks affect coastal eutrophication, and how their effects propagate through the scales involved in the coupled Baltic land-coast-sea system. This aim has been addressed by focusing on three main study objectives:

A. Investigate in a numerical modelling framework the influences of land-catchment, coastal, and open sea interactions and their associated scales on coastal eutrophication conditions and management opportunities. (**Papers I & II**; Vigouroux et al. (2019, 2020), circled in beige in Figure 1.1, Panel B).

B. Investigate how overarching hydroclimatic conditions interact with the land-coast-sea continuum in propagating change-impacts to coastal eutrophication and affecting its management (**Papers II & III**; Vigouroux et al. (2020, 2021), circled in orange in Figure 1.1, Panel B).

C. Investigate how the complexity of coastal eutrophication processes has been considered and handled in research on the Baltic Sea system, and identify possible key research gaps (**Paper IV**; Vigouroux and Destouni (2021), circled in dark orange in Figure 1.1, Panel B).

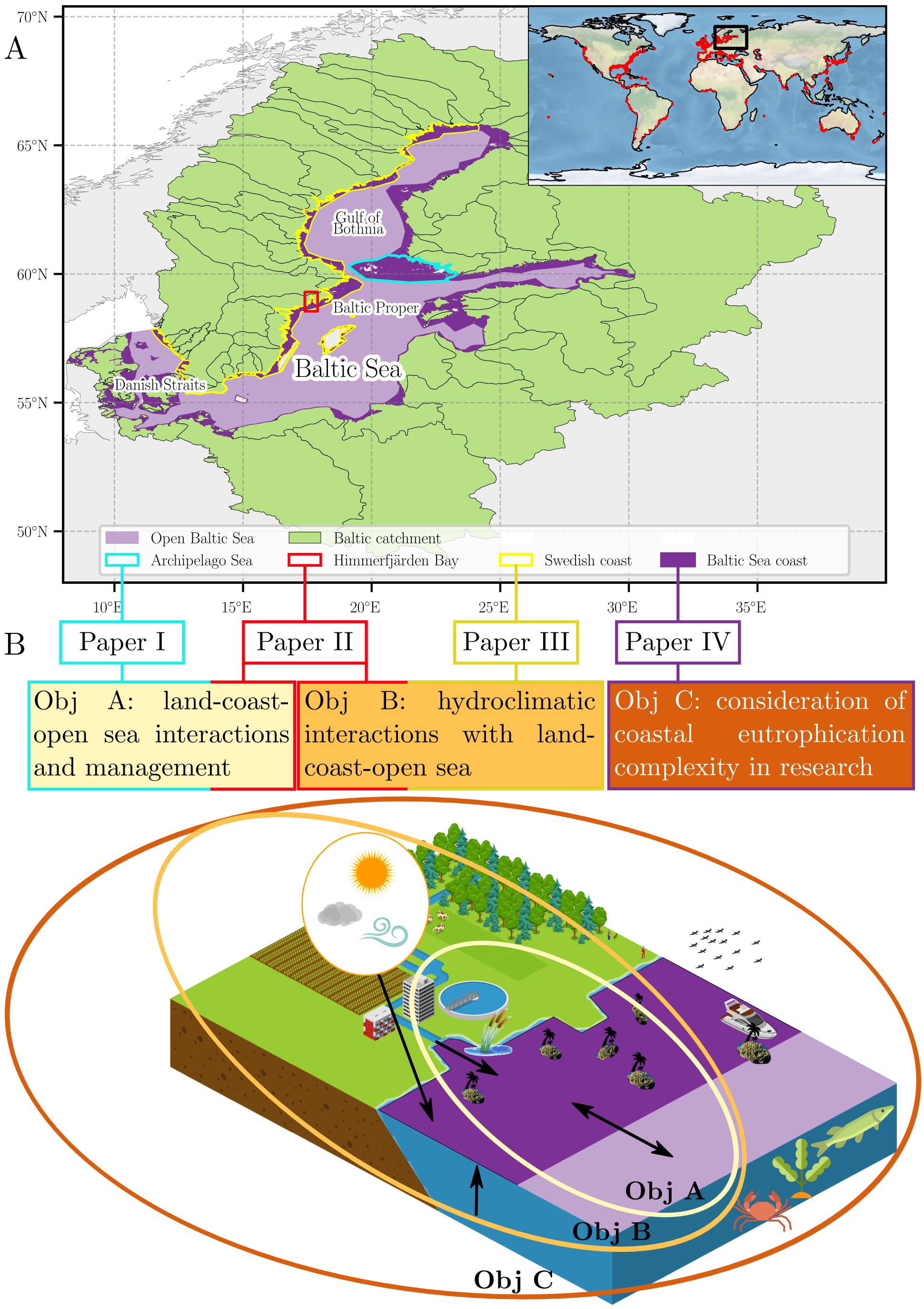

Figure 1.1. Panel A top right: Map of the coastal areas suffering from eutrophication (red dots; data from Diaz et al. (2011)), with the location of the Baltic Sea circled in black. Map of the Baltic Sea study system (Panel A), including the coastal waters (dark purple), the open sea waters (light purple), and the whole Baltic catchment area (light green), divided into local catchments (thin dark grey line). The study areas considered in the papers (cyan: Archipelago Sea, Paper I; red: Himmerfjärden Bay, Paper II; yellow: Swedish coast, Paper III; dark purple: whole Baltic coast, Paper IV) forming this thesis are linked to the study objectives encompassing the various components of the schematically represented study system (Panel B).

2 Study areas

2.1 Baltic Sea

The Baltic Sea (Figure 1.1, Panel A), situated in northern Europe, is one of the largest brackish bodies of water in the world (Leppäranta and Myrberg, 2009). The Baltic Ice Lake has formed recently, following the last ice age deglaciation, 16 000 years ago, and became brackish, close to its present shape, around 9800 years ago (Andrén et al., 2011). The Baltic Sea has a volume of $21\,200\,\mathrm{km}^3$ and an area of $393\,000\,\mathrm{km}^2$ (Leppäranta and Myrberg, 2009). It is relatively shallow (mean depth of $54\,\mathrm{m}$), but has a complex bathymetry (maximum depth of $459\,\mathrm{m}$ in the Baltic Proper; Figure 1.1, Panel A) and includes several marine basins separated by sills that have formed by glacier retreat (Leppäranta and Myrberg, 2009; Voss et al., 2011). The Baltic Sea is connected to the North Sea through the shallow Danish Straits (depth of $20\,\mathrm{m}$; Figure 1.1, Panel A). Owing to its young age and its brackish conditions, stressing both marine and freshwater organisms, the Baltic Sea is characterized by a low species diversity, which makes it particularly vulnerable to pressures and changes (Reusch et al., 2018).

The hydrological land catchment (in light green Figure 1.1) of the Baltic Sea has an area of $1\,745\,100\,\mathrm{km}^2$ (Schiewer, 2008a), approximately four times that of the sea. The Baltic Sea and its catchment experience an oceanic temperate climate in the south and humid sub-polar climate in the north (Kottek et al., 2006), with average precipitation ranging between $400\,\mathrm{mm{\cdot}year^{-1}}$ and $800\,\mathrm{mm{\cdot}year^{-1}}$ (Voss et al., 2011). The catchment area is inhabited by approximately 85 million people and shared between fourteen countries, of which nine border the Baltic Sea (HELCOM, 2018b). Almost $50\,\%$ of the total catchment area is covered by forest and $20\,\%$ is used for agriculture (Schiewer, 2008a). In contrast, $34\,\%$ of the catchment of the Baltic Proper marine basin is forested and $52\,\%$ is used for agriculture, related to and reflecting the higher population density in the south (Voss et al., 2011).

The Baltic Sea receives on average $480\,\mathrm{km^3{\cdot}year^{-1}}$ of riverine discharge, corresponding to $2.5\,\%$ of its volume (Bergström et al., 2001), of which $200\,\mathrm{km^3{\cdot}year^{-1}}$ flow into the Gulf of Bothnia, $100\,\mathrm{km^3{\cdot}year^{-1}}$ into the Gulf of Finland, and $180\,\mathrm{km^3{\cdot}year^{-1}}$ into the Baltic Proper and southern parts of the sea (Voss et al., 2011). The riverine discharge peaks between April and June and is lower between August and January (Voss et al., 2011). Riverine discharge is the main nutrient loading pathway, representing about $70\,\%$ of the total nitrogen (TN) loads and about $89\,\%$ of the total phosphorus (TP) ones. Airborne loads account for $27\,\%$ and $6\,\%$ of TN and TP, respectively, and direct point sources for $4\,\%$ and $5\,\%$ of TN and TP loads, respectively (HELCOM, 2018c). Natural loads represent one third of the riverine TN and TP loads, with a higher contribution in the northern parts of the catchment (HELCOM, 2018c). N-fixation by cyanobacteria is also an important input, estimated to about $400\,000\,\mathrm{t{\cdot}year^{-1}}$ in the Baltic Proper for the period 2013–2017, exceeding the total external N loads (riverine, point sources and atmospheric deposition; Olofsson et al., 2021). Nutrient loads are strongly dependent on the precipitation patterns, which are in turn coupled to the North Atlantic Oscillation that creates decadal and multi-decadal cycles in the Baltic Sea (Voss et al., 2011). Moreover, due to its northern location, the Baltic Sea has experienced a stronger rise in temperature than the global average, and precipitation is expected to increase with possible associated increases of riverine water discharges and nutrient loads (Reusch et al., 2018).

The Baltic Sea is a microtidal sea (average tidal amplitude of $15\,\mathrm{cm}$), so that changes in its water level, ranging between $3\,\mathrm{m}$ above to $2.5\,\mathrm{m}$ below sea level, are mostly driven by differences in air pressure and by wind (Schiewer, 2008a). Due to the large freshwater discharges and slow water exchange with the North Sea, the Baltic Sea has an important salinity gradient, from 0–$2\,\mathrm{PSU}$ in the north of the Gulf of Bothnia to 15–$18\,\mathrm{PSU}$ in the Danish Straits (Figure 1.1, Panel A; HELCOM, 2018b). In addition, the complex bathymetry of basins separated by sills hinders vertical mixing, which creates a strong stratification, with a permanent halocline (zone of maximum vertical salinity gradient, 40–$80\,\mathrm{m}$) over which forms a thermocline (zone of maximum temperature gradient) during the summer (Liblik and Lips, 2019). Thereby, there is little mixing between the sea's surface waters and deep saline waters. The latter are renewed through Major Baltic Inflows (MBIs), during which saline (17–$25\,\mathrm{PSU}$) water from the North Sea penetrates via the Danish Straits. The magnitude and duration (day-week) of these saltwater inflows, occurring mostly in winter, are governed by large scale atmospheric conditions and the salinity front at the Baltic Sea entrance (Voss et al., 2011). MBIs reoxygenate the deeper stagnant waters, while also reinforcing the halocline and thus decreasing vertical mixing (Voss et al., 2011; Carstensen et al., 2014). Water circulation in the Baltic Proper is counter-clockwise with closed streamlines (Voss et al., 2011), which separate the open sea and coastal dynamics, and transport riverine discharges alongshore. Sediments are then transported from the shallower northeastern and western parts towards the central part of the Baltic Proper where they accumulate (Schiewer, 2008a).

Due to its large catchment area, closed circulation and long residence time, anthropogenic nutrient loads have been accumulating in the Baltic Sea (HELCOM, 2018b; Voss et al., 2011) and are fuelling primary production. The Baltic Sea is characterized by a spring bloom dominated by diatoms and dinoflagellates, a summer bloom of N-fixing cyanobacteria under warm and calm conditions (Spilling et al., 2018; Wasmund, 1997), and around 220 days with Chlorophyll a (Chl-a) concentrations of $3\,\mathrm{mg{\cdot}m^{-3}}$ or more (Kahru et al., 2016). Primary production is generally limited by N in the open sea, except for the P-limited Gulf of Bothnia (Savchuk, 2018). Denitrification in the sediments and the water column has been estimated to be similar to the external TN loads to the Baltic Sea, while P has been accumulating in the sediments (Savchuk, 2018). Due to the restricted vertical mixing and oxygenation of the deep water layer and high oxygen consumption by biological processing of phytoplankton sedimentation, hypoxia is common in the deep parts of the Baltic Sea, and has expanded to around $20\,\%$ of the Baltic area at its maximum (Jokinen et al., 2018; Conley et al., 2009b). During hypoxic conditions, iron-bound P in the sediment is released to the water column, further increasing primary production (internal P loading; Savchuk, 2018; Vahtera et al., 2007). Moreover, most higher life forms cannot endure such conditions and the sediments and energy flow become dominated by microbes (Diaz and Rosenberg, 2008).

2.2 Baltic coast (Paper IV)

The coastal zones constitute around $25\,\%$ of the Baltic Sea area (total area of $97\,000\,\mathrm{km^2}$) and are outlined in Figure 1.1 (Panel A; dark purple), according to the WFD definition. Although other definitions of the coast have been proposed (e.g., using suspended particular matter; Kratzer and Tett, 2009), the WFD definition has been used throughout the thesis as it is consistent with the national implementations of the WFD and with the HELCOM definition of the coast. The whole Baltic coast is considered in Paper IV and is characterized by high variety, with archipelagos, fjards, cliffs, moraine coasts, lagoons, sandy coasts and deltas (Schiewer, 2008b). Due to the absence of tides, coastal characteristics can be highly different in terms of physical (e.g. morphology, hydrology), chemical, and biological (e.g. species distribution) conditions, even at the local scale (Schiewer, 2008b). Thereby, similar anthropogenic and natural pressures can yield different ecosystem responses. Salinity, temperature, and light levels in the coastal waters are highly variable due to their shallowness (average depth of approximately $5\,\mathrm{m}$; Schiewer, 2008b). This, in turn, affects coastal biodiversity, as only species that can tolerate the variability associated with a coastal zone can survive and thrive (Schiewer, 2008b).

2.2.1 Swedish coast (Paper III)

The Swedish coast, circled in yellow in Figure 1.1 (Panel A), is the focus area of Paper III. It is dominated by hard bottoms, forming fjards interspersed by archipelagos, except for the moraine coasts of southern Sweden (Schiewer, 2008b). Most of the Swedish water discharges to the coast go into the Baltic Sea, from the Gulf of Bothnia in the north to the Danish Straits in the south-west. The exception is in the westernmost part of Sweden that borders the transitional waters between the North Sea and the Baltic Sea (Skagerrak and Kattegat, west of the Danish Straits). Thereby, the Swedish coast covers the whole hydroclimatic gradient of the Baltic system.

2.2.2 Archipelago Sea (Paper I)

The Archipelago Sea, circled in cyan in Figure 1.1 (Panel A), is the focus area of Paper I. Situated in southwestern Finland, it marks the transition between the southern (Baltic Proper) and northern (Gulf of Bothnia) parts of the Baltic Sea, with water flowing from the Baltic Proper to the Gulf of Bothnia through the Archipelago Sea and back along the Swedish coast (Helminen et al., 1998). The archipelago includes more than 17 700 isles aggregated into clusters over an area of around $9500\,\mathrm{km}^2$, and separated by a complex bathymetry, with a mean depth of $23\,\mathrm{m}$ and trenches deeper than $100\,\mathrm{m}$ (Helminen et al., 1998). The Archipelago Sea is fed by eight rivers, spread over a catchment area of $8900\,\mathrm{km}^2$, of which $28\,\%$ is covered by agricultural fields (Helminen et al., 1998). It is characterized by considerable seasonal variations, with ice cover during the winter and water temperatures reaching $20\,^{\circ}\mathrm{C}$ during the summer (Hänninen et al., 2000).

The Archipelago Sea is affected by multiple nutrient sources, from land-based agricultural runoff, fish farms and industries, as well as from inflowing waters of the Baltic Proper and Gulf of Finland, where nutrient concentrations have been increasing (Peuhkuri, 2002). This has given rise to increased eutrophication in the Archipelago Sea, which hinders nutrient buffering and filtering effects (Bonsdorff et al., 1997a). Its complex topography creates varied physical and biogeochemical conditions, which form multiple biotopes linked by a complex food web (Hänninen et al., 2000). These are threatened by eutrophication that unifies the trophic status between inner and outer coastal waters (Bonsdorff et al., 1997b).

2.2.3 Himmerfjärden Bay (Paper II)

The Himmerfjärden Bay, circled in red in Figure 1.1 (Panel A), is the focus area of Paper II. Situated in the north-west of the Baltic Proper, south-west of the Stockholm archipelago, it is a semi-enclosed fjard connected to the outer coast and the open sea primarily through a sound in the south and secondarily through shallow channels. The Himmerfjärden Bay has an area of $174\,\mathrm{km}^2$, with a mean depth of $17\,\mathrm{m}$. Its catchment area is $1286\,\mathrm{km}^2$, larger than the bay area, of which around $20\,\%$ is coved by agricultural fields, and hosts approximately 17 000 inhabitants (Kautsky, 2008). Two main rivers flow into the bay, the Södertälje canal in the north (mean discharge of $0.19\,\mathrm{km}^3{\cdot}\mathrm{year}^{-1}$ during 2000–2009; data from Swedish Meteorological and Hydrological Institute (SMHI)) and the Trosa river flowing into the southwestern part (mean discharge of $0.13\,\mathrm{km}^3{\cdot}\mathrm{year}^{-1}$ during 2000–2009; data from SMHI). Together, these account for $68\,\%$ of the freshwater discharge to the bay. Smaller streams scattered through the catchment account for the remaining $32\,\%$ of the discharge. The Himmerfjärden Bay is thermally stratified during summer, thereby often creating seasonal hypoxia mostly in its inner waters (Bonaglia et al., 2014). Sea ice can form during the winter, mostly in the northernmost and sheltered areas, which only impedes wind-mixing during harsh winters (Engqvist and Stenström, 2009). The bay's residence time is relatively long, generally extending for more than 50 days (Engqvist, 1996).

External nutrient loads to the bay mainly originate from riverine loads (accounting for more than $50\,\%$ of the TN and TP loads) and direct point source loads to the coastal waters (accounting for about $40\,\%$ of TN and TP loads; Paper II, Figure 2). Point source loads are dominated by the Himmerfjärdsverket wastewater treatment plant (WWTP), which serves $300\,000$ people of the southern Stockholm region and discharges in the middle eastern part of the bay (Franzén et al., 2011). The TN load has increased from 1976 to 1983 and strongly decreased in the mid-1990s, while the TP load has decreased from 1976 to 1983 and plateaued since the late 1980s (Kautsky, 2008). Historically, N has been the limiting nutrient for primary production, which has shifted to P due to the increasing TN load and has shifted back to N since 1993 (Kautsky, 2008). The Himmerfjärden Bay is a net long-term exporter of nutrients (mainly P) to the sea, with inter-annual variations depending on nutrient loads and weather conditions (Kautsky, 2008).

3 Methods

3.1 Coastal water quality modelling (Papers I and II)

For Papers I and II, a water quality modelling approach has been developed and implemented to investigate effects of possible land, coast and sea-based management measures on coastal water quality (Objective A) and effects of hydroclimatic conditions on water quality and management measure effectiveness (Objective B).

3.1.1 Conceptualisation of modelling approach

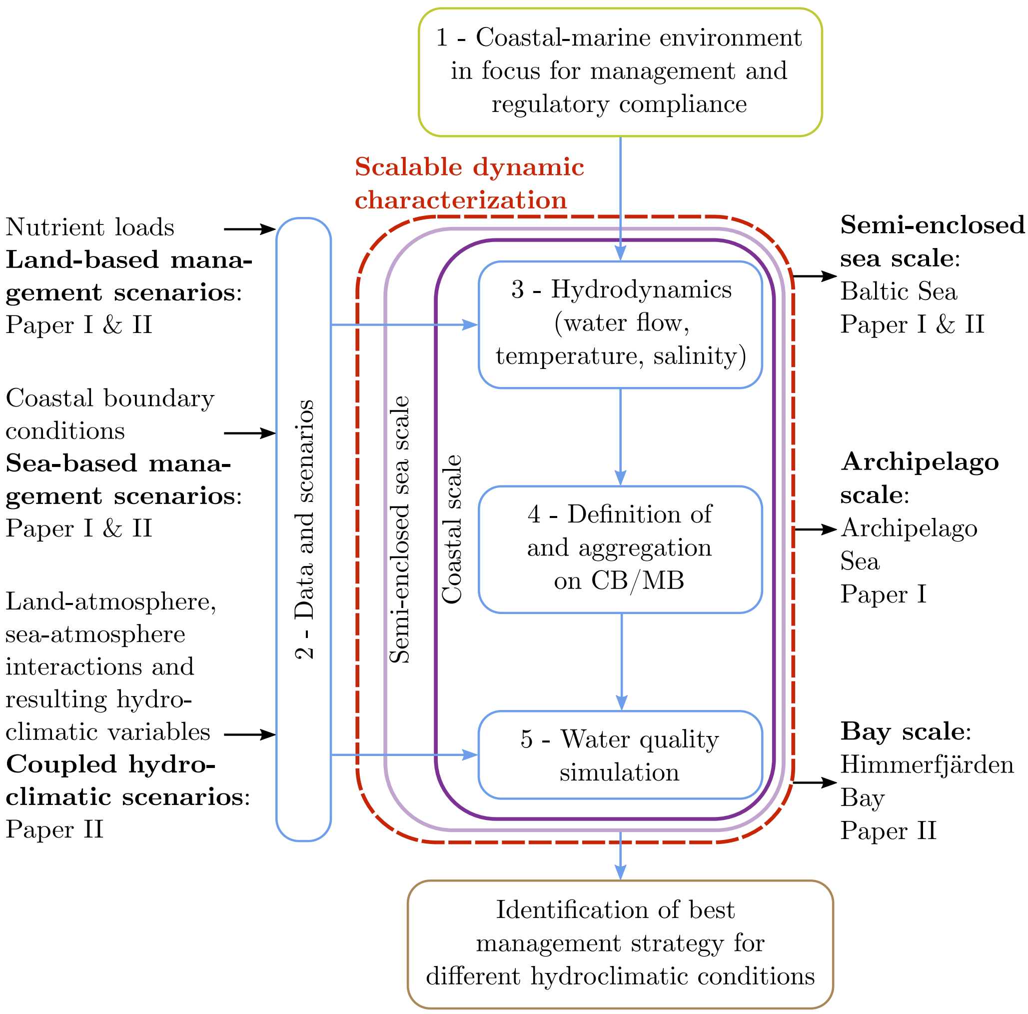

Figure 3.1. Schematic representation of the modelling approach and its main components (modified from Paper I Fig. 7).

Coastal water quality dynamics depend on various anthropogenic and natural drivers, such as nutrient loads and hydroclimatic conditions, including also the freshwater discharges carrying these loads from land to sea. The various drivers act on different scales, from the local land-based catchment and coastal waters to the open sea waters and the regional whole-sea catchment (Savage et al., 2010; Almroth-Rosell et al., 2016; Bring et al., 2015). Thereby, changes in conditions from the local to the whole regional scale may be essential in influencing local coastal water quality conditions and internal processes. As such, the effects of drivers, possible management measures and scale-coupling on coastal eutrophication need to be further investigated, which can be done, e.g., through scenario simulations. The modelling approach developed in this thesis aims to relatively simply and robustly characterize water quality conditions and project their change evolution for various management and hydroclimatic scenarios (Figure 3.1). In the case of semi-enclosed seas, such as the Baltic Sea, both anthropogenic and natural drivers can also strongly affect the open sea conditions, which in turn can influence the coastal conditions and thus need to be coherently considered in the coastal modelling approach.

Drivers of coastal water quality conditions are inherently dynamic with substantial seasonality (e.g., in nutrient loads, hydroclimate; Stålnacke et al., 1999). Similarly, coastal water quality responses to these drivers are also dynamic, with important internal processes and feedbacks. To capture these linked dynamics, the developed modelling approach considers both hydrodynamic and biogeochemical water quality processes. For simulation efficiency, the approach is simplified by decoupling the hydrodynamic and water quality simulations. This is done because three-dimensional hydrodynamic simulations are computationally costly, and while the hydrodynamics affect the nutrient and algae transport and water quality processes, the latter do not substantially affect the hydrodynamic processes (Lehtoranta et al., 2009). The modelling approach is also scalable, i.e., can have varying spatial resolution in its system representation. The variable model resolution allows for increased spatial resolution of coastal areas in focus for the modelling, and model simplification with coarser resolution in other areas, depending on their physical and morphological conditions. It also allows for use of a similar modelling approach, ensuring data and model consistency, in modelling at the different local coastal and larger open-sea scales. Openly available coastal water quality data with relatively high (monthly) temporal resolution are generally scarce for the Baltic coasts, which complicates rigorous model calibration and validation. This is not the case for the open sea, and thus using the same modelling approach on both the coastal and the open sea scales allows for relevant model testing against the monitoring data that are available on the open sea. Open sea testing and validation can to some degree build confidence also for the coastal-scale model, at least for approximate scenario projections rather than specific, precise predictions. The main steps of the developed modelling approach are presented in Figure 3.1. Section 3.1.2 summarises, and Papers I and II and their Supplementary Materials (SMs) explain in more detail the approach, implementation and application for the Baltic Sea (Paper I, simulation period: 2001-2009), and the coastal areas of the Archipelago Sea (Paper I, simulation period: 2001–2005; outlined in cyan in Figure 1.1, Panel A), and Himmerfjärden Bay (Paper II, simulation periods: 2000, 2003 and 2005; outlined in red in Figure 1.1, Panel A).

3.1.2 Application of modelling approach

Hydrodynamics (Step 3)

For the Baltic Sea (Paper I), the hydrodynamics used to force the water quality model has been simulated by Dargahi et al. (2017) using the three-dimensional hydrodynamic model Generalized Environmental Modelling System for Surface waters (GEMSS®) for the period 2000–2009. The model has been validated for the period 2000–2009 (Dargahi et al., 2017) and has a horizontal resolution of $5\,\mathrm{km}$ and 47 vertical layers. The model forcings include an open boundary south-west of the Baltic Proper, wind stress, surface heat flux and freshwater discharge from the 69 main rivers flowing to the Baltic Sea. For the Archipelago Sea (Paper I), the hydrodynamics has been simulated by downscaling the Baltic Sea model, using GEMSS® for the period 2001–2005 (Dargahi and Cvetkovic, 2014). The coastal model has a horizontal resolution of approximately $1.3\,\mathrm{km}$ and its boundary conditions are set using the Baltic Sea model results. For Paper II, both the Baltic Sea (Chen et al., 2019a) and the Himmerfjärden Bay hydrodynamics have been simulated separately using the three-dimensional Finite Volume Community Ocean Model (FVCOM) (Chen et al., 2003).

Definitions of and aggregation on Marine and Coastal Basins (Step 4)

The delimitation of the Baltic Sea, excluding the western transition to the North Sea, into 12 Marine Basins (MBs) is presented in Figure 3.2B and has been made according to the Baltic Sea bathymetry. The delimitation follows the sills that play an important role for hydrodynamic processes (Leppäranta and Myrberg, 2009; HELCOM, 2013a) to ensure greater horizontal mixing inside each basin than among them.

Each MB has been separated into two vertical layers to distinguish between the differing processes of the mixed surface layer, where most of the primary production takes place, and of the deeper layer, which can experience anoxic conditions. This separation has been made at the pycnocline, where the density gradient is maximal and the vertical mixing is minimal (Reissmann et al., 2009), which corresponds most often to the halocline in winter and the thermocline in summer. Each MB layer is assumed fully mixed. The hydrodynamic model results are further used to calculate the horizontal and vertical flow between the MBs and their layers, governing the transport of algae and nutrients, as well as volume-averaged temperature and salinity used to compute the biogeochemical rates. Both the fully mixed assumption and the division in only two layers are important simplifications, especially for deep waters where gradients of nutrient concentrations can be strong (Conley et al., 1997). However, these simplifications are more suitable for the surface waters above the halocline (approximately $65\,\mathrm{m}$), which represent $70\,\%$ of the Baltic Sea volume (Stigebrandt and Wulff, 1987) and are the main focus for management (HELCOM, 2009).

The finer-resolved delimitations of the Archipelago Sea into 82 Coastal Basins (CBs) and of the Himmerfjärden Bay into 13 CBs are presented in Figure 3.2C and A, respectively. In analogy with the coarser-resolved MB delimitation, the finer-resolved CB delimitations follow the sills and straits of the coastal bathymetry. For the Himmerfjärden Bay, the CB delimitation also follows the Swedish implementation of the WFD (Naturvårdsverket, 2006). These CBs differ in their horizontal resolution for the two coastal study areas, to capture a greater variability in the Archipelago Sea case. Vertically, the CBs have been divided into two layers at $10\,\mathrm{m}$ depth for the Archipelago Sea, and at the pycnocline for the Himmerfjärden Bay.

Water quality (Step 5)

The water quality simulations have been performed using a relatively simple carbon-based water quality model, developed and validated by Kiirikki et al. (2001, 2006). The model includes a pelagic and a benthic part. The pelagic part describes two algae groups, the N-fixing cyanobacteria and the other phytoplankton (diatoms and dinoflagellates), as

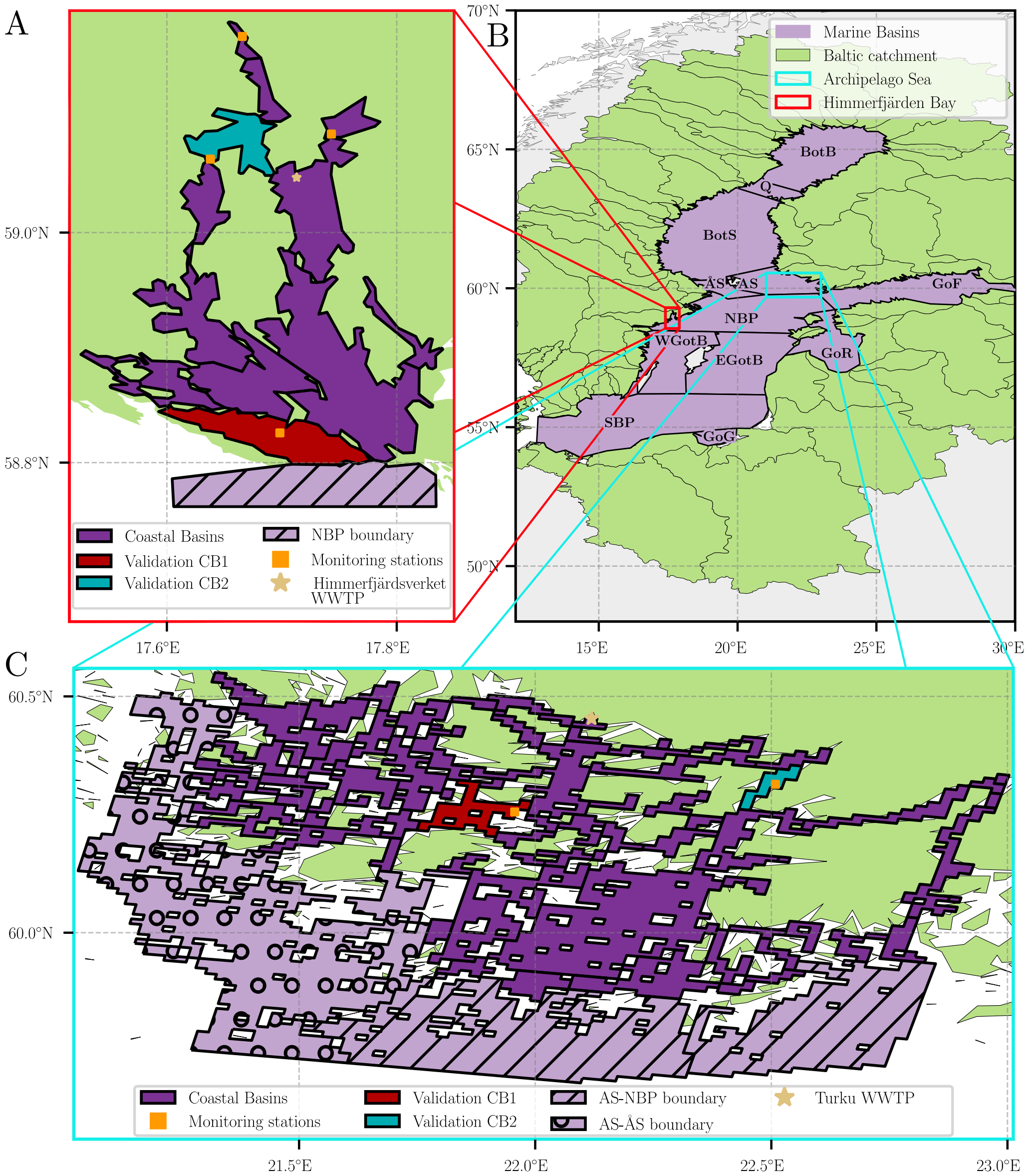

Figure 3.2. Maps of the study areas and their delimitation into Marine Basins (MBs; BotB: Bothnian Bay, Q: the Quark, BotS: Bothnian Sea, ÅS: Åland Sea, AS: Archipelago Sea, GoF: Gulf of Finland, GoR: Gulf of Riga, NBP: Northern Baltic Proper, EGotB: Eastern Gotland Basin, WGotB: Western Gotland Basin, SBP: Southern Baltic Proper, GoG: Gulf of Gdansk) for the Baltic Sea (Panel B) and Coastal Basins (CBs) for the Himmerfjärden Bay and Archipelago Sea (Panel A and C, respectively). The monitoring stations are shown in orange and the CBs for which the validation is shown in the thesis are coloured in blue and red.

well as concentrations of dissolved inorganic (dissolved inorganic nitrogen: ammonium, nitrite and nitrate (DIN); dissolved inorganic phosphorus: phosphate (DIP)) and organic nutrients (N, P and carbon (C) in sedimenting algae) (Kiirikki et al., 2001). The benthic part describes the sedimentation and nutrient processes of mineralization, denitrification and internal loading of iron-bound P in the mobilizable sediments (Kiirikki et al., 2006). These nutrient processes are modelled as a function of the sediment CO_2 release (Kiirikki et al., 2006), which is used to represent oxygen consumption and microbial mineralization pathways in the mobilizable sediments (Kiirikki et al., 2006; Lehtoranta et al., 2009). In Paper I, the sediment layer of a CB/MB is considered anoxic, leading to internal loading of P when the sediment CO_2 release exceeds a threshold (Kiirikki et al., 2006). This representation has been improved in Paper II by considering an anoxic area, described as a linear function of the sediment CO_2 release that limits the internal loading. The sediment processes are described as a function of CO_2 release rather than oxygen concentrations because the latter can exhibit vertical gradients (Conley et al., 1997) that would not be captured by the simplified vertical basin division in only two layers.

The water quality model is applied to each vertical layer of each CB for the Archipelago Sea and Himmerfjärden Bay, and to each MB for the Baltic Sea. Biogeochemical process rates in the water quality model depend on water temperature, which is provided by the hydrodynamic model results, aggregated over each basin and layer, and on solar radiation that quantifies the energy available for photosynthesis. Moreover, the model is forced by nutrient and algae concentrations at the model open boundary that quantifies the exchange of nutrients with the open Baltic Sea and the North Sea. These boundary conditions are obtained either from monitoring stations or from the Baltic Sea scale model results, and by atmospheric and land-based nutrient loads.

Data sources and scenarios (Step 2)

Table 3.1 presents the various data and their sources used for calibrating, validating and running the model for the open Baltic Sea, Archipelago Sea and Himmerfjärden Bay.

Table 3.1. Datasets and their sources and processing for the monitoring and forcing parameters used to calibrate, validate and run the water quality models for the Baltic Sea, Archipelago Sea and Himmerfjärden Bay.

Location and paper	Variable	Data processing and source
Baltic Sea – Papers I and II	Hydrodynamic forcings: water temperature, salinity and flow	Results from hydrodynamic models GEMSS® (Paper I[a]) and FVCOM (Paper II[b]), aggregated on each MB and vertical layer (see Section 3.1.2)
	Solar radiation	Daily solar radiation extracted at the centre of each MB from STRÅNG[c]
	Nutrient loads	Atmospheric, point source and riverine TN and TP loads for each MB from HELCOM PLC 5.5[d]
	Boundary conditions for DIN, DIP and algae concentrations	Concentrations at the HELCOM BY2 monitoring station (southwestern Baltic Proper) from SMHI SHARK[e]
	Calibration and validation: DIN, DIP and Chl-a concentrations	Open sea monitoring data for the MBs (one to three monitoring stations available for most MBs) from SMHI SHARK[e] and ICES[f] databases (see Paper I SM S3 Figure S1)
Archipelago Sea – Paper I	Hydrodynamic forcings: water temperature, salinity and flow	Results from hydrodynamic model GEMSS® downscaled to the Archipelago Sea[g], aggregated on each CB and vertical layer (see Section 3.1.2)
	Solar radiation	Spatially uniform daily solar radiation extracted at the centre of the Archipelago Sea MB from STRÅNG[c]
	Nutrient loads	Point source (industries, WWTPs, fish farms) and riverine loads for the year 2000 and repeated for the simulation period 2001–2005 obtained from Ympäristö[h]
	Boundary conditions for all pelagic water quality variables	Obtained from the Baltic Sea water quality model results of Paper I
	Calibration and validation: DIN, DIP and Chl-a concentrations	Monitoring data at the stations Nau 2361 Seili intens and Pala 115 Tryholm from SYKE[i]

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Location and paper	Variable	Data processing and source
Himmerfjärden Bay – Paper II	Hydrodynamic forcings: water temperature, salinity and flow	Results from hydrodynamic model FVCOM for the Himmerfjärden Bay[j], aggregated on each CB and vertical layer (see Section 3.1.2)
	Solar radiation	Spatially uniform daily solar radiation extracted at the centre of the Himmerfjärden Bay from STRÅNG[c]
	Nutrient loads	Point source, atmospheric and riverine loads for the period 2000–2009 obtained from the SMHI HYPE model[k]
	Boundary conditions for all pelagic water quality variables	Obtained from the Baltic Sea water quality model results of Paper II
	Calibration and validation:DIN, DIP and Chl-a concentrations	Surface and bottom monitoring data during summer at four stations from SMHI SHARK[e] for the validation. Weekly to monthly measurements during 2015[l]for calibration of site specific rates.

[a] Dargahi et al. (2017)
[b] Chen et al. (2019b)
[c] SMHI mesoscale solar radiation model (STRÅNG; Landelius et al., 2001)
[d] HELCOM Pollution Load Compilation (PLC) 5.5 (HELCOM, 2013b)
[e] SMHI Swedish Ocean Archive (SHARK) database
[f] International Council for the Exploration of the Sea (ICES) Dataset on Ocean Hydrography
[g] Dargahi et al. (2017)
[h] Finnish Environmental Agency (Ympäristö; `http://www.ymparisto.fi`)
[i] Finnish Environmental Institute (SYKE; `http://www.syke.fi/en-US/Open_information`)
[j] Paper II, SM S2.1 and S5
[k] SMHI Hydrological Predictions for the Environment (HYPE) (version 5.4.1) (Lindström et al., 2010)
[l] Sjöholm (2015)

For Paper II, three representative hydroclimatic condition-years have been defined based on available data over the period 2000–2009, by classifying types of hydroclimatic conditions in terms of river discharges to the coast and sea, and net heat flux over coastal and marine waters, which quantifies the net solar energy budget over the coastal and sea areas. These hydroclimatic parameters have been identified as main drivers of hydrodynamic processes, such as water salinity and temperature, which also influence the biogeochemical processes by driving nutrient loads and water temperature (Chen et al., 2019b; Meier et al., 2012). The hydroclimatic classification shown in Table 3.2 is based on the relative differences between the specific condition-years and the corresponding 2000–2009 period-average values for runoff and heat flux, averaged between the Himmerfjärden Bay and the Baltic Sea conditions, since the latter also influence the Himmerfjärden Bay through the open sea boundary. Thereby, the condition-year 2005 is classified as representative of dry-cold year, the condition-year 2003 as a dry-warm year, and the condition-year 2000 as a wet-warm year. The dry-warm and wet-warm condition-years are selected to reflect possible trends towards projected wetter and warmer scenario conditions for the Baltic Sea (The BACC II Author Team, 2015), while the dry-cold condition-year is more representative of the 2000–2009 average conditions.

For Paper II, these condition-years are further used to investigate the effectiveness of possible eutrophication mitigation measures under various hydroclimatic conditions. The management scenarios considered are outlined in Table 3.3, and include land-based, coastal, and sea-based mitigation measures and effects. These scenarios are simulated for 30 years by repeating the forcing data for each condition-year in order to reach a

Table 3.2. Classification of hydroclimatic condition-years for Paper II based on forcing data for the Himmerfjärden Bay and the Baltic Sea for river discharges (from SMHI and HELCOM, respectively) and net heat flux (from the Woods Hole Oceanographic Institution).

	Location	dry-cold	dry-warm	wet-warm	Period average
Year / Period		2005	2003	2000	2000 – 2009
River discharge in $m^3{\cdot}s^{-1}$ (% relative difference from period average)	Himmerfjärden Bay	11.6 (−20)	9.8 (−32)	20.1 (39)	14.5
	Baltic Sea	13 876 (1.4)	10 752 (−21)	16 325 (19)	13 683
Net heat flux in $W{\cdot}m^{-2}$ (% relative difference from period average)	Himmerfjärden Bay	3.1 (33)	6.0 (158)	17.6 (655)	2.33
	Baltic Sea	−4.2 (−237)	8.2 (167)	16.3 (431)	3.07

pseudo steady-state that removes transient effects and describes the evolution trend of the system towards equilibrium at that steady-state. For Paper I, the considered management scenarios (Table 3.3), involving land- and sea-based mitigation measures and effects, are simulated transiently for the period 2001–2005, using nutrient loads for the year 2000. Model set-ups for the Archipelago Sea and Himmerfjärden Bay are described in more detail in Papers I and II, respectively.

The various management scenarios (Table 3.3) have been considered and simulated in Papers I and II in order to investigate the effects of such potential measures and their combinations on coastal water quality and eutrophication. The baseline scenarios correspond to no further mitigation taking place in relation to the nutrient loads of year 2000 for Paper I, and of the three condition-years for Paper II. Scenarios S_{PS_AS} and S_{PS_HB} correspond to improved nutrient removal technologies in the Turku and Himmerfjärdsverket WWTP, respectively. The riverine scenario (S_{R_HB}) is only considered for the Himmerfjärden Bay and corresponds to effects of possible land-catchment measures to mitigate riverine nutrient loads, such as improved wastewater treatment, and capture-reduction of nutrients from agricultural leakage and/or releases from legacy sources (Destouni and Jarsjö, 2018) by buffers and wetlands in the catchment. The coastal scenario (S_{Aer_HB}) is only considered for the Himmerfjärden Bay and corresponds to possible geoengineering solutions to improve oxygen conditions in the bay, e.g., by oxygenation, or chemical binding of sediment P (Conley et al., 2009a). This scenario also facilitates investigation of the role of internal P loading in the bay for the management of coastal eutrophication conditions. The sea-based scenarios S_{Sea_AS} and S_{Sea_HB} correspond to GEnS, as defined by HELCOM, being reached in the open sea MBs at the boundary of the Archipelago Sea and Himmerfjärden Bay, respectively, through application of combined land and coastal measures throughout the whole Baltic catchment and coast.

Ecological status analysis

In Paper II, Ecological Status (EcS) has been calculated for each water quality component (DIN, DIP and Chl-a) for the CBs in Himmerfjärden Bay. The EcS for each component depends on the CB type, the average summer or winter value of the component, and a reference condition that depends on the salinity. Depending on its EcS value, each CB

Table 3.3. Management scenarios considered in Papers I and II, for possible mitigation measures on land, at the coast, and/or in (or affecting) the sea.

Paper	Scenario	Mitigation type	Assumed mitigation
Paper I	S_{0_AS}	Baseline	No mitigation
	S_{PS_AS}	Land-based	50 % reduction in nutrient loads from the Turku WWTP
	S_{Sea_AS}	Sea-based	Assumed reduction in DIN, DIP and Chl-a concentrations corresponding to the open sea boundary close to GEnS
	S_{PS+Sea_AS}	Land- and sea-based	Combination of S_{PS_AS} and S_{Sea_AS}
Paper II	S_{0_HB}	Baseline	No mitigation, hydroclimatic conditions repeated for each condition-year
	S_{PS_HB}	Land-based	50 % reduction in nutrient loads from all point sources (mainly from the WWTP in the Himmerfjärden CB)
	S_{R_HB}	Land-based	Same absolute load reduction as for S_{PS_HB}, allocated over the rivers proportionally to their load share
	S_{PS+R_HB}	Land-based	Combination of S_{PS_HB} and S_{PS_HB}
	$S_{(PS+R)_N\_HB}$	Land-based	S_{PS+R_HB} with only TN reduction
	$S_{(PS+R)_P\_HB}$	Land-based	S_{PS+R_HB} with only TP reduction
	S_{Aer_HB}	Coast-based	Oxic water and sediment conditions assumed in the whole Himmerfjärden Bay (no internal P loading)
	S_{Sea_HB}	Sea-based	Assumed reductions in nutrient and algal concentrations corresponding to the open sea boundary reaching GEnS
	$S_{PS+R+Sea\_AS}$	Land- and sea-based	Combination of S_{PS+R_HB} and S_{Sea_HB}

water quality component is further classified as poor, insufficient, moderate, good and high, and the area weighted proportion of the Himmerfjärden Bay reaching good EcS is calculated as a comparative analysis.

3.2 Trend analysis (Paper III)

In Paper III, the links of various coastal eutrophication variables and their possible drivers spanning the anthroposphere, atmosphere, and hydrosphere are investigated. This is done by analysing the degree to which temporal change trends in driver conditions can explain corresponding change trends in observed coastal eutrophication variables, and how the explanatory power varies with location and scale (Objective B). The analysis is conducted on the Swedish coast, stretching from the southwestern transition between the Baltic and North Seas until the northern Gulf of Bothnia (outlined in yellow in Figure 1.1). The study period is 1990–2020, during which the Baltic Sea has undergone considerable hydroclimatic changes, greater than the world average (The BACC II Author Team, 2015), while its nutrient loads have decreased. The considered water quality variables and drivers are presented below in Section 3.2.1. Various spatial aggregation scales are considered for these variables and drivers, as presented further below in 3.2.2. Furthermore, the calculations of change trends and their correlations are presented in Sections 3.2.3–3.2.4.

3.2.1 Water quality variables and drivers

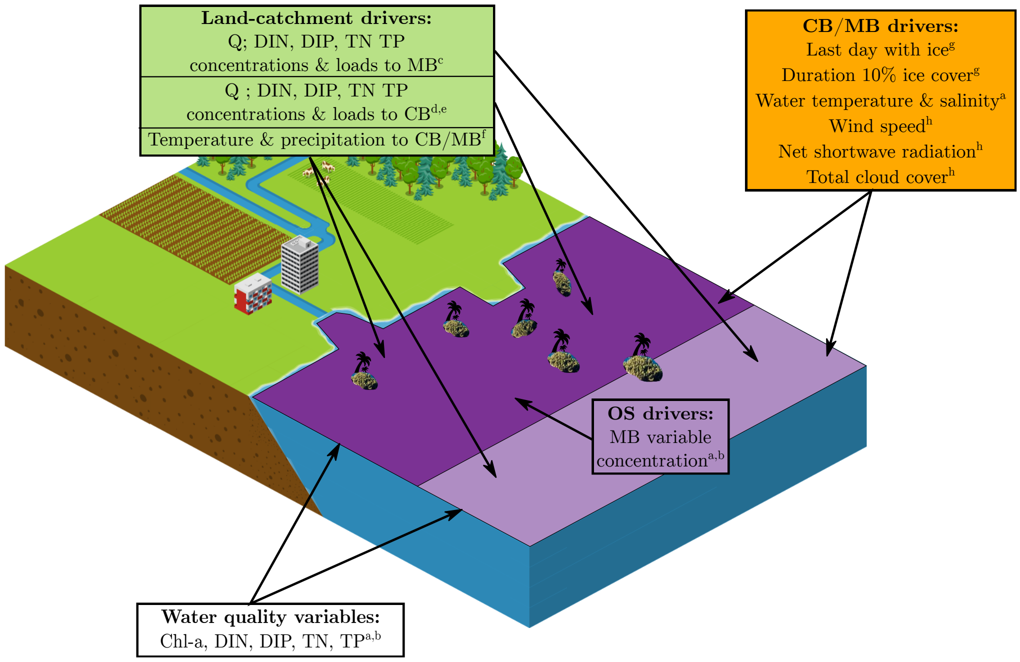

Figure 3.3. Schematic representation of the study system with the catchment in green, the Coastal Basins (CBs, both more and less isolated CBs) in dark purple and the open sea Marine Basins (MBs) in light purple. The black arrows represent the scales associated with the water quality variables (white box), land-catchment drivers (green box), Coastal and Marine Basin drivers (orange box) and open sea drivers (light purple box) that are analysed in Paper III. Data sources: [a]SMHI Swedish Ocean Archive (SHARK) database; [b]International Council for the Exploration of the Sea (ICES) Dataset on Ocean Hydrography; [c]HELCOM Pollution Load Compilation (PLC) 5.5 (HELCOM, 2013b); [d]SMHI Hydrological Predictions for the Environment (HYPE) (version 5.9) (Lindström et al., 2010); [e]Soil, Water, Environment database (MVM) database from Swedish University of Agricultural Sciences (SLU; `https://miljodata.slu.se/mvm/Default.aspx`); [f]Climate Research Unit Time Series (CRU TS) version 4.03 (Harris et al., 2020); [g]Finnish Meteorological Institute (FMI) ice dataset (Berglund and Eriksson, 2015); [h]ERA5 dataset (Copernicus Climate Change Service (C3S), 2017).

The considered water quality variables (white box in Figure 3.3) represent prevailing eutrophication conditions. Their selection has been based on the water quality indicators defined by the Swedish implementation of the WFD to be relevant for eutrophication (Havs- och vattenmyndigheten, 2019). These include surface concentration of Chl-a during the summer period (Summer Chl-a), Summer TN and TP, and Winter TN, TP, DIN and DIP (surface concentration (within 0–10 m) of the variable during the summer or winter period). The summer and winter periods are June-August and November-March, respectively, with the summer months covering the growth period following the spring bloom, and the winter months covering the period of low primary production before the spring bloom (HELCOM, 2017). Summer Chl-a is the main variable commonly used as a proxy for eutrophication. In addition, other variables relating to eutrophication management are also considered, such as the ratio between Winter TN and Winter TP (Winter TN:TP).

The considered drivers have been classified into three categories, depending on their associated scales and locations in the land-coast-sea system. First, the land-catchment drivers (light green in Figure 3.3) represent changes in anthropogenic and hydroclimatic conditions over the local coastal and/or the regional marine catchment. These include nutrient loads and concentrations, freshwater discharges, and catchment temperature and precipitation. Second, the coastal or marine basin drivers (orange in Figure 3.3) acting directly in or over coastal or marine waters represent changes in local coastal or regional marine hydrospheric conditions or atmospheric forcings. These include sea ice, seawater salinity and temperature, wind, net shortwave heat flux and cloud cover. Third, the open sea drivers (light purple in Figure 3.3) represent changes in open sea nutrient and Chl-a concentrations that can also affect coastal eutrophication.

3.2.2 Coastal and marine component scales and classification

The open sea waters of the Baltic Sea have been divided into MBs, following the HELCOM definition (HELCOM, 2017). Their eutrophication conditions are investigated in relation to regional drivers, and the MBs conditions are also investigated as open sea drivers of coastal eutrophication. The coastal waters of Sweden have been divided into CBs, following the Swedish implementation of the WFD (Naturvårdsverket, 2006). To investigate the effects of coastal location on eutrophication driver-responses relationships, the CBs have been further partitioned into more and less isolated CBs, as described by the Swedish Environmental Protection Agency (2000). CBs with a water exchange rate lower than 40 days are classified as more isolated and CBs with a water exchange rate of 40 days or more are classified as less isolated CBs. The water exchange rate is determined based on the morphological links between CBs (strait, archipelago, open coast or bay) and the number of CBs separating a given CB from the open sea (Swedish Environmental Protection Agency, 2000). The less isolated CBs are directly linked to the open sea or to an open coast that is linked to the open sea. The more isolated CBs are situated further from the open sea or connected to a CB linked to open sea through a narrow strait or in an archipelago. CBs and MBs with sufficient data availability for the considered water quality variables have been included in the analysis and are shown Figure 3.4A (less isolated CBs in light orange, more isolated CBs in blue-green, and MBs in light purple). This classification (more isolated CBs; less isolated CBs; all CBs; MBs) is used to aggregate and analyse relationships between drivers and water quality variables.

3.2.3 Processing of water quality variables and drivers

The datasets that have been analysed and spatially and temporally aggregated for each CB and MB are shown in Figure 3.3. Datasets based on direct measurements have been preferably used, but reanalysis datasets have also been considered when direct measurements were unavailable or unpractical to use (e.g., due to limited data availability, challenging interpolation). For all variables and drivers, the available data have been resampled to yearly resolution to calculate their longer-term temporal trends.

The water quality variables have been aggregated onto the CBs following the Swedish WFD guidelines (Havs- och vattenmyndigheten, 2019) and onto the MBs following the HELCOM BSAP (HELCOM, 2018a), using surface measurements within 0–10 m depth during the winter or summer period months for each CB or MB. For each variable, only CBs and MBs with at least 50 % available data within a 10-year or longer time series have been analysed. For the investigation of open sea driver influences, each CB has been paired to the nearest MB with sufficient data (or to the MB that it has the highest water exchange with when linked to more than one MB).

For the drivers, the aggregation depends on the spatial and temporal resolution of the dataset. Land-catchment drivers have been calculated for the local catchments associated with each CB, and the regional catchments associated with each MB. Freshwa-

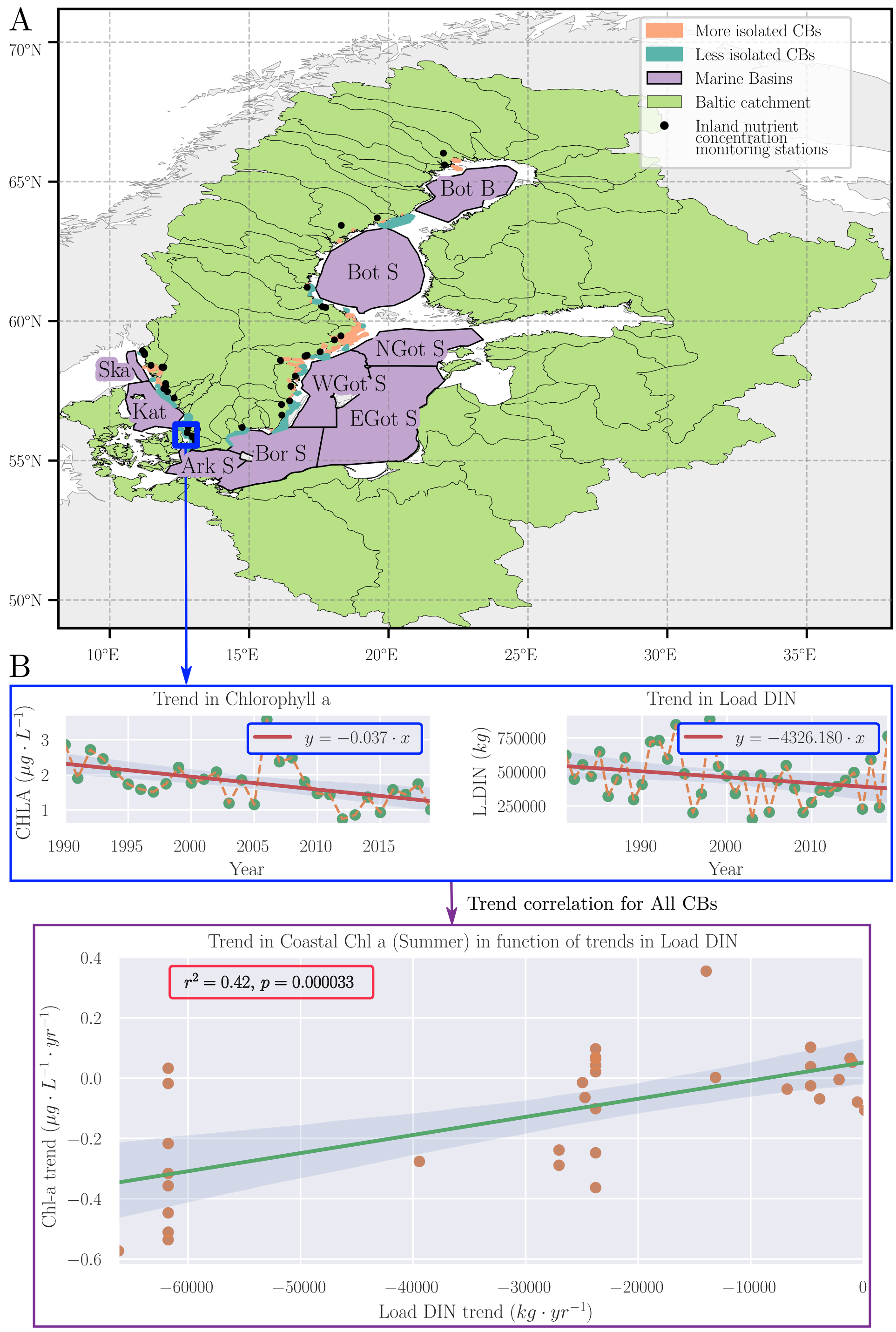

Figure 3.4. Panel A: Map of the Baltic Sea and of its Coastal Basins (CBs) included in the analysis of Paper III, divided into more isolated CBs (in blue-green) and less isolated CBs (in light orange). The division is explained in Section 3.2.2. The Marine Basins (MBs) of the Baltic Sea (in light purple circled by black lines) that are associated to the CBs included in the study are Skagerrak (Ska), Kattegat (Kat), the Arkona Sea (Ark S), the Bornholm Sea (Bor S), the Eastern Gotland Sea (EGot S), the Western Gotland Sea (WGot S), the Northern Gotland Sea (NGot S), the Bothnian Sea (Bot S) and the Bothnian Bay (Bot B). The monitoring stations providing nutrient concentrations for inland waters are represented by the black dots (modified from Paper III Fig. 1). Panel B: Exemplification of the trend correlation analysis (explained in Section 3.2.4). Temporal trends of variables (top left, Summer Chl-a) and drivers (top right, DIN Load) are calculated for each CB. These trends are then correlated for all CBs/MBs with sufficient data availability in a given class (bottom, for the class All CBs).

ter discharges and nutrient loads (TN and TP) to the MBs have been retrieved from the HELCOM PLC5.5 dataset, from which flux averaged nutrient concentrations have been calculated. Each CB has been associated with the nearby catchments discharging either directly into that CB or into an upstream CB. Nutrient concentration measurements in the coastal catchments have been obtained from the SLU MVM database (Figure 3.4) and freshwater discharges at the mouth of rivers with available nutrient measurements have been obtained from the hydrological HYPE model (Figure 3.4). These datasets have been used to calculate the total freshwater discharge and nutrient loads of DIN, DIP, TN and TP associated with the CB (by multiplying monthly averaged concentrations by monthly averaged discharges), from which the total flux averaged nutrient concentration has been derived by dividing each nutrient load by the total freshwater discharge. Temperature and precipitation have been averaged over the local and the regional catchments associated with each CB and MB, respectively. Coastal and marine basin drivers have been aggregated directly over each CB or MB. For seawater salinity and temperature, the aggregation is similar to that for the nutrient variables. Concentrations of sea ice (Figure 3.4) have been averaged over each CB and MB and used to calculate the last winter/spring day of the year with more than $10\,\%$ ice concentration, and the duration with $10\,\%$ or more ice cover during the winter and spring. Net surface shortwave radiation, total cloud cover and wind speed, retrieved from the ERA5 reanalysis dataset, have been averaged over the area of each CB and MB.

3.2.4 Trend and correlation analysis

Relationships between trends in two variables, or in a water quality variable and a driver, have been investigated by trend correlation evaluation. Slopes of temporal trends have been calculated for each variable-variable or variable-driver pair over their common period with available data for each CB and MB, as exemplified in Figure 3.4B for the Summer Chl-a variable and the DIN load driver for a specific CB. Variable trends have then been correlated with driver or variable trends for all MBs and CBs in each class (MBs; all CBs; more isolated CBs; less isolated CBs), using the Pearson correlation. This is exemplified in Figure 3.4B, where trends in the Summer Chl-a variable are correlated with trends in the DIN load driver for the class of All CBs. This yields the Pearson coefficient of determination (r^2), showing the degree to which trend variations in one possible explanatory variable or driver can explain trend variations in another variable, and the p-value, showing how statistically significant the relationship is. Minimum corresponding lengths of time series have been chosen to ensure a sufficient amount of temporal data for trend calculations and spatial data for trend correlations. These have been set to 10 years for driver-variable comparisons, because some driver time series have relatively short time series lengths (e.g., sea-ice), and to 20 years for variable-variable (and open sea driver) comparisons, because these data have longer time series with possible greater variability and associated uncertainty (e.g., variability in measurement depths and considerable horizontal heterogeneity; Scheinin et al., 2020).

This trend correlation analysis yields the proportion of the spatial variance in a variable trend that is explained by the spatial variance in another variable or driver trend. Thereby, it cannot identify the possible explanatory power for drivers that do not exhibit any trend. It also cannot detect non-linear or delayed relationships between variables and drivers. Moreover, there is no 20-year Summer Chl-a time series for the northern CBs, thus statistical relationships may not apply for this part of the Baltic Sea coast.

3.3 Research links and gaps analysis (Paper IV)

Following the investigation of interlinkages between drivers and coastal eutrophication dynamics through the trend analysis in Paper III, Paper IV analyses how the interlinkages between processes, drivers, ecosystem impacts, and management of coastal eutrophication in the Baltic Sea have been studied in the published scientific literature (Objective C). This is done through a rapid scoping literature review aiming to map important connections and identify key gaps in the research on Baltic coastal eutrophication, in order to highlight needed research efforts that bridge the gaps and support management solutions.

3.3.1 Research screening

The literature review has been performed using the scoping review approach that allows quantifying and mapping the research effort on a particular topic, for example to uncover research gaps (Munn et al., 2018). This approach is more suitable for the broad and open research aim of this literature screening than a traditional systematic review, which aims to gather and assess evidence on a particular question (Munn et al., 2018). The scoping review has mostly considered the study abstracts, in order to focus on the key points selected by the authors, and has been performed by only one reviewer, simplifying some systematic components of the scoping review and thereby making it a rapid review (Khangura et al., 2012).

The rapid scoping review process used here can be divided in four steps: identification, screening, inclusion, and data extraction. The process for identifying studies that may be relevant for the literature review is described as follows:

- Definition of the search key to target investigation of the Baltic Sea (3.1), the coastal zone (3.2), and eutrophication (including also internal eutrophication processes) (3.3):

$$(((\text{"}Baltic\ Sea\text{"})\ AND \tag{3.1}$$

$$(coast^{*}\ OR\ estuar^{*})\ AND \tag{3.2}$$

$$\begin{aligned}&(eutrophic^{*}\ OR\ (nutrient^{*}\ NEAR/2\ concentration^{*})\ OR\\&(alg^{*}\ NEAR/2\ bloom^{*})\ OR\ (hypox^{*})\ OR\ (water\ NEAR/3\ quality)\\&OR\ ((ecosystem\ OR\ environment^{*})\ NEAR/3\ status))))\end{aligned} \tag{3.3}$$

- Identification of 1855 studies published in English through a Web of Science$^{\mathrm{TM}}$ (WOS) database search on January 4$^{\mathrm{th}}$ 2021.

- Retrieval of literature search results (title, abstract, keywords, authors, publisher, publication data and number of citations) for screening.

The screening and inclusion process has been carried out by manually identifying studies that focus on Baltic coastal eutrophication and including them for data extraction via the following steps:

- Definition of the inclusion criteria:
  - The Baltic Sea focus criterion includes studies:
    * focusing solely on the Baltic Sea system
    * considering the Baltic Sea together with other marine areas (e.g., reviews)

  - The coastal focus criterion includes studies:
    * directly and generally linked to coastal areas and processes
    * investigating specific Baltic coastal areas (dark purple in Figure 1.1)
  - The eutrophication focus criterion includes studies:
    * focusing solely on eutrophication and its biogeochemical processes (e.g., algae blooms)
    * focusing on eutrophication in relation to other issues and processes
    * focusing on water quality and ecosystem status where eutrophication is relevant

- 1855 studies screened on title and abstract against the three inclusion criteria:
  - 359 studies excluded for not meeting one of the criteria
- Retrieval of the full text for 1496 studies:
  - 11 studies excluded for lack of full text
- 1485 studies screened on full text against the three inclusion criteria:
  - 653 studies excluded for not meeting one of the criteria
- 832 studies included for data extraction.

The data extraction process has been carried out by a manual categorization of each scientific paper via the following steps:

- Definition of the data extraction categories (shown in Figure 3.5), representing topics, methods, subsystems, issues and processes mentioned or considered by the study in relation to coastal eutrophication, and of the categorization criteria (Table 1 in SM paper IV).
- First categorization of the 832 included studies, mainly based on title, abstract and keywords to focus on the main messages, but also on full text when clarification have been necessary.
- Update of the data extraction categories to account for missing categories.
- Second categorization of the included studies to account for the updated categories and to clarify studies where doubts subsisted.

3.3.2 Data structure and analysis

The final categories and their structural organization are presented in Figure 3.5 and fully described in Paper IV SM Table 1. The topic super-categories "Eutrophication drivers" (in short, Drivers), "Ecosystem impacts of eutrophication" (Impacts), and "Characterization of eutrophication conditions" (Characterization) represent the main eutrophication focus, and each study is classified in at least one of these super-categories. Three category sets have further been defined, which are used to categorize the studies in terms of: (i) Methods of coastal system study (Methods); (ii) Coastal system links to other water systems (on land and the open sea) (Coastal links); and (iii) Coastal system components and aspects (Coastal system). These category sets are further structured in terms of main categories and related sub-categories displayed in Figure 3.5, and the share (percentage) of

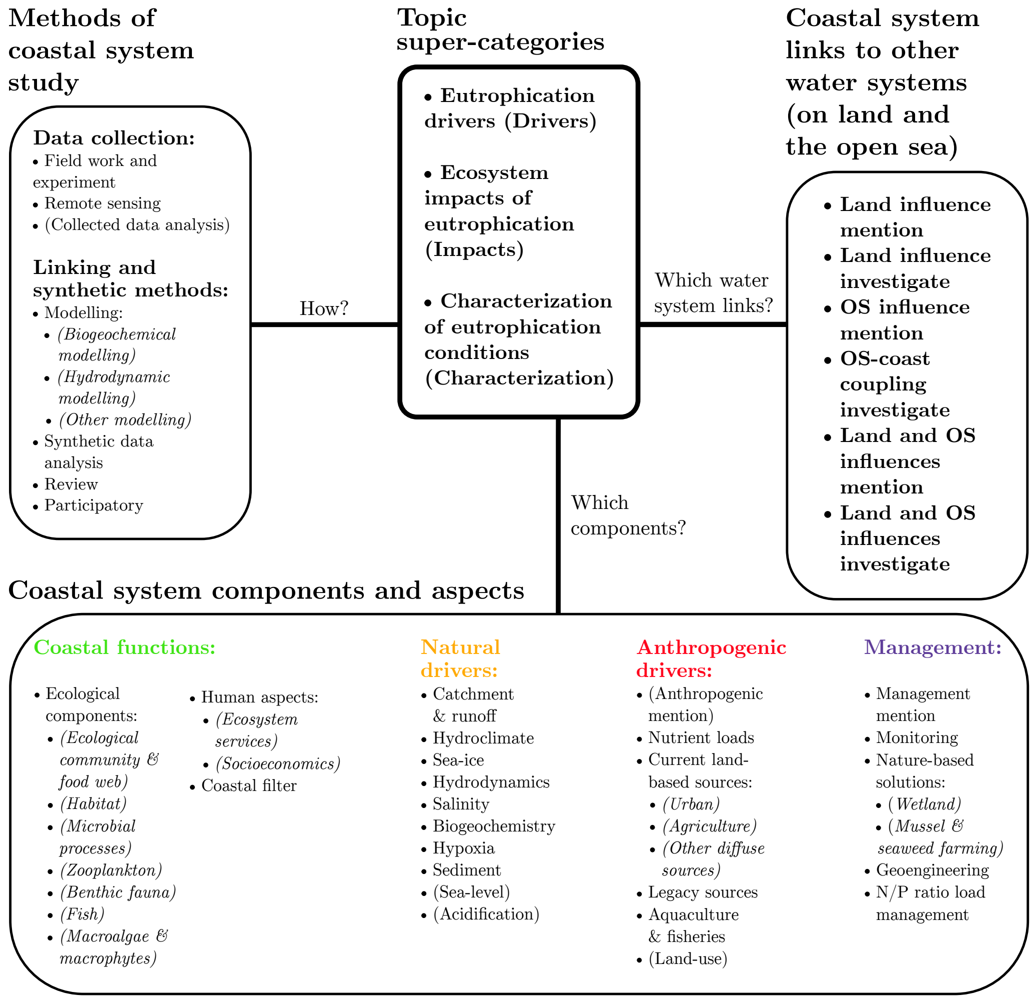

Figure 3.5. Schematic representation of the structure of the topic categories considered in the Paper IV literature review. Each scientific publication is categorized into super-categories, category sets and their main categories (in bold inside the boxes), sub-categories and sub-sub-categories (in italic). Categories in parentheses are not shown in the analysis but are still considered through their respective higher category levels (the full results including these categories are shown in Paper IV SM Figure S1; modified from Paper IV Fig. 2).

publications associated with each topic super-category, main category and sub-category has been calculated.

Undirected network diagrams are used to analyse and visualize the categories (weight depending on their research frequency) and connections between category pairs (weight depending on the frequency of papers classified in both categories). The weighted and undirected network graph has been implemented in the graph database Neo4j (Cattuto et al., 2013). The Louvain algorithm (Blondel et al., 2008), which is used to detect network communities by optimizing the modularity (relative measure of the edge density inside versus outside a community), has been applied to this weighted network graph using Neo4j to compare the manual categorization structure and classification with the arising Louvain topic communities.

4 Results

4.1 Model validation

Figure 4.1 shows the comparison between model results and observation data, quantified for the Baltic MBs over the validation period (2002–2008) using Taylor diagrams (Taylor, 2001). The Taylor diagrams show that the model representation of surface DIN (Figure 4.1, first column) is good for most MBs, with correlations greater than 0.8 and standard deviations of modelled results close to those of observations. For surface DIP (Figure 4.1, third column), the correlations are good for most MBs, around or greater than 0.7, but the modelled variabilities are underestimated for most MBs (between 0.5 and 1 of the observation standard deviations), indicating that the model still captures most of the variability. For deep DIN (Figure 4.1, second column), the model representation is acceptable with correlations around or greater than 0.5 and modelled standard deviations around 0.5 of the observation ones for most MBs. For deep DIP (Figure 4.1, fourth column), the model representation is poor, with generally low correlations and normalized standard deviation, indicating that the model does not capture most of the deep DIP variability. These results show that the used relatively simple approach to water quality modelling performs well in representing surface nutrient and algae levels and dynamics, but the deep nutrient dynamics are generally fairly (DIN) to poorly (DIP) represented and their variabilities are underestimated. The discrepancies are strongest for the Baltic Proper MBs, which can be explained by the strong stratification of deeper waters in these MBs and by processes pertaining to permanently anoxic areas that are not accounted for in the model. Overall, the validation supports the ability of the modelling approach applied to the Baltic Sea to represent surface water quality conditions and their short- and medium-term dynamics for management purposes, and for further modelling use in setting local-scale coastal boundary conditions.

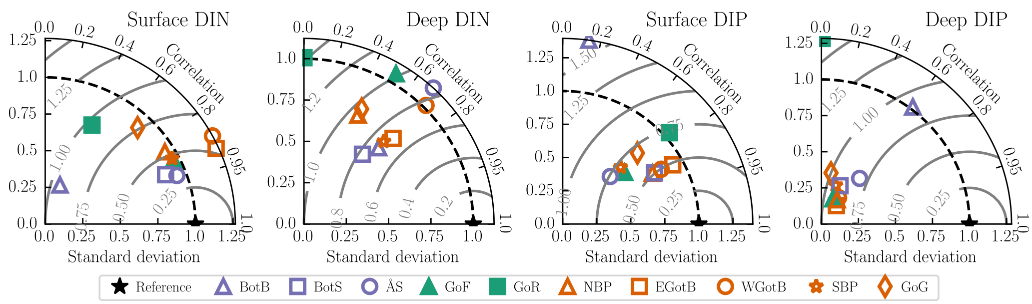

Figure 4.1. Taylor diagrams comparing modelled results with observations for all MBs with monitoring data for the validation period (2002–2008). First column: surface DIN; second column: deep DIN; third column: surface DIP; fourth column: deep DIP. The standard deviation of the model results is normalized by that of the observation data for each MB, and shown by the abscissa and ordinate. The angular coordinate shows the model result correlation with observations and the distance between the observation (abscissa, black star) and the model points is proportional to the root-mean-square error between model results and observations. BotB: Bothnian Bay, BotS: Bothnian Sea, ÅS: Åland Sea, GoF: Gulf of Finland, GoR: Gulf of Riga, NBP: Northern Baltic Proper, EGotB: Eastern Gotland Basin, WGotB: Western Gotland Basin, SBP: Southern Baltic Proper, GoG: Gulf of Gdansk (modified from Paper I Fig. 3).

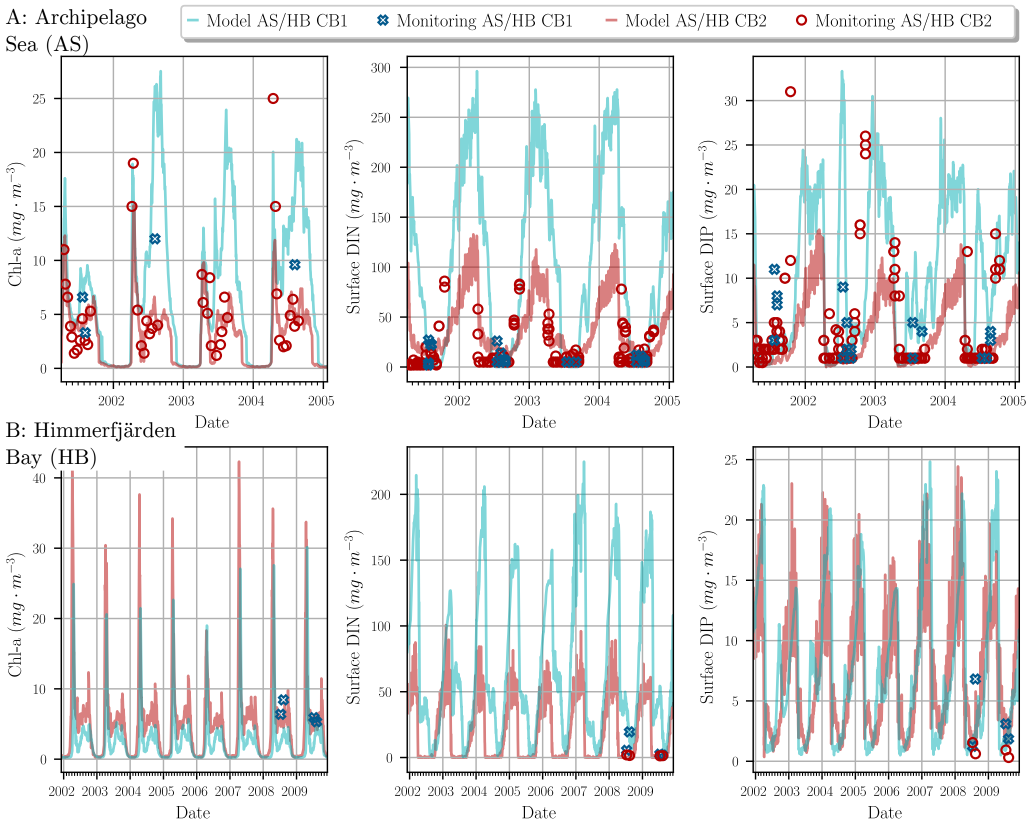

Figure 4.2. Comparison of modelled water quality dynamics for Surface Chl-a (first column), Surface DIN (second column) and Surface DIP (third column) with available observation data for the Archipelago Sea (Panel A) and the Himmerfjärden Bay (Panel B) for the simulation periods 2001–2005 and 2002–2009, respectively. For the Archipelago Sea (Panel A), the model-observation comparison is done for the central archipelago CB (CB2, red line for model results and red circle for observations from Nau 2361 Seili intens) and the eastern inner CB (CB1, blue line for model results and blue cross for observations from Pala 115 Tryholm). For the Himmerfjärden Bay (Panel B), the comparison of model results with observations is done for the central CB (CB1, blue line for model results and blue cross for observations) and the outer CB (CB2, red line for model results and red circle for observations). The validation CBs for the Archipelago Sea and Himmerfjärden Bay are represented in blue (CB1) and red (CB2) in Figure 3.2C and A, respectively.

On the local coastal scales, model validations are more challenging due to lower data availability. For Paper I, a precise validation of the Archipelago Sea model is not possible since nutrient loads from 2000 have been used to run the model for the period 2001–2005. Model results are still compared to available monitoring data in Figure 4.2, Panel A to check how the main water quality dynamics of the system are represented by the model at two locations, one situated in the eastern inner (CB1, blue line) and the other in the central (CB2, red line) parts of the archipelago (Figure 3.2, Panel C). For the central CB (red line and red open circles), modelled Chl-a levels and variations are in good accordance with measurements during spring and summer. Surface DIN and DIP levels and dynamics are also fairly well represented, with surface DIP being underestimated in the beginning of winter, and close to the monitoring data before the spring blooms. The modelled increases in surface DIN and DIP concentrations during late autumn are out of phase with the monitoring data, which could be due to a too long growing season and delayed algal mineralization. For the inner CB (CB1, blue line and blue open cross), receiving high nutrient loads from a nearby river, monitoring data are only available during the summer. The model captures both the very low surface DIN concentrations and the higher variabilities in DIP, with the latter possibly being due to internal P release from seasonal hypoxia. The few Chl-a measurements indicate an overestimation of summer

Chl-a concentrations by the model, possibly due to too early (July-August) modelled P release from anoxic sediments compared with oxygen data (August-September).

For the Himmerfjärden Bay model (Paper II), open monitoring data are only available for four CBs (two of which are presented in Figure 4.2, Panel B, for the central (CB1, blue curve) and outer (CB2, red curve) CBs of the bay). Only two summer measurements of DIN, DIP and Chl-a (only for the central CB) are available during 2008 and 2009, making an assessment of water quality dynamics and winter nutrient levels impossible. Chl-a values during the summer are underestimated by the model for the central CB. Summer DIN values are overestimated by the model, while the spatial variations are well captured, with higher values in the central CB, influenced by the nutrient loads from the Södertälje canal, and lower value for the outer CB, receiving no riverine or point source loads. Summer DIP concentrations are generally well in accordance with the available monitoring data.

4.2 Land-coast-sea interactions and effects on management measures

Figure 4.3 shows the area-weighted distribution of reduction in Winter DIN (left) and DIP (middle), and Summer Chl-a (right) for the various land-based, sea-based, coastal and combined scenarios compared to the baseline scenario for the Archipelago Sea (Paper I, Panel A) and the Himmerfjärden Bay (Paper II, Panel B). For the Archipelago Sea, the results represent transient effects of possible management measures on water quality, averaged over the three years of simulation. The local land-based scenario, assuming a 50 % reduction of the nutrient loads from the WWTP, yields a maximum reduction of Winter DIN of 20 %, with a only few CBs reaching around 10 % of reduction, and most of the Archipelago areas not experiencing any substantial improvements (mean and median close to 0). Both the sea-based and the combined scenarios result in a maximum decrease of Winter DIN of around 45 %, with 75 % of the Archipelago Sea area reaching reductions of more than 4 % and 11 %, respectively. Thereby, for Winter DIN, the land-based scenario has strong local effects to some degree spread towards the outer CBs, while the sea-based scenario has regional effects with substantial spreading inwards to the coast. For Winter DIP, the local land-based scenario leads to only small improvements, and also to slight increases in Winter DIP (2 % or less) at a few CBs, possibly due to changes in internal dynamics that may also influence nutrient transport. The sea-based and combined scenarios yield similar results, with Winter DIP reduced by more than 5 % over 75 % of the Archipelago Sea, and maximum reductions of around 50 %. For Summer Chl-a, the land-based scenario also yields just small improvements, with concentrations decreased by around 10–20 % for only a few CBs. The sea-based and combined scenarios lead to 75 % of the Archipelago achieving reductions higher than 16 % and 18 %, respectively, with maximum reductions around 55 %. Thereby, the land-based scenario has even more localised effects for Summer Chl-a and Winter DIP than for DIN.

For the Himmerfjärden Bay (Paper II, Figure 3.2, Panel A), pseudo steady state nutrient and algae concentrations averaged over the three hydroclimatic condition-years are used to calculate area-weighted concentration reductions. For Winter DIN, the land-based scenario S_{PS+R} yields the greatest decrease (up to 35 %), with reductions of more than 17 % over 75 % of the bay area. Both the sea-based and the coastal aeration scenarios increase Winter DIN by around 3 % on average for the bay. However, the combined land- and sea-based scenario yields concentration decreases by 19–33 % for 75 % of the bay area. For Winter DIP, the land-based and coastal scenarios have limited local impacts, with an average decrease of 5 % and substantial improvements only for some CBs.

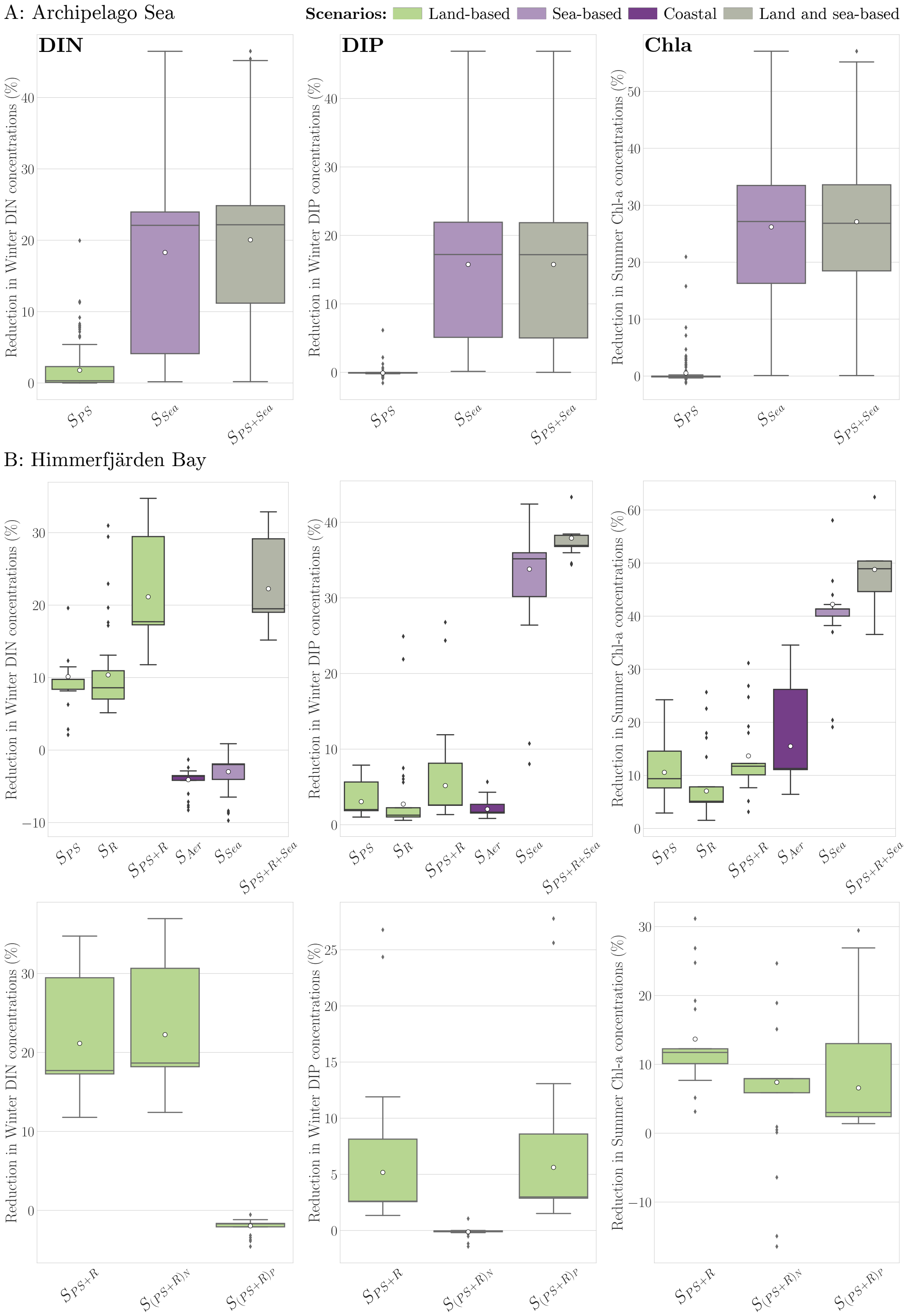

Figure 4.3. Percentage of reduction in Winter DIN (left), Winter DIP (middle) and Chl-a (right) for the various land-based, coastal, sea-based and land- and sea-based management scenarios compared the baseline scenario for the Archipelago Sea (Panel A) and the Himmerfjärden Bay (Panel B). For the Archipelago Sea (Panel A), the reduction is calculated using average Winter DIN and DIP and Summer Chl-a model results for the three years of simulation (transient effects). For the Himmerfjärden Bay (Panel B), the reduction is calculated using Winter DIN and DIP and Summer Chl-a model results averaged over the 5 last simulation years (pseudo steady-state) and for the three hydroclimatic condition-years. The last row (Panel B) focuses on reduction results when mitigating only N and only P (comparing land-based scenarios S_{PS+R}, $S_{(PS+R)_N}$, $S_{(PS+R)_P}$).

The sea-based scenario yields higher concentration decreases by 30 % or more for 75 % of the bay area, with only a few CBs reaching less than 15 % reduction. This is improved for the combined land- and sea-based scenario, which yields Winter DIP decreases of 34–43 % over the whole bay. Summer Chl-a effects fall between those for Winter DIN and Winter DIP. For this variable, the land-based scenario S_{PS+R} yields concentration decreases by 10–30 % for 75 % of the bay area, and the coastal aeration scenario performs somewhat better with reductions up to 35 %. The sea-based scenario results in Summer Chl-a decreasing by more than 40 % for 75 % of the bay area, with reductions of 20 % only for a few CBs, for which the conditions are improved under the combined scenario (36–62 % reductions over the whole Himmerfjärden Bay).

Results for Archipelago Sea and the Himmerfjärden Bay are not directly comparable, due to different spatial scales (the modelled area of the Himmerfjärden Bay is approximately one order or magnitude smaller than that of the Archipelago Sea), temporal scales (transient versus steady state results), and the considered load and concentration reductions for the land and sea-based scenarios. Nevertheless, some similarities still emerge from these two case studies. The land-based scenarios S_{PS} yield approximately similar absolute nutrient N and P load reductions and similar ranges of concentration reductions for the Winter DIN and DIP, and Summer Chl-a variables. The area-weighted average reduction is much lower for the Archipelago Sea case, due to its larger spatial scale. In both cases, the greatest reduction is achieved for Summer Chl-a, along with greater average reductions for Winter DIN and Summer Chl-a than for Winter DIP. This indicates that the effects of land-based load reductions propagate further towards the outer coast for Summer Chl-a and Winter DIN than for DIP. The effects of the regional sea-based measures vary between the two cases, due to different concentration reductions at the coastal boundaries to open sea. The differences are strongest for Winter DIN, explained by a reduction of boundary DIN concentrations of approximately 50 % in the Archipelago Sea case, and only small reductions in the Himmerfjärden Bay case. Thereby, the effects on Winter DIN in the Himmerfjärden Bay are due to changes in the internal dynamics from improved boundary concentrations of P and algae. Boundary concentration reductions in P and algae are similar in both case studies and yield important reductions of both Winter DIP and Summer Chl-a, with differences in ranges that can be explained by the differences in spatial and temporal scales. The inner CBs of the Archipelago Sea are situated further away from the open sea than those of the Himmerfjärden Bay, implying longer propagation times for sea-based measure effects on the Archipelago Sea. The sea-based scenarios also depend on actual land-based and possible other remediation measures applied over the whole Baltic catchment scale. This coupling between the local coastal scales and the whole open sea and associated regional catchment scale is discussed in more detail in Section 5.1.

In both case studies, reducing the land-based N and P loads by the same proportion leads to different effects on Winter DIN and DIP, and Summer Chl-a, which propagate further for DIN and are more localised for DIP. For the transient Archipelago Sea simulations, this could be explained by the longer residence time of P in the system (stronger sediment feedbacks). However, similar results are also obtained from the pseudo steady-state simulations for the Himmerfjärden Bay, which indicates that land-based sources generally have more influence on DIN than DIP dynamics. To test this, simulations considering 50 % reduction of the land-based loads of only N or only P have also been carried out for the Himmerfjärden Bay (Figure 3.2, Panel B last row). The results show that reducing only N loads yields average Winter DIN decease by 20 %, while reducing only P loads yields average Winter DIP decease by only 5 % and even somewhat increases Winter DIN. The latter result indicates a build-up of DIN in the coastal waters, possibly due to less denitrification, which also explains the Winter DIN increase in the sea-based scenario. For Summer Chl-a, reducing only N or only P loads yields similar results on average over the whole-coast scale (7 % and 6 % decrease, respectively). However, reducing only N loads yields locally increased Summer Chl-a in several CBs, which is may be due to nutrient imbalances that favour N-fixing cyanobacteria. Thereby, optimizing nutrient load reduction in relation to the local coastal water quality dynamics is needed to avoid negative environmental effects resulting from both increased DIN export to the open sea waters, such as experienced in the Wadden Sea and Norwegian Coast (Conley et al., 2009c), and increased coastal cyanobacteria blooms (Chen et al., 2013).

4.3 Analysis of internal dynamics and land-coast-sea drivers

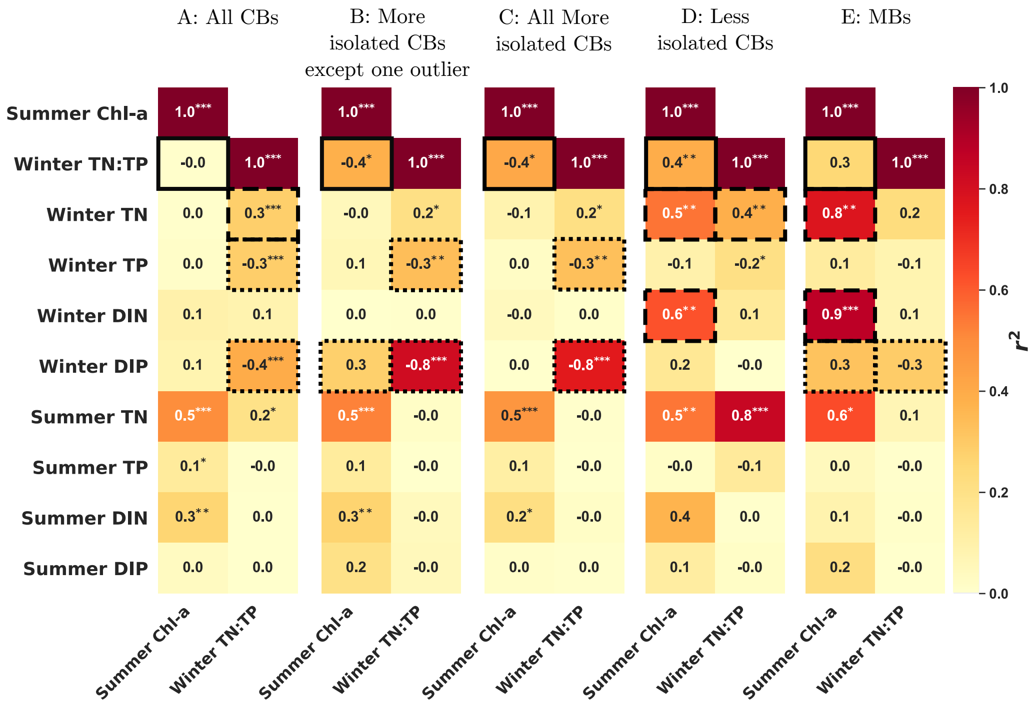

Figure 4.4. Resulting coefficient of determination (r^2) from trend correlations in water quality variables, focusing on the main water quality variables from Paper III (Summer Chl-a and Winter TN:TP), for the coastal and marine classes; A: all Coastal Basins (CBs), B: the more isolated CBs except the outlier CB Inre Oskashamnsområdet (shown in Paper III SM Figure S1), C: all the more isolated CBs, D: the less isolated CBs, and E: the Marine Basins (MBs). Relations between the main variables Summer Chl-a and Winter TN:TP are circled in plain black line. Main relations of Summer Chl-a and Winter TN:TP with Winter N and P variables are circled in dashed and dotted lines, respectively. The number in the cell indicates r^2. - sign indicates a negative correlation. *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$ (modified from Paper III Fig. 3).

In Paper III, the internal water quality dynamics in the coastal and marine classes of more isolated CBs, less isolated CBs, all CBs, and MBs are investigated for medium to long time periods (20–30 years) by correlating the trends in water quality variables with each other. The main results, focusing on Summer Chl-a as the main proxy of eutrophication, are displayed Figure 4.4, which shows how much variability in the trends of one water quality variable can be explained by the variability in the trends of another variable. For all classes, the trends in Summer Chl-a correlates with trends in Summer TN and DIN. These are both directly dependent on summer algal growth and levels (e.g., through N-fixation), and thus add limited information on driving nutrients for al-

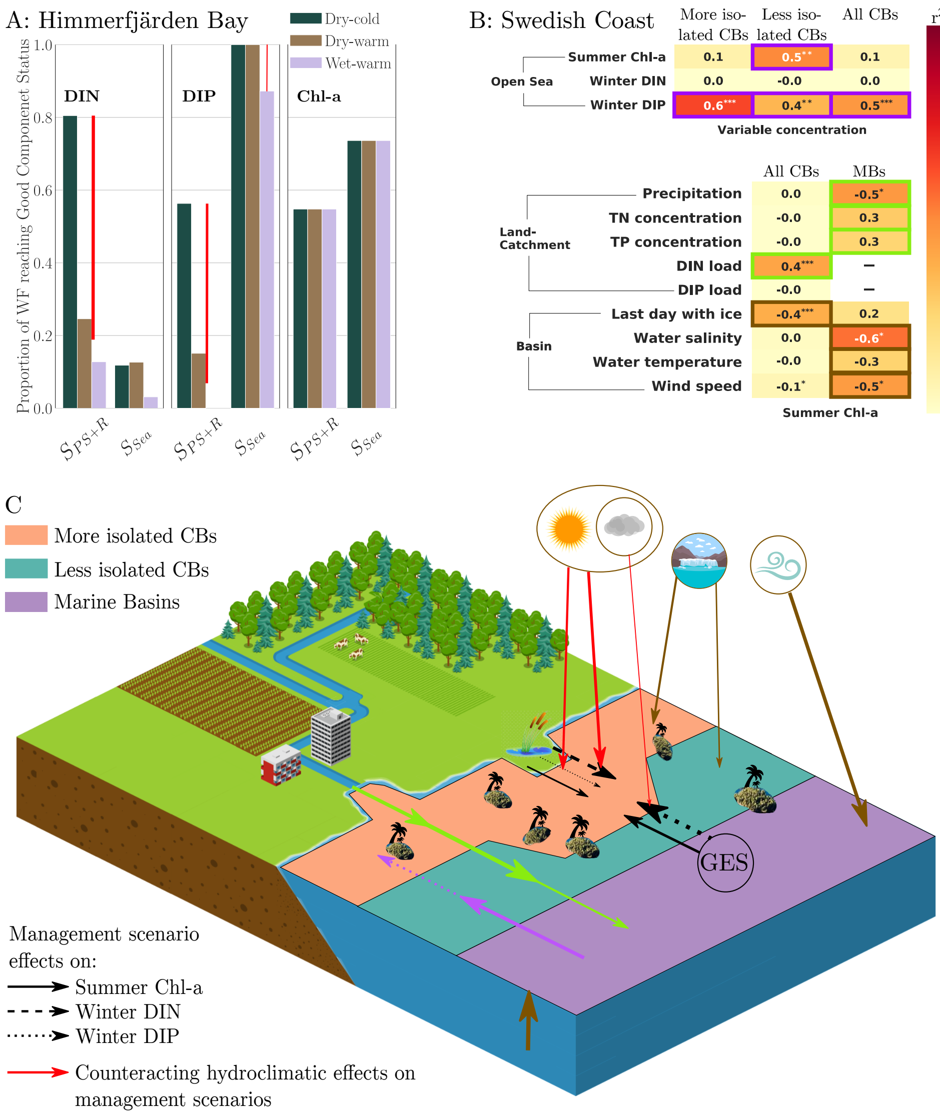

Figure 4.5. Panel A: Results from Paper II showing the proportion of the Himmerfjärden Bay area reaching Good Component Status for Winter DIN (left), Winter DIP (middle), and Summer Chl-a (right) under the land-based scenario S_{PS+R} and the sea-based scenario S_{Sea}. Panel B: Results from Paper III showing the coefficient of determination (r^2) from correlations of trends in Summer Chl-a, Winter DIN and DIP with the corresponding open sea variables for the coastal classes (more isolated CBs, less isolated CBs and all CBs), and from correlations of trends in Summer Chl-a with main identified drivers for all CBs and the MBs. Main correlations between Summer Chl-a trends and driver trends are circled (in green for land-catchment drivers, brown for coastal and marine basin drivers and purple for open sea drivers). The number in the cell indicates r^2. - sign indicates a negative correlation. *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$. Panel C: Schematic illustration of the main drivers and their dominant influences on the coastal and marine scales (green arrows for land-catchment drivers, brown arrows for coastal and marine basin drivers and purple arrows for open sea drivers) and of the management measure effectivenesses (Paper II, black arrows) and hydroclimatic impacts on these (red arrows).

gae growth. For the more isolated CBs (Panel C), Summer Chl-a trends also correlate moderately (r^2 of 0.4), significantly and negatively with trends in the Winter TN:TP ratio (circled in black plain line), which in turn correlates very strongly and negatively with trends in Winter DIP (circled in black dotted line). Trends in Summer Chl-a would thus be expected to correlate positively with trends in Winter DIP, for which a moderate correlation emerges only when removing an outlying CB from the analysis (Paper III, SM Figure 1). For both the less isolated CBs and the MBs (Panels D and E), trends in

Summer Chl-a exhibit similar relationships with trends in the other variables, correlating strongly (r^2 of 0.5 or higher), significantly and positively with trends in Winter DIN and TN (circled in black dashed line). They also correlate moderately and positively with trends in Winter TN:TP, which are further moderately correlated to trends in Winter TN for the less isolated CBs. Summer Chl-a dynamics differ between the more and less isolated CBs, exhibiting opposite correlations with trends in Winter TN:TP and indicating mixed dynamics, somewhat dominated by P for the more isolated CBs and by N for the less isolated CBs. In the Himmerfjärden Bay (Paper II), for which all CBs are classified as more isolated, only N or only P reductions were found by modelling to yield similar reductions in Summer Chl-a (Figure 4.3, Panel B second row), in consistency with the more mixed dynamics emerging from trend correlation analysis for this coastal class. The opposite Summer Chl-a dynamics between the more and less isolated CBs also explains the absence of correlation between Summer Chl-a and other water quality variables than Summer TN and DIN when considering all CBs (Panel A).

In Paper II, the effects of hydroclimatic conditions on management measure effectiveness are investigated by considering three hydroclimatically distinct (dry-cold, dry-warm and wet-warm) condition-years. Figure 4.5 (Panel A) shows the proportion of the Himmerfjärden Bay area reaching Good Component Status (GCS) for Winter DIN and DIP, and Summer Chl-a calculated for the land-based S_{PS+R} and sea-based S_{Sea} scenarios under pseudo steady-state conditions that isolate and emphasize the hydroclimatic trend effects. The land-based scenario S_{PS+R} yields strong improvement of the DIN status under dry-cold conditions, with around $80\,\%$ of the bay reaching GCS, and more limited improvements under dry-warm and wet-warm conditions. It also yields some improvements of the DIP status under dry-cold conditions, with GCS reached in $55\,\%$ of the bay, but almost no improvements under warmer conditions. However, the land-based scenario only improves Chl-a status under dry-warm and wet-warm conditions, indicating a spatially limited propagation of land-based effects for Summer Chl-a. The sea-based scenario does not improve the DIN status under any hydroclimatic conditions, but allows the whole bay to reach GCS for DIP under drier conditions and around $85\,\%$ of the bay to reach GCS for DIP under wetter conditions. It also results in $75\,\%$ of the bay reaching GCS for Chl-a under all hydroclimatic conditions. Thereby, these results indicate that warming and wetting conditions reduce the effectiveness of management scenarios, with only scenarios that include the sea-based measures improving DIN and DIP status. This complicates coastal eutrophication management in comparison to dry-cold conditions. Management scenario effects on coastal water quality are indicated schematically in Figure 4.5, Panel C (plain, dotted and dashed black arrows for Summer Chl-a, Winter DIP and DIN, respectively), together with counteracting hydroclimatic effects on these scenarios (red arrows).

In Paper III, trend correlations between selected water quality variables (Summer Chl-a and Winter TN:TP) and potential drivers acting on different parts of the land-coast-sea system identify main drivers of coastal eutrophication (as indicated in Figure 4.5, Panel B). The trend correlation results also test whether the effects seen in scenario simulations for the Himmerfjärden Bay (Paper II) can be observed and generalized through data analysis. Changes in open sea Summer Chl-a conditions are strongly, positively and significantly associated with changes in Summer Chl-a for the less isolated CBs (circled in purple, and shown by purple arrow in Figure 4.5 Panel B and C, respectively). Moreover, trends in coastal Winter DIP are also strongly (r^2 of 0.5) and significantly correlated with trends in open sea Winter DIP across all coastal classes, but changes in coastal Winter DIN are not associated with corresponding changes in the open sea. This indicates a strong role of open sea conditions for coastal P dynamics, and supports the limited effects of simulated land-based measures on DIP conditions for the Himmerfjärden Bay.

For the coastal waters, relations between Summer Chl-a and land-catchment drivers are dominated by nutrient loads (circled in green in Figure 4.5 Panel B). DIN load trends, especially, correlate strongly (r^2 of 0.4–0.5), positively and significantly with trends in Summer Chl-a for all CBs. This effect of DIN loads for all CBs is also consistent with the scenario simulation results for the Archipelago Sea and Himmerfjärden Bay, indicating that effects of land-based DIN reductions propagate further out the coast than those of DIP reductions. Thereby, land-based DIN loads play an important role for coastal N dynamics. On the regional scale, trends in Summer Chl-a for the MBs are not correlated with loads but moderately with nutrient concentrations, possibly reflecting the strong negative correlation of Summer Chl-a with precipitation over the associated MB catchment.

For the MBs, changes in Summer Chl-a are principally associated with changes in the marine basin driver, acting directly over or in the MBs (circled in brown in Figure 4.5, Panel B). Water salinity and wind speed are both strongly and negatively correlated with trends in Summer Chl-a (brown arrows in Figure 4.5, Panel C), possibly indicating stratified and calm conditions that favour cyanobacteria blooms (Wasmund et al., 2005). For the coastal basin drivers, only sea-ice correlates moderately (r^2 of 0.4) and significantly with Summer Chl-a for all CBs, with increasing sea-ice conditions associated with decreasing Summer Chl-a. Even though the simulations have not considered the direct effects of sea-ice, such as the formation of nutrient plumes that can affect the spring bloom and phytoplankton species composition (Kari et al., 2018; Granskog et al., 2005; Haecky et al., 1998), sea-ice conditions may be a relevant proxy for hydroclimatic conditions that affect coastal eutrophication.

4.4 System interlinkages in coastal eutrophication research

Results from Papers II and III show that various anthropogenic and hydroclimatic drivers influence coastal eutrophication, such as local nutrient loads and sea-ice conditions. Moreover, open sea conditions, which are also influenced by hydroclimatic (e.g., wind speed) and hydrospheric (e.g., water salinity) drivers, also affect coastal water quality. Thereby, coastal waters are simultaneously affected by a melting pot of drivers and influences from the land, coast and open sea parts of the system. To study how coastal eutrophication research reported in the scientific literature has handled these interlinked drivers and influences on coastal functions, and eutrophication and its management, a literature screening that quantifies the associated research effort is carried out in Paper IV. Figure 4.6 presents the main results, showing the connections between the super-categories (drivers, impacts, characterization) of coastal eutrophication and the main categories and sub-categories in each category set (coastal links, coastal system). In terms of the topic super-categories, fewer studies focus on characterizing the eutrophication conditions (blue super-category in Figure 4.6; $20\,\%$ of all publications) than on investigating ecosystem impacts (yellow super-category; $39\,\%$) and drivers (red super-category; $52\,\%$).

Coastal eutrophication research mentions the coastal influences from land ($31\,\%$) and the open sea ($43\,\%$), indicating that the importance of these links is generally understood. However, much fewer studies actually investigate these links ($10\,\%$ for either land or open sea), and then mostly in driver studies ($15\,\%$ for either) and much less in impact studies (less than $5\,\%$ for either).

The main system components (anthropogenic drivers, natural drivers, coastal functions, and management) are generally well-investigated with more than $45\,\%$ of all publications considering or mentioning each of them. Characterization research (blue) is

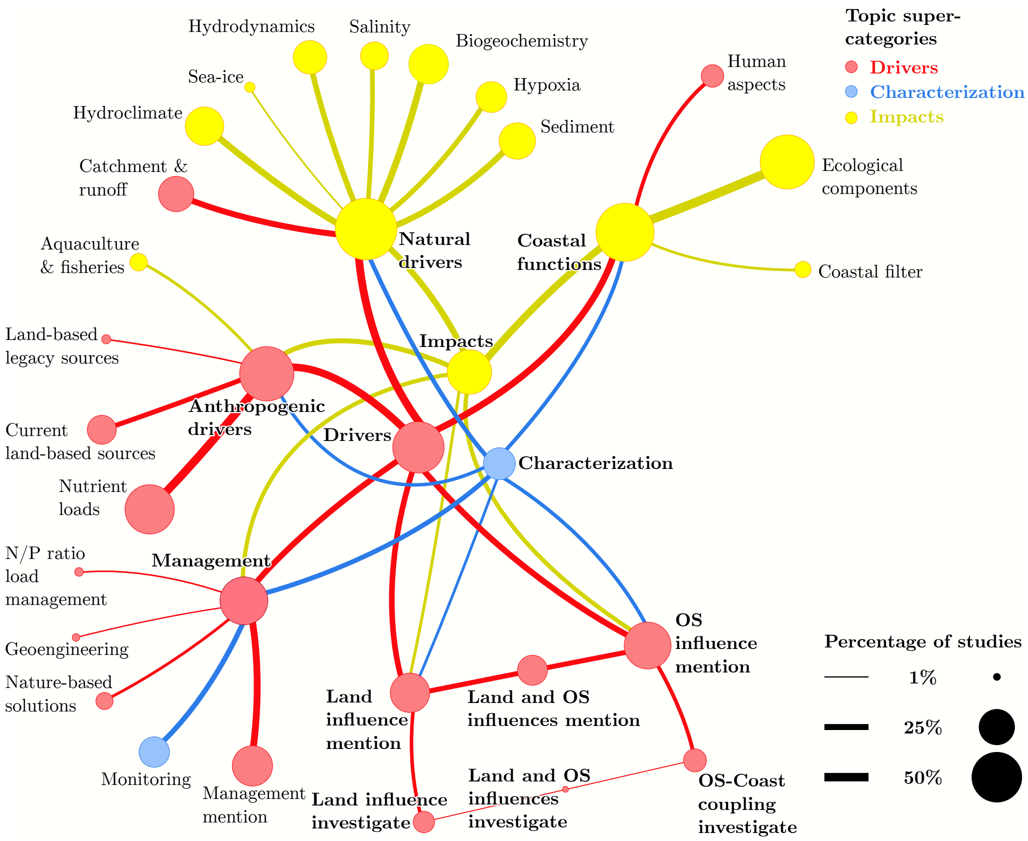

Figure 4.6. Research effort and study links between the Drivers (red), Impacts (yellow), and Characterization (blue) super-categories and the main and sub-categories in the three overarching category sets (Coastal links; Coastal system; category structure shown in Figure 3.5). The size of each circle and link is proportional to the square root of the percentage of publications classified in that category and in both categories, respectively. The Louvain algorithm is used to classify the categories into communities, represented by their node colour (blue: drivers, yellow: impacts, red: characterization).

only associated with monitoring, while impact research (yellow) considers both coastal functions and natural drivers. Driver research (red) is more associated with management and anthropogenic drivers, but considers also natural drivers and coastal functions (in 80 % and 60 % of driver studies, respectively). These results show that Baltic coastal research spans over the complexity of coastal eutrophication, represented in Figure 4.6. However, few studies investigate both drivers and ecosystem impacts of coastal eutrophication together, indicating that some crucial couplings between the main system components, and between them and the system components sub-categories, may be missed or under-investigated in the research.

With regard to specific driver, most of the natural drivers sub-categories have received similar research effort, considered in 15–30 % of the studies (Figure 4.6). Sea-ice, on the other hand, is understudied in relation to coastal eutrophication, even though sea-ice conditions have been found to be robustly negatively correlated with Summer Chl-a conditions in Paper III. Sea-ice can also lead to the formation of nutrient-rich under-ice freshwater plumes (Granskog et al., 2005) and further affect phytoplankton species composition (Haecky et al., 1998). Hydroclimate is relatively well-investigated (29 %) and has been found to influence coastal (Paper II) and marine eutrophication (Paper III). Catchment & runoff is the only natural driver type that is more strongly associated with the driver super-category (red in Figure 4.6; considered in 35 % of driver publications) than with the impact super-category (yellow; considered in 9 % of impact publications),

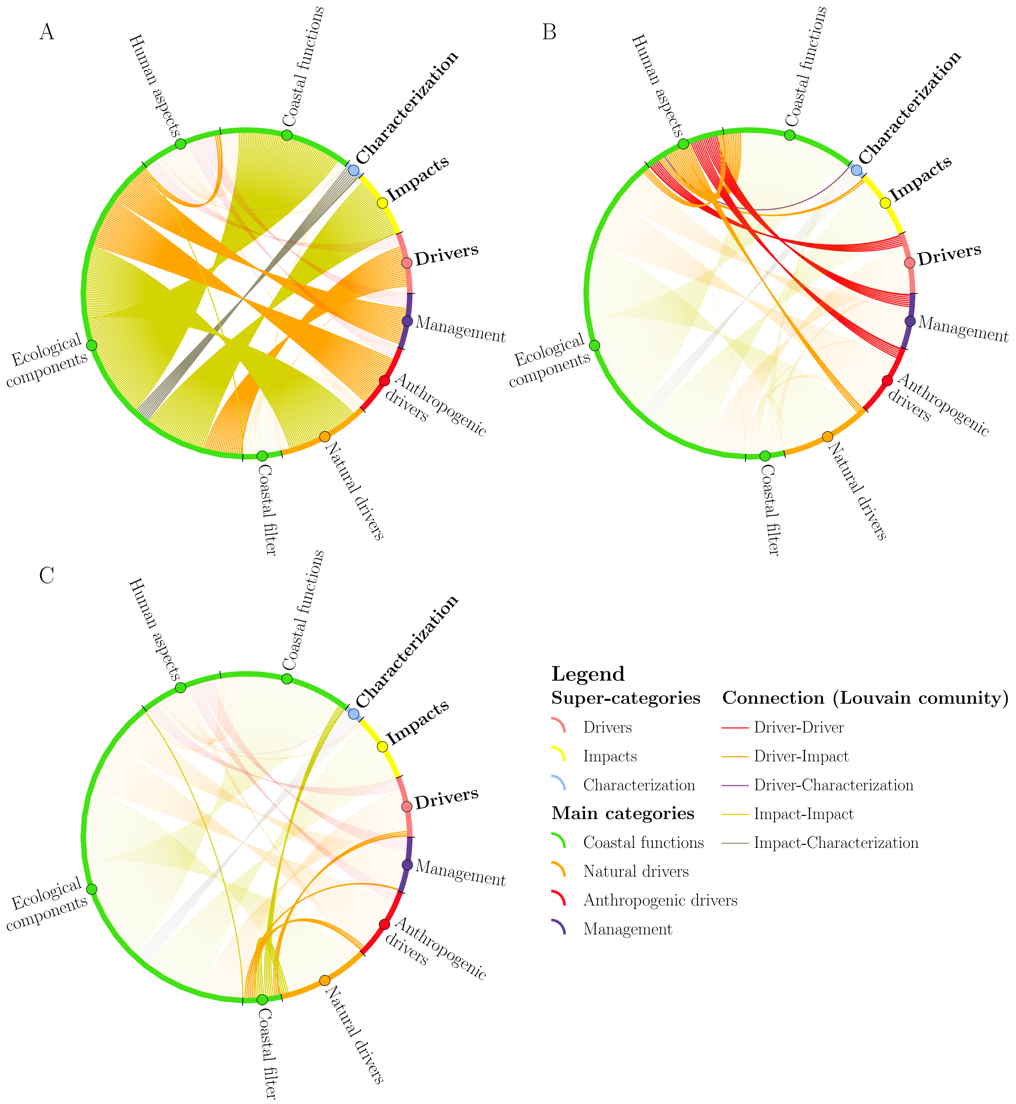

Figure 4.7. Research connections among the coastal functions sub-categories of human aspects (Panel A), ecological components (Panel B), and coastal filter (Panel C), and between these and the super-categories and other main categories. The node perimeter of a category is proportional to the percentage of study linking the category with any other category in the diagram (relative degree of the node). The number of links between two categories is proportional to the percentage of publication classified in both categories, and the colour of the link depends on the topic super-category associated with each node (see Louvain classification in Figure 4.6). Links considering less than 1% of publications are not shown.

even though the chemistry of freshwater discharges can impact ecological communities and food webs (e.g., humic substances favouring bacteria over phytoplankton; Andersson et al., 2015). Anthropogenic drivers are dominated by nutrient loads, considered in 50 % of all publications and well linked to both natural and anthropogenic drivers. Much fewer studies consider specific nutrient sources on land. Current active land-based sources are mentioned in around 20 % of the studies, while legacy sources are only mentioned in 1 % even though they have emerged as one of the main pathways of nutrient loading that complicates coastal eutrophication management (Le Moal et al., 2019). Aquaculture & fisheries are also understudied in relation to coastal eutrophication and more strongly associated with impact (yellow super-category in Figure 4.6) than driver (red super-category) studies, even though they can modify the food web in addition to nutrient loads.

Coastal eutrophication also interacts with the ecological, human and biological functions of the coastal zone, e.g., by affecting species distribution, and being impacted by changes in ecological communities (e.g., fish; Bergström et al., 2019). The research links between the coastal function sub-categories, and between them and the super- and main coastal component categories are shown in Figure 4.7. The ecological components, representing communities and their interactions, dominate the research on coastal functions (considered in 58 % of all publications, Panel A) in comparison to human aspects (Panel B). The latter, representing human considerations of the coast, and remediation-contributing coastal attenuation of nutrients (also referred to as natural remediation, or coastal filter, Panel C) are considered in 10 % and 5 % of the studies, respectively. The ecological components are strongly associated with impact studies, considered in 89 % of them, but are also considered in 50 % of both driver and characterization studies. These ecological components are more often investigated together with natural drivers (40 % of all publication considering both) than with anthropogenic drivers (29 %) and management (20 %). Human aspects (driver super-category) are instead more investigated in studies of anthropogenic drivers and management, than in impact and characterization studies, with few connections to natural drivers and other coastal functions. Natural remediation of nutrient loads is almost only investigated in driver studies, together with anthropogenic and natural drivers. These results show that the ecological components and the human and natural remediation aspects of the coastal system are seldom linked in coastal eutrophication research.

Integrated management of coastal eutrophication necessitates understanding of the complex interactions and, feedbacks between drivers and impacts of coastal eutrophication. However, only 6.5 % of studies investigate both eutrophication drivers and impacts. Moreover, even though management is associated with the driver super-category and is well mentioned in the research (33 %), much fewer studies actually investigate specific management solutions such as nature-based solutions (mussel & seaweed farming, wetlands) or geoengineering methods. Moreover, very few studies mention the importance of controlling the N:P ratio of nutrient loads, which can affect the nutrient export to the sea and also impact ecological communities (Elmgren and Larsson, 2001). Finally, the main management category is less linked to the impact super-category than to the driver and characterization ones, even though managing coastal eutrophication requires consideration of the coastal ecosystem interactions, impacts, and feedbacks with human aspects.

5 Discussion

5.1 Land-coast-sea system and scale couplings

This thesis has investigated the propagation of water quality changes over the land-coast-sea continuum through scenario analysis in Papers I and II, and trend analysis in Paper III. This investigation has been limited to archipelagoes and bays of the Archipelago Sea and along the Swedish coast. Change propagation aspects may differ in the eastern coast of the Baltic Sea, which hosts larger rivers (e.g., Neva, Daugava) forming nutrient plumes that spread further towards the open sea. In the model scenario analysis of archipelagoes and bays, nutrient load reductions from local land-based sources and associated riverine loads (green arrows in Figure 5.1) influence a relatively limited coastal area. The effects are stronger in the vicinity of the source and decrease in the outer and less isolated coastal parts, with a sharper decrease for P than for N, as seen for the Archipelago Sea and Himmerfjärden Bay. The trend analysis of actual changes in land-based nutrient loads from monitored rivers along for the Swedish coast, including also larger rivers than in the model simulations, shows that only the changes in DIN loads propagate through the coastal waters. The model scenario analysis of improved regional open sea conditions on local coastal waters (purple arrows in Figure 5.1) shows that effects are strongest in the less isolated outer coastal waters but can also substantially improve the more isolated inner coastal waters (e.g. in Paper II for Winter DIP) and propagate to a wider coastal area than effects of local land-based reductions. These simulated propagation results are also consistent with results from Paper III in indicating that effects of improvements in open sea Summer Chl-a conditions spread to most coastal waters, except the most isolated ones, and changes in open sea Winter DIP affect all coastal waters along the Swedish coast.

The coast-sea coupling has implications for controlling the N:P ratio of nutrient loads flowing to coastal waters. The relative importance of N and P for managing eutrophication has been a long-standing debate, which has reached a consensus that N reductions are generally more important for coastal waters in the temperate zone and P reductions are generally less important for coastal waters than for lakes, due to lower coastal N-fixation by cyanobacteria and higher coastal P release from sediments (Howarth and Marino, 2006). Therefore, reducing loads of both nutrients is needed to manage eutrophication in the land-coast-sea continuum, as well as to account for variability of water quality conditions in coastal and inland waters (Howarth and Marino, 2006). In the Baltic Sea system, the importance of controlling both nutrients is exacerbated by the vicious circle (red dashed and dotted arrows in Figure 5.1; Vahtera et al., 2007) of N-fixation by cyanobacteria fuelling the spring bloom and increasing P internal loading from sediments. Thereby, N reductions are needed for most coastal waters (Papers II and III) and to decrease the N-limited spring bloom in the open sea (Elmgren and Larsson, 2001), while P reductions are needed for the most isolated bays (Paper II) and to limit cyanobacteria blooms by decreasing the P concentrations in the open sea (Elmgren and Larsson, 2001).

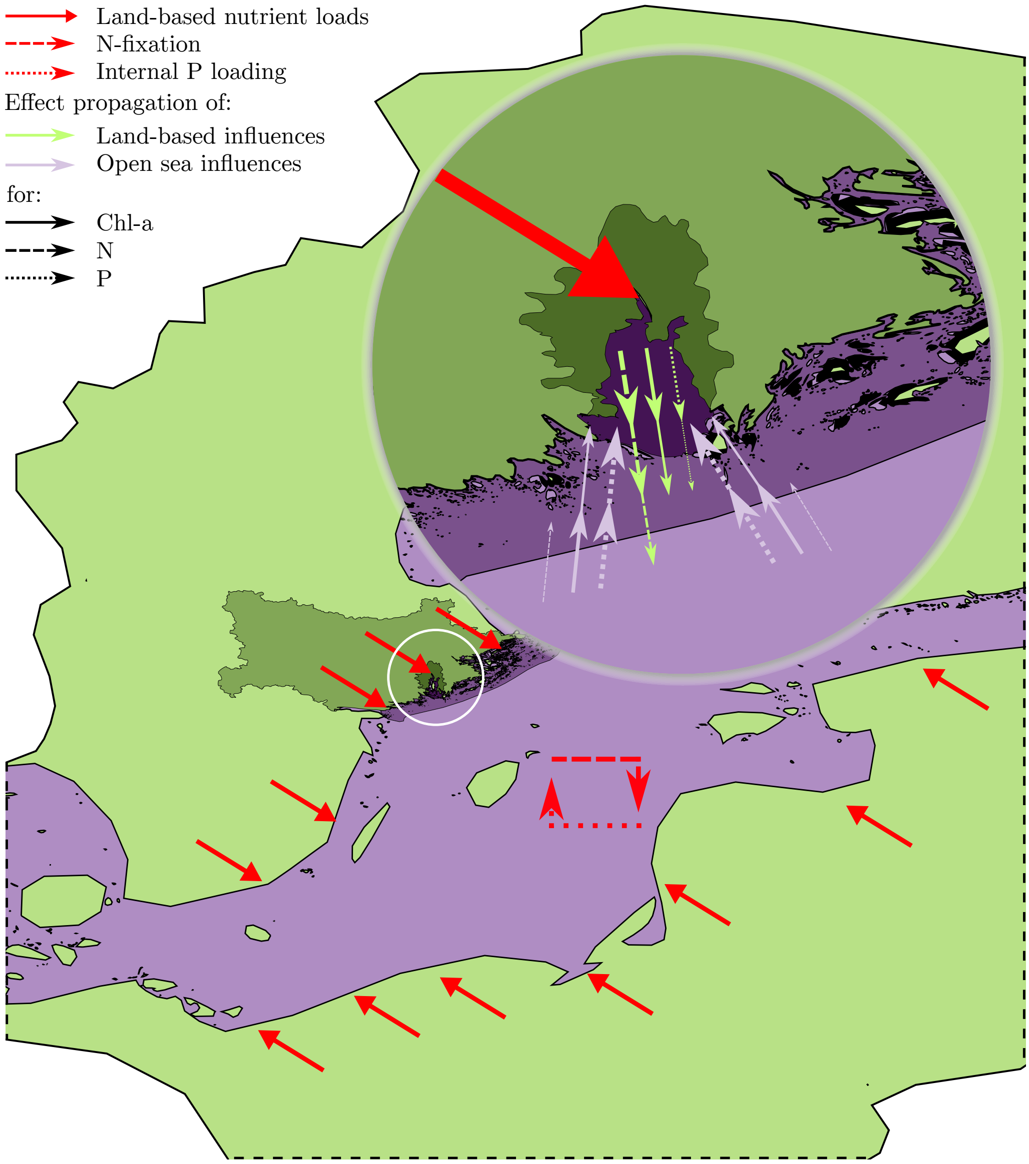

Figure 5.1. Schematic representation of influence propagations through the coupled regional scale, comprising the open sea (light purple) and its catchment (light green), local scale comprising a bay (dark purple) and its catchment (dark green), and intermediary scale comprising an archipelago (purple) and its catchment (green). Integrated land-based loads from the whole catchment influenced open sea conditions (plain red arrows) in addition to internal P loading (dotted red arrow) and N-fixation (dashed red arrow), which together forms the vicious circle. Influences of open sea conditions propagate to the archipelago and the bay (light purple arrows, plain for Chl-a, dashed for N and dotted for P). Land-based influences propagate from the bay towards the open sea (light green, plain for Chl-a, dashed for N and dotted for P).

Moreover, N needs to be managed in relation to P on local coastal and regional scales to increase the coastal filter efficiency (denitrification and sediment burial of P in the coastal wone) and avoid nutrient imbalances. The latter can lead to cyanobacteria blooms and increased export of N to the open sea, as indicated in Paper II, and negative changes in phytoplankton community that affect the food web (Elmgren and Larsson, 2001).

The found crucial role of coast-sea coupling for the coastal conditions is consistent with results for other parts of the Baltic Sea, such as the Gulf of Finland, where coastal conditions along the outer brink estuaries are found to be dominated by open sea conditions (Raateoja and Kauppila, 2019). This has also been found in other parts of the world, e.g., the Waitunna Lagoon (New Zealand), for which eutrophication conditions could be reduced by improving the connection with the sea (Jones et al., 2018) and the French Atlantic coast, where even non-eutrophic open sea conditions may greatly contribute to the inorganic nutrients in the coastal waters (Ménesguen et al., 2018). However, this substantial coastal influence of open sea conditions does not imply that local eutrophication management can just passively wait for open sea conditions to improve. Coastal waters with high recreational values may be situated in inner and isolated coastal waters, for which local land-based load reductions may have stronger coastal effects than open sea improvements (e.g., for the Himmerfjärden Bay, valued for its recreational activities and housing; Franzén et al., 2011). Moreover, open sea conditions depend on the integrated land-based loads from the total catchment and along the entire coastline of the whole Baltic Sea (Figure 5.1, red arrows). Thereby, local efforts to reduce the nutrient loads from all local coast catchments around the Baltic Sea are needed to improve conditions over the whole regional open sea, so that they can in turn improve local coastal conditions. Finally, the great importance of the coast-sea coupling challenges a simple unidirectional source-to-sea view of coastal waters as mainly influenced by and buffering local land-based nutrient loads before they reach the open sea. Such a view neglects the even stronger role that the open sea conditions, and management measures to improve them, play for local coastal eutrophication.

5.2 Drivers of coastal eutrophication

Hydroclimatic influences have been investigated in Paper II through pseudo-steady state simulations under distinct hydroclimatic conditions. These simulations do not consider the climate change and the associated coastal system evolution to these conditions. Transient simulations accounting for the hydroclimatic change implied by climate model projections would be required to account for this evolution. However, the use of climate model projections to drive local and regional hydrodynamic and water quality models is not straightforward. It requires identification of which global and regional climate model combinations (with the former driving the latter) perform well in driving the hydrological discharges and nutrient loads on land, and the hydrodynamics and water quality in the coasts and the open sea. Climate model results also need to be statistically corrected for bias and locally downscaled. In addition, accounting for climate-driven evolution requires long-term simulations of open sea water quality conditions, for which the coastal-focused water quality approach developed in this thesis is not well suited. Even though transient coastal simulations driven by climate projections could give valuable information, climate variability and change trends would also confuse the distinction of specific variable effects on coastal eutrophication in comparison to the relatively simple pseudo-steady state simulations used in Paper II.

In Paper II, using pseudo-steady state simulations, dry-warm hydroclimatic conditions with relatively low nutrient loads are found to complicate eutrophication management in the Himmerfjärden Bay. Under such climatic conditions, indicative of drier and warmer summers projected for the Baltic catchment (The BACC II Author Team, 2015), coastal eutrophication is maintained by increased prevalence of hypoxic areas and associated P recycling. These results are consistent with expected effects of warmer water temperatures projected to worsen coastal eutrophication conditions worldwide, also through increased algal growth and nutrient recycling (Glibert et al., 2014). Even though the Paper III study does not show increased water temperatures in coastal waters as being directly associated with rising Summer Chl-a conditions, earlier thawing of sea-ice is found to favour increased Summer Chl-a, possibly through modification of winter nutrient and spring blooms dynamics or by aggregated winter and spring hydroclimatic conditions. In addition to these effects, projected increases in precipitation may also increase nutrient loads to coastal waters and further complicate reaching the BSAP and WFD management targets (Bring et al., 2015). This is indicated in Paper III by the strong positive correlation between Summer Chl-a and increase in DIN loads for all Swedish coastal waters, in consistency with other results for possible colder and rainy summers (Golubkov and Golubkov, 2019). However, on the Baltic Sea scale, simulations by Saraiva et al. (2019a) indicate that, regardless of projected climate change scenarios, reaching the BSAP nutrient reduction targets will considerably improve eutrophication conditions by the end of the century and lead to important decreases in P concentrations, N-fixation and hypoxic area. Such improvements are expected to also propagate to most coastal waters in archipelagoes and bays, especially if they are complemented by site-specific local nutrient load reductions, as indicated in Papers II and III.

5.3 Gaps in coastal eutrophication research

Paper IV has shown that many Baltic coastal eutrophication studies focus either on drivers or on impacts, with few investigating both together. This is an important gap since ecological changes, both from eutrophication itself and in synergy with other pressures, can in turn feed back and further affect the coastal eutrophication (Malone and Newton, 2020). For example, overfishing, which targets predatory fish, has led to increase in pelagic fishes preying on zooplankton, thereby decreasing the top-down control on phytoplankton and thus acting in synergy with the bottom-up nutrient load control to increase eutrophication (Eriksson et al., 2009; Bergström et al., 2019). Increased mussel beds, on the other hand, filter phytoplankton and organic matter, reducing turbidity and providing a negative feedback to eutrophication. The lack of research consideration of the eutrophication feedbacks of ecosystem impacts is also a limitation of this thesis. In Papers I and II, the water quality model uses a temperature dependent rate for phytoplankton grazing, which represents the entire food web from zooplankton and higher trophic levels and thereby does not dynamically account for feedbacks of changes in coastal ecological conditions. Ecological impacts have not been investigated in Paper III either, in order to focus on physical and biogeochemical drivers, but also because distinguishing between drivers and feedbacks of coastal eutrophication would require more advanced statistical analysis that may not be supported by the limited available data. Studies that do consider ecosystem feedbacks on eutrophication are over-represented in the 10% most cited studies (Paper IV, SM Figure 1), which indicates both an interest and a need for research linking the drivers, impacts, and feedbacks of coastal eutrophication.

The lack of studies focusing on connecting the drivers with the impacts on and the feedbacks from ecosystems is further exemplified by the decoupling in research between human aspects, ecological components, and natural remediation (filter function) of the coastal zone that has been found in Paper IV. For example, papers on human aspects tend to take into account management cost and acceptance aspects (e.g., Ahtiainen et al., 2014; Elofsson, 2012), but commonly without considering potentially important influences and feedbacks from ecological and natural processes (e.g., hydroclimate; Boesch, 2019). The natural remediation performed by the coastal filter links human pressures on coastal waters to the natural nutrient sequestration performed by microbial and benthic communities (Carstensen et al., 2020; Thoms et al., 2018). However, very few studies consider the natural remediation together with other coastal functions, even though the coastal filter efficiency depends on ecological communities in addition to biogeochemical conditions. Moreover, very few studies consider the natural remediation function of the coastal filter together with other management aspects, even though it influences the transport and loading of nutrients to other coastal waters and to the open sea and thus the efficiency of other management measures (Almroth-Rosell et al., 2016; Gren, 2013). The research decoupling between the human, ecological and natural remediation aspects results from disconnected scientific fields (e.g., social, hydrological and marine sciences; Boesch, 2019), which yields a fragmented understanding of these aspects and their interactions with each other and fragmented management goals. Tools that allow linking these aspects can improve understanding of processes and quantify the connections between different aspects (e.g, fine-scale ecological models; Skov et al., 2020), while tools that allows consistent monitoring across the ecological and human dimensions may help identify and linkages between pressures and between management goals (e.g., Baltic Health Index; Blenckner et al., 2021).

5.4 Scenarios and measures for coastal eutrophication management

The management scenarios investigated in Papers I and II assume reductions of land-based nutrient loads and sea-based nutrient concentrations. However, reaching land-based nutrient load reductions is not straightforward due to the high share of nutrient loads from diffuse legacy sources accumulated in soil, sediments and slow moving groundwater (Chen et al., 2021; Destouni and Jarsjö, 2018) in the Baltic catchment. These diffuse legacy sources have long residence and travel times (Juston et al., 2016; McCrackin et al., 2018), estimated to 30 years for P on average in the whole Baltic catchment. Thereby, land-based measures over the last two decades have not considerably reduced nutrient loads to the sea (Destouni et al., 2017).

Furthermore, simulating nutrient load reductions under different hydroclimatic conditions is also challenging. In Paper II, the same absolute riverine load reduction has been used under wetter and drier conditions to compare the effect propagations of similar load reductions between point and riverine sources. This assumes decreasing efficiencies of management measures under wetter conditions, which could be obtained by measures targeting fast-flowing surface nutrient loads (McCrackin et al., 2018) but without offsetting nutrient mobilization and transport from legacy sources under wetter conditions. Measures that specifically target legacy sources, such as ditches, buffer zones, wetlands, are needed to trap more nutrients as loads from those sources increase with increasing water flows. For the open sea, reducing nutrient concentrations is also not straightforward as such reductions depend not only on the effectiveness of land-based measures, but also on various internal sea processes that maintain and accentuate eutrophication (Stigebrandt, 2018).

The shift from nutrient loads dominated by point sources to loads dominated by legacy and other diffuse sources constitutes a paradigm shift for coastal eutrophication management (Chen et al., 2021; Destouni and Jarsjö, 2018; Le Moal et al., 2019). This shift has begun to be acknowledged in management with the recent adoption of the Fertilizing Products Regulation by the EU in 2019 (Huygens et al., 2019). Improvement of agricultural practices to reduce currently active fertilizer application and leaching are nevertheless crucial for coastal and open sea eutrophication management, since such improvements dictate the faster-flowing diffuse nutrient loads as well as future legacy sources, and more broadly for reaching sustainable development goals linked to food production (target 2.4; HELCOM, 2018a) and sustainable P use (Nedelciu et al., 2020). However, such measures are unlikely to be sufficient for a quick recovery of coastal and open sea eutrophication in the Baltic Sea, as residence time of P in the Baltic Sea itself is estimated to be around 30 years (Saraiva et al., 2019b). Thereby, nutrients accumulated in the open sea and coastal sediments also act as diffuse legacy sources within the Baltic Sea that further fuel coastal and open sea eutrophication. Therefore, lengthy recovery time-scales can be expected from land-based source remediation measures, and needs to be acknowledged in the application of the WFD (Carvalho et al., 2019) and other management frameworks.

Management solutions that can enhance the efficiency of the coastal filter could be useful to trap indiscriminately both current and legacy source loads. Such measures are under-investigated and may include nature-based and/or geoengineering solutions. The latter generally target reduction of the release of legacy P stored in sediments and have been successfully applied in some isolated bays of the Baltic Sea (Rydin et al., 2017), but may also present substantial risks (Conley et al., 2009a). Nature based solutions can instead target ecosystem improvements that reduce eutrophication symptoms, such as limiting fishing of predatory fish to increase phytoplankton grazing, but also nutrient capture solutions, such as mussel and seaweed farming (Kotta et al., 2020; Hasselström et al., 2020). Moreover, coastal wetlands may reduce nutrient loads leaching to the sea, as well as coastal nutrient concentrations (Berthold et al., 2018), while also providing habitat for predatory fishes and promoting their recruitment (Nilsson et al., 2014). Coastal management solutions may be useful complements to land-based measures that could facilitate reaching the BSAP nutrient reduction targets, necessary to decrease open sea and in turn also coastal eutrophication.

6 Conclusion

This thesis has investigated relationships between coastal eutrophication and hydroclimatic, hydrospheric, and land-based drivers and feedbacks, and their effect propagations through the Baltic land-coast-sea system and its multiple associated scales. It has also assessed and identified important gaps in how published research on the Baltic coastal eutrophication considers and handles the various system components and their coupling. Main conclusions relating to each of the thesis objectives are summarized as follows:

Objective A:

- A relatively simple characterization approach for scalable modelling of water quality dynamics has been developed and applied to the regional open Baltic Sea scale, the local coastal Himmerfjärden Bay scale, and the intermediate Archipelago Sea scale. Water quality is represented by a carbon-based biogeochemical model driven by results from three-dimensional hydrodynamic modelling. Validation of the model shows good model performance for surface waters and moderate to poor performance for deeper waters.
- Numerical simulations for the Archipelago Sea show that reductions of open sea eutrophication can improve coastal water quality and eutrophication conditions, especially for the outer part of the Archipelago, while land-based measures have more localised effects that are also more pronounced for nitrogen than phosphorus. On the local coastal scale of the Himmerfjärden Bay, combined sea-based and land-based measures are needed for reaching good status for the DIN, DIP and Chl-a water quality components.

Objective B:

- Trend analysis for the coastal waters along the Swedish coastline further emphasizes the importance of both land-catchment and sea-based conditions for coastal eutrophication. For less isolated coastal waters, trends in Summer Chl-a are strongly correlated with those in the open sea. For all coastal waters, trends in Summer Chl-a are also significantly and strongly correlated with land-based riverine DIN loads, while coastal Winter DIP trends are strongly correlated with those in the open sea.
- Trend analysis distinguishes local internal dynamics of Summer Chl-a, dominated by nitrogen for less isolated coastal waters, which exhibit similar relationships as in the open sea, and more mixed dynamics with somewhat stronger phosphorus influence for the more isolated coastal waters.
- Simulations for the Himmerfjärden Bay show that coastal eutrophication is easier to mitigate under drier-colder conditions and harder to mitigate under wetter-warmer conditions. Hydroclimatic conditions thus emerge as a key non-human driver that can strongly affect the effectiveness of management measures for coastal eutrophication, and may complicate management under projected wetter and warmer conditions for Sweden and the Baltic region.

- For the Swedish Baltic coastal waters, trends in Summer Chl-a concentrations are strongly correlated with trends in sea-ice conditions, which may represent a proxy for winter and spring hydroclimatic conditions of relevance for eutrophication. In the open sea, trends in Summer Chl-a conditions over the last 30 years are strongly correlated with wind speed, salinity and precipitation trends, and moderately correlated with trends in freshwater nutrient concentrations.
- Trend analysis and simulation results combine in highlighting that the influences on coastal waters of a mix of coupled and competing drivers from land, coast, open sea, and atmosphere. Nevertheless, useful information for coastal eutrophication management can be obtained by classifying the coastal waters based on their water exchange rates with the open sea, which can distinguish different dynamics and driver responses between more and less isolated coastal waters.

Objective C:

- Investigation of coastal eutrophication research on the Baltic Sea system shows that the main system components of drivers, management, and ecosystem impacts of coastal eutrophication are in themselves well-investigated in the published literature.
- The human aspects, the natural remediation (coastal filter) function, and the ecological eutrophication feedbacks of the coupled land-coast-sea system, however, are not well-connected, and the driver and ecological feedback coupling of coastal eutrophication is often under-investigated in research, and also in this thesis.
- Dual-nutrient management strategies (N and P) and specific management measures that can also be effective against legacy sources are also under-investigated in Baltic coastal eutrophication research.

7 Perspectives for future work

Long-lived legacy sources on land, which have emerged as contributors of a large proportion of the nutrient loads to coastal waters, have not been much accounted for in coastal eutrophication research so far, as shown in Paper IV. Coastal eutrophication management requires distinguishing between legacy sources and other diffuse and point nutrient sources with faster coastal load contributions, for which source reductions would lead to relatively quick decline in nutrient loads (McCrackin et al., 2018). Methodological developments are needed to both characterize land-catchments in terms of their nutrient source distributions (e.g., through data analysis; Chen et al., 2021) and locally pinpoint the nutrient origins (e.g., through high-spatial-resolution monitoring; Dupas et al., 2021).

Paper IV has also revealed that few studies on coastal eutrophication actually analyse the coastal influences from land. Such land studies are however needed to estimate likely recovery times of legacy sources of nutrient from land, and how these interplay with the eutrophication recovery times along the whole land-coast-sea system. Moreover, improved understanding of recovery times on land can facilitate the definition of more realistic local and regional nutrient load management scenarios. Relatively simple coastal-focused modelling approaches, such as the one used in Papers I and II, can then be used to screen the management scenarios and identify well-performing measures, as well as facilitate stakeholder involvement for implementing them.

In addition to measures on land, practical coastal management solutions, like mussel and seaweed farming, coastal wetlands and geoengineering, may help to combat eutrophication by improving the efficiency of the coastal filter and being effective also against legacy sources (Boesch, 2019). Such coastal management solutions have received little attention in research on Baltic coastal eutrophication, as shown in Paper IV. Therefore, more research is needed to quantify their potential for nutrient load removal (e.g. for wetlands; Arheimer et al., 2004; Jansson et al., 1998), both through experiments and modelling studies. Moreover, the coastal measures impacts on the whole ecosystem health also need to be further investigated and, if found suitable, policy instruments need to be developed and applied to incentivise their implementation (e.g., compensation for nutrient removal; Schernewski et al., 2019, industry development; Hasselström et al., 2020).

Improving coastal eutrophication conditions and ecosystem health requires understanding of the drivers and impacts of coastal eutrophication, but also the feedbacks of the ecosystem impacts back to eutrophication, which are understudied (Paper IV). Thereby, more studies investigating drivers and impacts together can further our understanding of driver-impact trade-offs and synergies, by e.g., investigating the interlinkages between eutrophication and ecological communities, and the relationships between top-down and bottom-up drivers and pressures. Data analysis methods that can identify potential thresholds and tipping points to shifted ecosystem states, and fine-scale ecosystem models that can help to assess local and regional impacts of ecosystem changes (Skov et al., 2020) are thus needed to support coastal recovery from undesirable eutrophic states.

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